A policy-enabling framework for the ex-ante evaluation of marine protected areas

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Abstract

A marine protected area (MPA) potentially generates a wide range of consumptive use, non-consumptive use and non-use values that include: critical habitat protection, conservation of marine biodiversity, recovery of threatened and endangered marine species, increased recreational benefits and increased biomass of harvested marine species. To help assess whether such values exceed the potential costs of MPAs, this paper provides a policy-enabling framework that reviews the existing theoretical and practical instruments and approaches that can be used in the ex-ante evaluation of MPAs. This framework is in three parts and identifies the factors that are relevant to understand the benefits and costs associated with the establishment of a MPA. First a range of alternative monetary and non-monetary techniques to estimate three key economic benefits of MPAs: consumptive, non-consumptive use and non-use values are presented. Second, three decision protocols that can be applied to determine the desirability of establishing MPAs are described. Third, caveats of these approaches and the need to accommodate the social needs of the communities are provided. The framework shows that biological and ecological considerations together with economic viability and socio-economic factors can and should be taken into account when deciding about when and where to establish MPAs and of what size.

Key words: Marine Protected Areas, use value, non-use value, benefit-cost analysis, ex ante evaluation

1. Introduction

Marine protected areas (MPAs) are considered to be the cornerstone of marine biodiversity conservation policies (TEEB 2010). A MPA is commonly defined as "…any area of intertidal or sub-tidal terrain, together with its overlying water and associated fauna, historical and cultural features, which has been reserved by law or other effective means to protect part or all of the enclosed environment" (Kelleher 1999). The characteristics of a MPA vary within a wide range of spectrum (Agardy et al. 2003). 'No take' areas, or Category I or II zones managed mainly for science, wilderness for ecosystem conservation and recreation under the World Conservation Union's Guidelines, are locations where no harvesting is permitted (Eagles et al. 2002). Such zones often form a part of larger MPAs where there may be multiple-use areas, or Category VI zones, that allow for some consumptive use, and are managed for the sustainable use of natural ecosystems.

At the 10th Conference of the Parties (COP 10) of the Convention on Biological Diversity held in Nagoya, Japan on October 2010, environmental leaders from 193 countries agreed to extend the share of MPAs from less than 1 per cent to 10 per cent by 2020. It has been estimated that 20 to 30 per cent conservation of global oceans through a network of MPAs could create a million jobs and sustain US$70-80 billion/year worth of marine fish catch (Balmford et al 2004). MPAs also help conserve threatened, endangered and rare marine species and increase recreational opportunities. Empirical evidence of the benefits of MPAs in fisheries, especially for overexploited species, is supported in various case studies synthesized by Gell and Roberts (2002). Côté et al. (2001) in a meta-analysis of 19 MPAs show that abundance of targeted fish species was 28 per cent higher within such areas. Such benefits, at least for some MPAs, have spilled over to adjacent exploited areas as evidenced by increased catches per unit of effort and increased population size in these areas (Gell and Roberts 2003, Roberts et al. 2001), in addition to harvests of larger and often higher valued individuals (Bhat 2003).

These positive payoffs, however, must be set alongside any potential costs that may arise from a lack of access to fishing grounds, increased fishing pressure on stocks outside of protected areas, other harvesting costs or increased management costs. These possible losses could, for example, include higher fuel costs to harvest fish when fishing outside of traditional fishing areas. Furthermore, it has been argued that MPAs may negatively influence the lives and livelihoods of the communities that are dependent on marine resources by restricting their access to the quantity and type of tangible and intangible benefits that flow from marine ecosystems (Mascia et al. 2010). A large and growing number of empirical evidence suggests that the alternative livelihood opportunities, such as tourism, within a MPA not only do not benefit local communities, but in some cases may exclude them by restricting their access to marine resources in the protected areas (Rosendo et al. 2011).

The benefits and costs of a MPA vary depending on the biological, ecological, socioeconomic, cultural and institutional conditions of the site (Agardy et al. 2003). The site-specific benefits and costs need to be carefully assessed ex ante to understand the justification of establishing a MPA. This is becoming increasingly important given concerns over the success of MPAs (Lowry et al. 2009) and their possible adverse impacts on livelihood vulnerability and poverty of local communities (McClanahan et al. 2009). A number of different approaches are currently in use to evaluate existing or potential MPA management alternatives (see for example Alder 2002;

Dalton 2003). However, these approaches do not formally account for the marketed and non-marketed benefits and costs of a MPA arising from consumptive, non-consumptive and non-use values.

In this paper we provide a policy-enabling framework to assist decision makers to understand the existing theoretical and practical instruments and approaches that may be used in the ex-ante evaluation of MPAs. Some of the methods we describe require specialist training that may not currently reside within fisheries management agencies, but a clear understanding of what tools and approaches are available allows decision makers to make informed choices about what should be evaluated, and how, prior to the establishment of a MPA. Our intention is not to replace or substitute for scientific analysis (Sobel and Dahlgren 2004) or the need to include uncertainty and stakeholder engagement (Grafton and Kompas 2005) into the evaluation of MPAs. Instead, we aim to complement these approaches by providing a socio-economic framework of analysis that will allow decision makers better understand (1) the factors that are relevant to measure the benefits and costs associated with the establishment of a MPA, (2) the monetary and non-monetary instruments that are employed to estimate these benefits and costs, (3) the monetary and non-monetary decision-making approaches to determine the desirability of a MPA and (4) the limitations of these approaches in capturing the welfare impacts of MPAs on stakeholders. Although our framework focuses on the quantifiable benefit-cost components of a MPA, we acknowledge that a range of qualitative social benefits (e.g. increased livelihood opportunities) and costs (e.g. social displacement) can be associated with the establishment of a MPA. We also stress that social capital, community preferences and governance structures in fisheries are critically important to successful marine fisheries management (Charles 2001;

Grafton 2005; Gutiérrez et al. 2011) and these considerations should also be part of an overall policy-enabling framework.

In section two we review the possible quantifiable values of MPAs, in section three we present instruments to measuring changes in consumptive values with MPAs while section four reviews the instruments for evaluating changes in non-consumptive use and non-use values. In all cases, examples are provided, where available, to illustrate the instruments. Section five describes three possible decision protocols to determine whether a MPA should be established and, if so, what form should it take. In section six we provide concluding remarks.

2. Total Economic Value of MPAs

Figure 1 presents a summary of values that can be generated from a MPA. The total economic value of MPAs consists of use and non-use values. The use values include both consumptive (such as fishing) and non-consumptive uses obtained from direct use of species for recreational purpose, such as whale watching or marine wildlife viewing activities. Non-consumptive use values arise from activities that do not subtract from or diminish the quality of the environment. In terms of non-consumptive values, MPAs can increase aesthetic and recreational values because of higher population densities and/or larger individuals both within no-take areas and adjoining areas (Bhat 2003). Bohnsack (1998) summarizes these values under three headings:

1. Protect ecosystem structure, function and integrity;
2. Increase knowledge and understanding of marine systems; and
3. Improve non-consumptive opportunities.

The first category of value (protect ecosystem structure, function and integrity) refers to MPAs role in protecting physical habitat structure from fishing gear and other anthropogenic impacts, restoring population size and age structure, maintaining food web and trophic structure and so on. Examples of the second category of values (increase knowledge and understanding of marine systems) of MPAs include long-term undisturbed monitoring sites, availability of experimental sites needing natural areas and availability of natural reference areas for assessing anthropogenic impacts. The third category, to improve non-consumptive opportunities of MPAs, refers to the possibility of enhanced and diversified economic opportunities and social activities, enlarged aesthetic experiences and spiritual connection to natural resources, higher opportunities for recreational activities, wilderness experiences and so on.

Another important value of MPAs is their indirect use value. It represents the value of ecosystem services associated with species conservation and habitat protection. It includes MPA benefits such as enhanced ecosystem resilience that might arise from reduced habitat damage (Turner et al. 1999), an increased ability to assist in ecological cycling, the contribution of the endangered species to surrounding habitats or ecosystems. Non-use values of a MPA arise from conservation of threatened, endangered and rare marine species. They are the benefits obtained without any direct or indirect use and consist of two components: existence value and bequest value. Existence value reflects benefits from knowing that the species protected by a reserve exists, even if it is never utilized or experienced (Loomis and White 1996). Bequest value refers to benefits from ensuring the ecosystem services of MPAs are available for future generations (Moran and Pearce 1994).

The non-consumptive use value of a MPA is likely to increase the larger are the recreational opportunities within the MPA. A meta-analysis by Brander et al. (2007) showed that the geographical location of the MPA and its size have significant positive impacts on the recreation value of the reef. Wallmo and Edwards (2008) observed a diminishing marginal utility or values for MPA sizes. They found in their particular study that smaller reserves with 'liberal use' policies produce the largest increases in utility. Edwards and Gable (1991) showed that distance to the beach and the quality of the marine ecosystems both have a strong impact on property values around the MPA.

3. Instruments for Assessing the Costs and Benefits of Consumptive Use from MPAs

3.1. Non-monetary Instruments

The emergy synthesis method introduced by Odum (1996) is one of the most commonly used non-monetary instruments to evaluate environmental policies. The method relies on intrinsic value of a resource rather than relying on consumer preferences. It is based on the principle that the amount of energy embodied in a resource determines its value (Angelo and Brown 2007). Emergy is defined as the sum of all energy (usually solar energy) that is used up directly or indirectly in a process to deliver an output (Odum 1996). The emergy synthesis method determines the ultimate amount of solar energy, which is called solar emergy Joules (sej), embodied in each type of energy, material or currency used to operate the system of interest (Odum 1988). The difference between the emergy costs of an output and its contribution to the environment is called its net emergy (Brown and Ulgiati 2004). In principle, an output should contribute emergy to a system at least equal to the cost (emergy required) of obtaining it. If the emergy cost is greater than its contributions, the system is not competitive with one that gets more ‘net emergy’ from its sources. The emergy synthesis method has been applied to evaluate a wide range of environmental policy interventions including the evaluation of MPAs (see for example Franzese et al. 2008).

3.2. Monetary Instruments

The most direct way of evaluating the consumptive use value of a MPA using a monetary instrument is to calculate the expected losses in terms of gross value of production to fishers from the establishment of ‘no take’ areas. Although this approach involves the least amount of complexity, it may potentially overestimate the consumptive losses with establishing a MPA. The other deficiency of this approach is that it ignores the potential benefits of MPAs that may arise from spillovers, or the offsetting payoffs from fishing in a different area. We outline three alternatives instruments below that are superior to the gross value of production approach as they allow accounting for spillover benefits, uncertainty and fishers’ behaviors.

3.2.1 Bioeconomic Models

Bioeconomic models form the core of fisheries economics and combine measures of revenues and costs with an underlying biology, or stock-recruitment relationship. To capture the full impacts of MPAs, bioeconomic models must be stochastic, account for ‘normal’ uncertainty, or the usual fluctuations in stock and harvest in a fishery, as well as ‘unusual’ events that may more dramatically affect the fishery over time.

Grafton et al. (2006) have developed an approach that incorporates uncertainty into bioeconomic models of marine reserves. They show that MPAs increase resilience and allow for quicker recovery following a negative shock that benefits fishers. In other words, even if harvesting is optimal, the population is persistent and there exists no uncertainty over the size of the current population, a MPA can increase economic profits and reduce the recovery time for a harvested population in the presence of negative shocks. Grafton et al. (2009) have applied this approach in the context of the Northern cod fishery of Canada and calculate that a marine reserve with optimal harvesting would have generated returns of some C$2 billion (in 1991 prices) more than what actually occurred over the period 1962-1991. Even with optimal harvesting they find that a reserve would have generated extra payoffs of some C$162 million.

3.2.2 Effort Displacement Models

An important cost issue of MPAs is the reallocation of fishing effort from areas where fishing I restricted or prohibited. This reallocation would be expected to change the average value of landings and costs, especially if the stock abundance is not constant across the fishery. Two approaches that can be used to estimate fishing effort displacement are the stochastic frontier method and the random utility method. Both require individual vessel level data to generate suitable estimates of the impact of MPAs on fishers.

The frontier approach imposes no a priori assumption about fisher behavior and simply uses spatial catch and effort data and individual fisher characteristics to model the impact of spatial closures on effort and catches. This approach can also be used to estimate the impact on costs and profits if there are adequate economic data at an individual vessel level. The method requires the statistical estimation of a model, such as the one shown below in equation (1):

$$\ln V_{ijt} = \beta_0 + \beta_1 \ln K_{ijt} + \beta_2 \ln Effort_{ijt} + \beta_3 (\ln K_{ijt})^2 + \beta_4 (\ln Effort_{ijt})^2 + \beta_5 \ln K_{it} \ln Effort_{ijt} + \sum_{m=2}^{12} \beta_m D_m + v_{ijt} - u_{ijt} \quad (1)$$

where V_{ijt} is the total value of landings in area *i* by vessel *j* at period of time *t*, K_{ijt} is a measure of boat capacity or engine power of the vessels *j* operating in region *i* at time *t*, $Effort_{ijt}$ is the total calculated effort in region *i* by vessel *j* at time *t*, D_m is a dummy variable representing each month in the fishing year, v_{it} is a stochastic error term and u_{it} is a an error term representing the 'inefficiency' associated with region *i* at time *t*. The u_{it} error term might be further parameterized to include specific management devises, such as input restrictions, on individual fisher performance.

Estimates of the effort displacement equation before the introduction of MPAs would provide information on the effect of the value of landings in each area, conditional on seasonal abundance and major inputs into fishing. This would allow decision makers to build a spatial picture of a fishery to indicate what changes in spatial fishing patterns and revenues might be realized with the introduction of MPAs.

The random utility modeling imposes particular assumptions about fisher behavior to model effort displacement. The approach models a sequential set of economic decisions made by fishers to determine whether they should go fishing and where they should fish (Wilen et al. 2006). Assuming fishers are motivated by net returns, it is possible to show how effort changes with the establishment of marine reserves. Schneider (2006) has used this approach to predict the redistribution of fishing effort by divers for abalone in New Zealand following the creation of a network of no-take areas. The method has also been used to estimate the effort displacement associated with the closure of the European anchovy (*Engraulis encrasicolus)* fishery (Vermard et al. 2008).

3.2.3 Stock-adjusted Productivity

Productivity represents the ratio between outputs and inputs and is a key indicator of economic performance. An understanding and measurement of productivity of fishers is useful in assessing the impacts of MPAs.

Explaining productivity performance, or understanding the causes of declines or increases in productivity by vessel and over time, is as important as measuring it. An easy-to-apply method is available that 'decomposes' changes in relative profit performance into differences in output prices and input prices, adjusted for their importance in the catch (outputs) and fishing effort (variable inputs), and fixed inputs, such as vessel size (Fox et al. 2003). The approach can also be used to account for spatial differences in productivity and, thus, assess the impacts of MPAs while explicitly accounting for changes in prices and fish stocks. Unlike the effort displacement approach, profit decompositions are not a statistical method and inferences are not based on a probability distribution. Instead, the decomposition method generates indexes to make comparisons across vessels and over time.

Fox et al. (2006) have used this method to evaluate the effects of a structural adjustment package in a fishery in terms of productivity in the south-east trawl fishery of Australia. Using data from 47 vessels over the period 1997-2000 they developed a stock-adjusted productivity index after accounting for changes in input and output prices and also changes in an aggregate stock index.

They find that productivity's contribution to profits increased following a buyback and removal of fishing capacity from the fleet, despite a fall in the over stock abundance.

4. Instruments for Assessing the Benefits of Non-Consumptive Use and Non-Use from MPAs

Non-consumptive use and non-use benefit assessments can be undertaken using either monetary or non-monetary valuation instruments. The monetary valuation techniques exploit quantitative information available from primary and/or secondary sources while the non-monetary techniques primarily rely on qualitative measures obtained through stakeholder participation and in group discussions. The former techniques are widely known as non-market valuation techniques in the environmental valuation literature (See Box 1 for further discussion).

INSERT BOX 1 HERE

4.1. Non-market Valuation Techniques

4.1. Travel Cost Method

In a travel cost method, an analyst first estimates a demand function for recreational travel by accounting for monetary and non-monetary expenditures related to recreational travel. The demand function relates to the number of visits that users/travelers/tourists make to the travel cost incurred, site characteristics, socioeconomic characteristics of the user population and substitute site information. The demand function can be written in the following form:

$$\text{TRIP}_i = \beta_0 + \beta_1 (\text{TRIPCOST}_i) + \beta_2 (\text{TRIPCOST\_SUB}_i) + \beta_3 (\text{SOCIO\_DEMOG}_i) + \beta_4 (\text{SITE\_SPECIFIC}_i) + \varepsilon_i \qquad (2)$$

where TRIP_i is the number of trips by individual i to the site over a specific time period, TRIPCOST_i refers to the cost of round trip to the site incurred by each individual i. The variable TRIPCOST_SUB_i represents the costs of trips to substitute sites, SOCIO_DEMOG_i denotes socio-demographic characteristics such as age, income, gender, education, of the traveler and SITE_SPECIFIC_i refers to the recreational facilities offered by the site such as swimming, diving, fishing. The βs are regression coefficients and ε stands for random error.

After the demand function is estimated based on available data, estimates of the consumer surplus can be obtained by calculating the area below the demand function and above the implicit price from visiting the site so as to obtain a traveler's willingness to pay to visit the site. The consumer surplus (CS) for an average sample visitor can be calculated by integrating the travel demand function, given in Equation 2, from an initial travel cost (TRIPCOST= TRIPCOST_0) to the choke price (TRIPCOST= TRIPCOST_M) at which the demand to visit the site becomes zero:

$$CS = \int_{TRIPCOST_0}^{TRIPCOST_M} (TRIP)\ (TRIPCOST)\,dx \qquad (3)$$

The average visitor consumer surplus can be aggregated over the total tourist population by multiplying by the number of visitors to a site each year. The aggregated amount provides an estimate of total non-consumptive use value obtained by the stakeholder.

Carr and Mendelsohn (2003) have used the travel cost method to estimate the recreational value of the Great Barrier Reef located in Australia to its two million visitors each year. Over 600 visitors from 39 different countries were interviewed in the year 2002. The study found that the annual recreational benefits of the Great Barrier Reef range between US$700 million to 1.6 billion in 2002 prices. Discounting this annual benefit at 4 per cent per year suggested that the Great Barrier Reef is worth between US$18 and 40 billion.

4.2. Hedonic Pricing Method

The hedonic pricing method assumes that consumers' valuations of a good depend upon a number of characteristics embodied within the good (Rosen 1974). By obtaining measures of these characteristics and incorporating them into a regression model, consumers' willingness to pay for each individual attribute of the good can be estimated. The hedonic price function can be expressed in the following form:

$$P = \beta X_i + \varepsilon \qquad (4)$$

where P is the market price of the good in question, X_i is a vector of attributes of the good, β is a vector of the parameters of the hedonic model to be estimated and ε is the error term. In the case of an *ex ante* evaluation of a MPA, the price could refer to prices of the real estate properties adjacent the sea and the vector of attributes could include property owners' socio-economic characteristics and the attributes of the marine environment.

Edwards and Gable (1991) examined the relationship between coastal property values and recreational opportunities at local public beaches by applying the hedonic pricing method in a small coastal town in the USA. They exploited a time series data set containing information of over 300 property transactions during the years 1979 to 1981. In addition to each property's distance from the nearest local public beach, the regression model controlled for the effects of the other major coastal resources (e.g. water frontage, water view, and distance from a coastal lagoon) on property values as well as the effects of structural attributes such as the number of bathrooms and floor spaces. The consumer surplus of beach recreation enjoyed by local users ranged from US$1,788 for a household with a US$10,000 annual income living 10 miles from a public beach to US$46,706 for a household making US$60,000 and living only 0.5 miles from a public beach. The average annual consumer surplus was about US$469 per person for all forms of beach recreation (e.g. swimming, sunbathing, surf fishing, winter walks) throughout a year.

4.3. Contingent Valuation Method

Contingent valuation method is used to estimate willingness to pay for an action, or the monetary amount or hypothetical payment by an individual required to ensure that she or he is as well off in utility or welfare terms after the provision of a desirable good or service as before. To calculate willingness to pay, individual welfare is represented by utility functions that are used to estimate how much utility (or satisfaction) an economic agent derives from consumption of different goods or services. In the case of a MPA designed to protect marine mammals from extinction, the utility functions could be written in the following form:

Without MPA: $$V^0 = \alpha X + \beta Y + \lambda\, MM^0 + \varepsilon^0 \qquad (5)$$

With MPA: $$V^1 = \alpha X + \beta(Y - WTP) + \lambda\, MM^1 + \varepsilon^1 \qquad (6)$$

In equations 5 and 6, V^0 is the base line utility function without the MPA. The individual is given the choice of paying a monetary amount (which reflects their willingness to pay) to finance the MPA that will protect the marine mammals from extinction. V^1 describes the new (and higher) utility function after implementation of the MPA. The term *MM* stands for the marine mammal species status, *Y* denotes income, *X* is the vector of individual-specific attributes affecting utility while α , β and λ refer to the regression coefficients, and ε is a random error term. The change in utility due to the proposed policy intervention is obtained by subtracting Equation (5) from Equation (6), that is,

$$V^1 - V^0 = \alpha\ X - \beta\ WTP + \lambda\ (MM^1 - MM^0) + (\varepsilon^1 - \varepsilon^0) \qquad (7)$$

By definition, the individual willingness to pay is an amount that makes $(V^1 - V^0) = 0$.

This implies:

$$\alpha X - \beta\ WTP + \lambda\ \Delta MM + \varepsilon = 0 \qquad (8)$$

where, $\Delta MM = MM^1 - MM^0$ and $\varepsilon = \varepsilon^1 - \varepsilon^0$

and which simplifies to:

$$WTP = \frac{1}{\beta}[\alpha X + \lambda \Delta MM + \varepsilon] \qquad (9)$$

To obtain an estimate of equation (9), respondents are generally asked to pay a pre-specified bid amount by creating a hypothetical situation to survey. Estimation of the probability that individual respondents say ‘Yes’ (accept the bid level) or ‘No’ (rejects the bid level) is undertaken as a function of the offered bid level and a set of theoretically expected explanatory variables. Mean WTP per respondent per year is estimated using Equation 9 and then aggregated over the relevant group of population to estimate the total non-use benefit from a MPA.

Lyssenko and Martinez-Espineira (2009) estimated non-use values of whale conservation using the contingent valuation method. In a telephone survey, over 600 Canadian adults were asked for their preferences to pay for a hypothetical whale conservation program. The aim of the hypothetical program was to subsidize and enforce the use of acoustic devices that reduce the likelihood that whales become entangled in fishing nets. The study estimated a mean willingness to pay of over C$80 per year per household for five years period. This amount reflects the non-use value that average Canadian in the sample obtains on whale conservation. The mean willingness to pay values can be extrapolated across the entire population to estimate the aggregate non-use value of whale conservation.

4.4. Choice Experiments

In a choice experiment, respondents are presented with a sequence of choices between alternative goods or scenarios. The scenarios are described by a number of characteristics or attributes, which have multiple levels that differ among the alternatives. Respondents are asked a series of questions in which a unique 'choice set' is presented each time. Before the choice sets are presented to the respondents, there is a description of the scenario, the research issues, the proposed policy changes, and the implications for the environmental attributes that are being modeled. The choice experiment method allows an analyst to collect more information of respondents' preferences about different attributes of a good, but may also impose significant amount of cognitive burden for the survey respondents.

Choice experiments have an advantage over the contingent valuation method in that they allow the analyst to estimate the values associated with *different* attributes of an environmental good or service. The choice experiment is suitable for estimating values for changes in attributes of a good in question while the contingent valuation method is more appropriate if the attributes of a good in question do not bear much significance for policy making. Choice experiments follow a similar utility maximization framework as the contingent valuation method but allow for different attributes associated with a MPA to be incorporated explicitly in the utility function.

For example, the utility functions with a MPA designed to reduce the threats to endangered species, can be written in the following way:

Without MPA: $$V_0 = \gamma_B B_0 + \gamma_{HS} HS_0 + \gamma_{BW} BW_0 + \lambda Y$$
$$= V^0 + \lambda Y \tag{10}$$

With MPA: $$V_1 = \theta + \gamma_B B_1 + \gamma_{HS} HS_1 + \gamma_{BW} BW_1 + \lambda(Y - WTP)$$
$$= V^1 + \lambda(Y - WTP) \tag{11}$$

In equations (10) and (11), θ refers to a constant, γ_is and λ refer to estimated coefficients, B stands for belugas, HS for harbour seals, BW for blue whales and Y for income. V_0 denotes individual utility from the current state of three threatened and endangered marine mammal species that would be protected better with a MPA. V_1 refers to the utility from an endangered species recovery policy following the establishment of a MPA. The maximum willingness to pay (WTP*) that would be paid is the amount that leaves an individual indifferent between V_0 and V_1 given the utility functions specified by equations (10) and (11). This implies,

$$V^0 + \lambda Y = V^1 + \lambda(Y - WTP^*)$$

$$\text{WTP}^* = \frac{-1}{\lambda}(V^0 - V^1) \tag{12}$$

Olar et al. (2007) conducted a choice experiment study to estimate the non-use benefit of marine mammal recovery in the St.Lawrence Estuary, Canada. About 2,000 respondents were interviewed using an internet panel. Using a set of pre-specified programs and the choices of respondents, program-specific mean willingness to pay was estimated. The average willingness to pay for marine mammal recovery programs in the St. Lawrence Estuary ranged from C$82 to C$242 per household per year. The aggregate benefit of marine mammal protection was calculated by multiplying the average willingness to pay per household by the total number of households in Canada in 2001. The aggregate willingness to pay ranged from C$948 to C$2,798 million per year depending on the magnitude of the expected recovery of the mammal population.

4.2. Qualitative Valuation Techniques

Qualitative valuation instruments have been successfully applied to choose MPA management alternatives around the world, particularly in developing countries (see for example Ferse et al. 2010). Non-monetary or qualitative valuation techniques range from structured individual interviews to more participatory approaches (e.g. focus group discussions, participatory rural appraisal and participatory action research). These methods provide useful information on the non-monetary and unquantifiable benefits and costs of environmental policy interventions to the relevant community in ways that the monetary valuation techniques fail to capture. These non-economic instruments are particularly useful in developing countries where people's ability to pay for environmental services is constrained by their limited financial income. A monetary valuation technique would underestimate of the true value of non-consumptive use and non-use values.

Furthermore, these qualitative techniques help understand the way local communities live their lives, their diverse relationships with one another and with marine resources, how they perceive management interventions in relation to their de-facto property rights, and their roles in the attendant processes. In many ways, the creation of MPAs affects people's rights of access to common-property resources resulting impoverishment, disempowerment and marginalization, to varying extents (Mwaipopo 2008). The qualitative techniques provide the opportunity to negotiate on maintaining a balance between community livelihoods and the conservation goals. Bunce et al. (1999) employed a rapid rural assessment technique to evaluate the Montego Bay Marine Park in Jamaica. Their evaluation provided a base of information on the user group characteristics, their usage patterns, and perceptions of reef management. Their results suggested several significant management implications regarding the need to increase awareness regarding the benefits of the Park and Park management activities, increase user involvement in Park management, and increase inter-sectoral coordination.

5. Decision Criteria for Establishing MPAs

The first step of establishing a MPA is to identify a marine area with ecological or biological significance that needs to be protected. The Conference of Parties (COP) to the Convention on Biological Diversity, at its 9th meeting, adopted a set of seven scientific criteria for identifying such areas (UNEP 2009). According to the COP, an area that fulfills one of the following seven criteria is considered ecologically or biologically significant:

1. an area that contains unique or rare species or habitats;
2. an area that supports critical life-history stages (e.g. breeding grounds, spawning areas, nursery areas, juvenile habitat) of individual species;
3. an area that is important for threatened, endangered or declining species and/or habitats;
4. an area containing a relatively high proportion of sensitive habitats, biotopes or species that are highly susceptible to degradation or depletion by human activity or by natural events or with slow recovery;

5. an area containing species, populations or communities with comparatively higher natural biological productivity;
6. an area containing comparatively higher diversity of ecosystems, habitats, communities, or species, or has higher genetic diversity; or
7. an area with a comparatively higher degree of naturalness as a result of the lack of or low level of human-induced disturbance or degradation.

Once an area is identified as ecologically or biologically significant, the next step of setting up a MPA is to estimate its desirability based on the associated benefits and costs. We outline three approaches that can be used to compare the benefits with the costs of MPAs.

5.1 Benefit-cost Analysis

A commonly used tool to economic decision making is benefit-cost analysis. This approach evaluates the incremental monetary costs and benefits associated with a given policy relative to the status quo. All relevant costs and benefits associated with each alternative policy options would be identified using the methods outlined previously.

Typically, a MPA will generate a stream of future costs and benefits over time. Thus a discount rate[1] is applied to calculate the present value of these benefits and costs. At the final stage of a benefit-cost analysis, a net present value (present value of net benefits) is calculated for each project under consideration by subtracting the present value of the total economic cost from the

[1] A discount rate often reflects a minimum or desired rate of return expected from an investment decision. Note that an opposite approach can be applied by calculating the internal rate of return which indicates the break-even discount rate – a rate at which an investment results in a zero net present value.

present value of the total economic benefit. The standard decision rule is that if the net present value is greater than zero, then establishing a MPA is worthwhile. When multiple projects (i.e. MPAs under different management regimes, MPAs at different sites or different sizes of MPAs) are being evaluated the option that produces the highest net present value in monetary terms is frequently viewed as the preferred policy. A benefit-cost analysis case study is presented in Box 2.

INSERT BOX 2 HERE

The distributional issue of the net present values is likely to be of critical importance for setting up a MPA. Table 1 illustrates the distribution of MPA benefits of the Hon Mun MPA in Vietnam across the two biggest stakeholders, namely tourism and fishing industry. According to the information presented in Table 1 there is a possible ‘win-win’ for both stakeholders as the benefit to both groups increases under the ‘with management scenario’, although the tourism industry gains more than the fishing industry in relative terms. In some circumstances, a MPA might involve negative benefits for one stakeholder and large economic gain for another. In such situations where a strict ban on commercial fishing activities in a MPA may generate net economic losses to fishers, decision makers may wish to provide compensation to the losers if the overall net benefits to the society are positive. Possible compensation schemes could involve job training for alternate livelihoods and temporary or permanent income or food assistance programs (Thur 2009). However, two caveats are embedded in such direct comparison based prophecy. First, it is important to keep in mind that the economic benefits to be obtained from tourism developments may sometimes be overestimated partly to promote vested interests that expect to benefit from the establishments of MPAs. This is of particular import in some developing countries if the benefits that accrue from say, tourism, accrue to only a few locals and fail to induce wide spread economic development or benefits (Khan 1997; Mbaiwa, 2005). Second, the possible costs of conservation induced social displacement and its consequences on local communities such as landlessness, joblessness, increased food insecurity and marginalization cannot be fully quantified in monetary terms. In such cases, even monetary compensation of people displaced from accessing locations that are designated as MPAs may be inadequate to mitigate the losses incurred by the local communities.

INSERT TABLE 1 HERE

5.2 Comparative Risk Analysis

Comparative risk analysis a non-monetary approach that compares the risks involved with each alternative policy following a risk analysis (hazard identification, dose-response assessment, exposure assessment and risk characterization). Central to comparative risk analysis is the construction of a two-dimensional decision matrix that contains project alternatives' scores on various criteria. Based on the magnitude of assessed risk levels, competing policy alternatives are ranked. A common decision rule is to select the policy that involves the lowest amount of risk. If only one policy is under consideration, then the level of assessed risk associated with the policy is compared against the threshold of acceptable risk.

Driscoll et al. (2002) provide an example that illustrates this decision making tool. Their study focused on contaminated sediment management issues in the New York/New Jersey Harbor.

Eight sediment management alternatives were identified for consideration and assessed according to their performance on the criteria of human health risk and ecological risk. Human health risk was evaluated based on three criteria: (1) the number of complete human exposure pathways; (2) the maximum cancer risk calculated from all the pathways; and (3) fish chemical-of-concern concentration. Ecological risk was estimated based on two criteria: (1) the number of complete exposure pathways and (2) the maximum calculated hazard quotient from all the pathways.

The assessment of these risks to human health and ecology involves hazard identification, exposure assessment and dose response analysis. The nature and extent of contamination and the selection of contamination of concern are determined at the hazard identification stage. The exposed populations, areas and potential exposure pathways are identified at the exposure assessment stage. A dose-response analysis then identifies the incidence of an adverse health or ecological effect in exposed populations or geographical areas. The comparative risk assessment results of Driscoll et al. are presented in Table 2. The scores presented under each risk category for each management alternative provides a platform to compare the magnitude of risk across alternatives. For example, the relative risk to ecology and human health for exposure to the undiluted sediment of the no-action alternative exceeds the relative risk of all other alternatives. Among management alternatives that involve mitigation actions, the island confined disposable facility (i.e. Island CDF) poses the highest amount of risks in almost all risk categories except in impacted area.

INSERT TABLE 2 HERE

5.3 The Risk-Benefit Analysis

Risk-Benefit Analysis is a compromise between comparative risk analysis and benefit-cost analysis. In a risk-benefit analysis, risks are valued in monetary terms and are treated as costs (Pearce et al. 2006). The common decision criterion whether a given policy under consideration is desirable is given below,

$$NPV_R = PV(TB) - PV\ (TC) - PV\ (Risks) > 0 \tag{14}$$

where NPV_R denotes 'Net Present Value Adjusted for Risks', PV stands for present value, TB refers to total benefit and TC is total cost.

It is also possible to incorporate risk directly into benefit-cost analysis if analysts are able to assign probability distributions to uncertain costs and benefits with either objective and/or subjective information. Using risk modeling with probability distributions it is possible using Monte Carlo simulations to map out a cumulative probability distribution for net present values associated with a particular project or policy decision (Campbell and Brown 2003).

The risk and simulation approach has been applied to benefit transfer where there is a great deal of uncertainty associated with the transfer of benefits from study site to a policy site (Akter and Grafton 2010). As far as we are aware, this approach has not been used in the context of monetizing the risks of alternative marine reserve sizes or designs. The risk-and-simulation approach allows decision makers to make their own judgments about the nature of these risks, their distribution and possible monetary effects and then simulate the effects. This risk-based approach to benefit cost analysis allows the decision maker to see the range of possible values and the probability that the NPV will have a particular value and consider the trade offs under various scenarios.

6. Conclusions

Marine protected areas are being increasingly used to conserve marine resources in the world's seas and oceans. The inability of some MPAs to achieve their stated objectives has raised concerns among practitioners and policy makers about what should be accounted for in their establishment. To assist decision makers we provide a policy-enabling framework that reviews the various socio-economic approaches that may be used to provide an ex-ante evaluation of MPAs.

We identify three broad assessment criteria that can be followed to successfully undertake an ex ante evaluation exercise. The first MPA assessment criterion is the biological and ecological significance of the site. The recommendations of the Conference of Parties (COP) to the Convention on Biological Diversity outlined in the previous section can be followed to identify areas that are biologically and ecologically significant and hence, needs to be protected. The second MPA evaluation criterion is its economic feasibility. A range of different marketed and non-marketed benefits and costs is associated with the establishment of a MPA. These benefits and costs need to be measured and compared to understand if the proposed MPA makes economic sense.

Our policy-enabling framework describes the various monetary and non-monetary valuation approaches to estimate the benefits and costs of a MPA. We also review three decision protocols that can be applied to compare the benefits against costs of MPAs. Regardless of the approaches used to evaluate MPAs or the decision protocol chosen, we stress that other factors may also need to be considered in the establishment of MPAs. In particular, an assessment of the qualitative and distributional impacts on communities, positive and negative, is required as well as full stakeholder participation in the decision-making processes.

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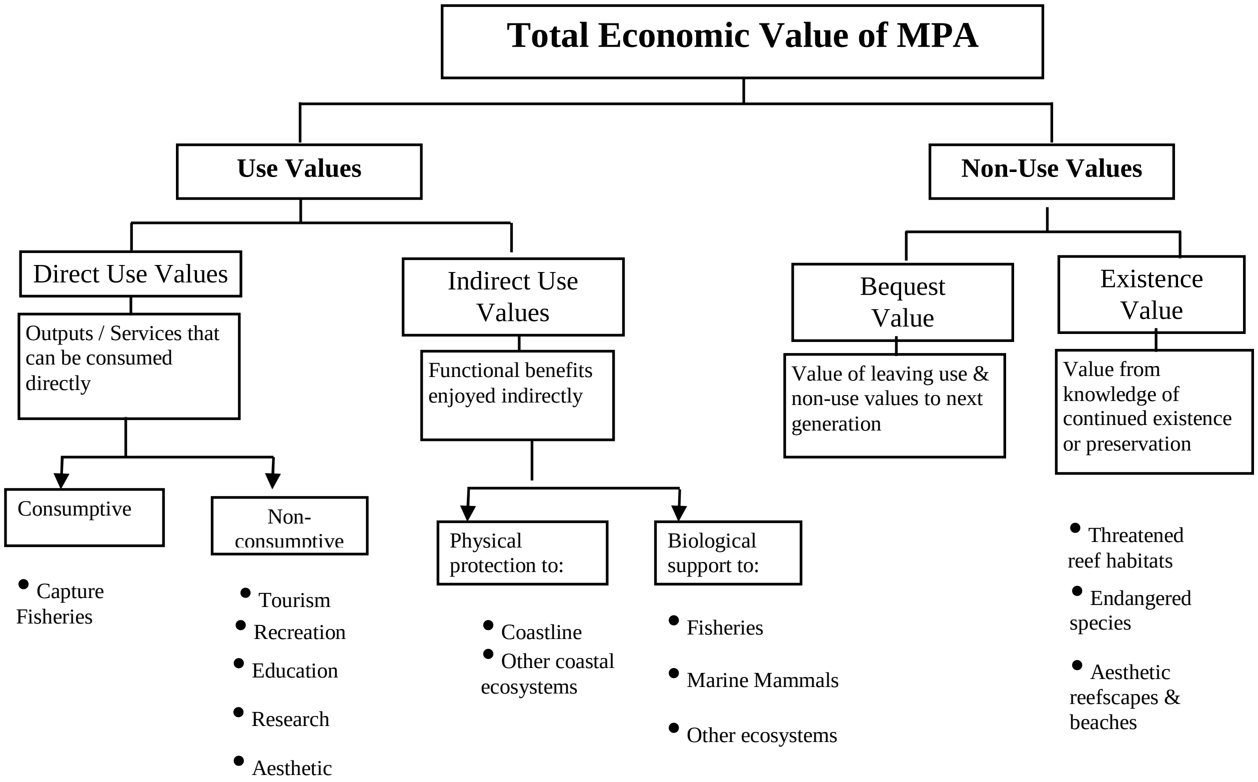
Figure 1 Total Economic Value of MPAs



Source: Emerton 2005

Table 1 Net present benefits and costs of management options for Hon Mun MPA. Present Values (in US$ million, 2004 prices)

	Tourism benefit	Fishery benefit	Conservation benefit	Total Benefits	Costs	NPV
'With management' Options	44.30	25.50	2.88	72.68	2.37	70.31
'Without management' options	30.47	23.64	0.00	54.12	0.22	53.89

Source: Nam et al. (2005)

Table 2 An example of a decision matrix for comparative risk assessment

	CAD[a]	Island CDF[b]	Near shore CDF	Upland CFD	Landfill	No Action	Cement lock	Manufactured soil
Criteria 1 (Impacted area over capacity of facility)	4,400	980	6,500	6,500	Not Applicable	Not Estimated	0	750
Criteria 2 (number of complete exposure ecological pathways)	23	38	38	38	0	41	14	18
Criteria 3 (magnitude of ecological hazard quotient-maximum exposure)	680	2100	900	900	Not Applicable	5,200	0.00002	8.7
Criteria 4 (number of complete exposure human health pathways)	18	24	24	24	21	12	25	22
Criteria 5 (Magnitude of maximum cancer probability-non-Barge worker)	2.80E-05	9.20E-05	3.80E-05	3.80E-05	3.20E-04	2.20E-04	2.00E-05	1.00E-03
Criteria 6 (Ratio of estimated concentration of COC[c]s in fish to risk based concentrations)	28	92	38	38	Not Applicable	220	Not Applicable	Not Applicable

Source: Driscoll et al. (2002).

<u>Explanatory Note:</u>
[a]Confined aquatic disposal facility.
[b]Confined Disposal Facility.
[c]Contaminants of concern.

Box 1 Revealed and stated preference approach of non-market valuation

Non-market valuation techniques can be divided into two different methods: revealed and stated preference approaches. Revealed preference techniques make inferences about non-market values of marine resources (such as whales) based on observations of actual choices or travel behaviors of the visitors (or tourists, travelers), but only in terms of non-consumptive use values. Travel cost method and hedonic pricing method belong to the revealed preference class of non-market valuation techniques.

The stated preference techniques estimate monetary values of non-market environmental services by analyzing individuals' stated behavior in hypothetical settings. The contingent valuation method and choice experiment belong to the stated preference class of non-market valuation techniques. These methods employ public surveys to ask the affected (or relevant) group of population about their willingness to pay to protect the threatened and endangered marine species by constructing a hypothetical market or referendum (Arin et al. 2002; Parsons and Thur 2008).

A positive feature of the stated reference approach is its flexibility. The features of the good (the reef to be protected, abundance of fish species, etc.) in question can be varied by designing survey questionnaire, and thus, estimating utility change for a number of alternative policy options. Its major weakness is its hypothetical nature such that, typically, revealed preference techniques are preferred over stated preference techniques when they can be used to avoide any potential hypothetical bias.

Box 2 Cost-benefit analysis of a MPA: A case study of the Hon Mun MPA in Vietnam

Nam et al. (2005) conducted a benefit-cost analysis of the Hon Mun MPA in Vietnam. The study was carried out in response to the growing need to establish and sustainably manage MPAs in Vietnam. The authors first estimated the total economic value of coral reefs through analysis of reef fisheries and reef-related tourism, as well as other services provided by reef ecosystems. The travel cost and contingent valuation methods were employed to estimate the non-marketed benefits of coral reefs. The present value of the hypothetical cost of operating Hon Mun as a MPA with the purpose of improving local communities' livelihoods, sustainable tourism development, and the conservation of marine biodiversity was subtracted from the discounted total economic value to obtain the net present value (NPV). The NPVs were calculated under two scenarios: (1) without management and (2) with management scenario. The 'without management scenario' refers to the absence of reef management mechanisms to protect the marine ecosystem that might allow unsustainable tourism activities, coral mining and destructive fisheries operation within the MPA area. This involves a zero management cost. The 'with management scenario' refers to a management option that aims at developing eco-tourism at its maximum allowable potential to protect the coral reefs, mangrove and lagoon ecosystem from depletion threats. However, the higher net present value is estimated from the 'with management' option. This implies that this option (i.e. 'with management') is more attractive from an economic perspective. Based on the findings, the authors conclude that the Hon Mun MPA is economically desirable. They furthermore recommend a user fee system to achieve financial sustainability of the MPA management regime.