ECONOMIC ANALYSIS OF MARINE PROTECTED AREAS. A LITERATURE REVIEW

EMPAFISH PROJECT BOOKLET Nº 3

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European Marine Protected Areas as tools for Fisheries management and conservation

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A Literature Review

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Contents

Forewords

At the seas, the need for protecting more or less extended coastal marine areas arose, not to preserve the littoral from urban growth and human space occupation, as in most of terrestrial zones, but as a fishery management tool for recovering overexploited populations and to make fishing sustainable. However, the fact that such protection leads to a quick recovering of underwater seascapes and marine animal and vegetal biodiversity, immediately means an attractive for diving and other recreational activities promoting the urban growth and built-up of tourist facilities in areas with previously low development.

In this way, marine protected areas (MPAs) became in much more than a management tool for an extractive activity, to develop in a tourist centre of attraction and an important economic engine. In most of the cases, the capability to generate riches from tourism and associated activities, as urban development, became much higher than the profits from fisheries, emerging shares conflict, competition among uses and between public administrations. At the same time, MPAs provided excellent reference sites for research and as controls for impact assessment studies, further than for educational activities.

Because of this, to take into account the modern conception of MPAs, according to which local communities have to be effectively involved in their management, become more imperative as public participation and awareness are essential if proper environmental management is to be implemented and as the only way to prevent and solve conflicts of interests.

The whole process, from the first ecosystem approaches in fisheries management and the need to preserve habitats and areas to make fishing sustainable, to the mass tourism development around MPAs, mainly on the basis of diving, and then, the conflicts of competences between tourism, fisheries and environment administrations, have take place in less than two decades. During this time, hardly we have begun to see and to understand the effects of protection and the mechanisms and ecological processes involved. However, the effectivity of marine reserves is obvious enough to become strongly advocated as an ideal tool for the management of coastal fisheries. Since 1997 to 1999 the number of fishing marine protected areas in the EU had doubled. As a consequence of this quick development, the heterogeneity in design, objectives, characteristics, management tools, monitoring plans and involved administrations is as large as the proper number of MPAs. In the last years, the European Commission has underlined the necessity to manage this situation and had promoted policy-oriented research to establish the potential of marine protected areas for marine environmental protection, by investigating the potential of different regimes of protected areas as measures to protect sensitive and endangered species, habitats and ecosystems from the effects of fishing.

In this context the EMPAFISH project (European Marine Protected Areas as tools for FISHeries management and conservation), supported by the European Commission, has as general objectives 1) to investigate the potential of different regimes of MPAs in Europe as measures to protect sensitive and endangered species, habitats and ecosystems from the effects of fishing; 2) to develop quantitative methods to assess the effects of marine protected areas and 3) to provide EU with a set of integrated measures and policy proposals for the implementation of MPAs as fisheries and ecosystem management tools.

Work package 3 of EMPAFISH is devoted to analyze the socio-economics impacts of MPAs and to select the best subset of indicators of MPA performance, under each management regime, considering socio-economic indicators and providing a quantitative analysis of the socio-economic performance of MPAs.

The present booklet reviews the socioeconomic literature dedicated to various aspects of marine protected areas (MPAs) related to ecosystem preservation, fisheries management, recreational activities, and distributional consequences of MPAs and methodological issues for cost-benefit analysis and its application to MPAs, and the specific problem of economic valuation of non-market values. MPAs became an experience field for economists and sociologists and we are just now in the starting point.

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Economic Analysis of Marine Protected Areas
A Literature Review

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This literature review is composed of two parts. The first part describes the socioeconomic literature dedicated to various aspects of marine protected areas (MPAs): ecosystem preservation, fisheries management, recreational activities, and distributional consequences of MPAs. The second part is devoted to methodological issues. It first deals with cost-benefit analysis and its application to MPAs, and then focuses on the specific problem of economic valuation of non-market values.

1. Experiences of economic analysis applied to MPAs

1.1. General overview

The literature on MPAs has dramatically increased during the 90s (Hoagland *et al.* 1995; Conover *et al.* 2000). However, social sciences, and, more specifically economics, share only a small part in this literature (e.g. Tisdell and Broadus 1989; Farrow 1996; Sumaila 1998a; Badalamenti *et al.* 2000; Milon 2000; Sumaila *et al.* 2000; Alder and Sumaila 2002). The neglect of the economic aspects of MPAs is remarkable given that the creation of an MPA may be regarded as an investment in natural capital, and typically pertains to the field of cost-benefit analysis (Hoagland *et al.* 1995; Emerton 1999; Sanchirico 2000a; Carter 2003)[1].

Economists consider ecosystems as capital goods generating valuable services, some of which are marketable while others are not (e.g. Point 1992). This approach is used, for instance, by studies attempting to justify the interest of biodiversity protection on economic grounds (e.g. Brown and Goldstein 1984; Boyle and Bishop 1987; Weitzman 1993, 1998; Alexander 2000). As a result, various studies focusing on the valuation of ecosystem services have been conducted (Costanza 2000; Heal 2000; Costanza and Farber 2002; De Groot *et al.* 2002; Farber *et al.* 2002; Howarth and Farber 2002). However, natural assets are often providers of multiple services, which creates difficulties when trying to assess their economic value (Desaigues and Point 1990a). A number of assessment methods such as travel cost, hedonic pricing, and contingent valuation have been developed to estimate the economic value of these services (e.g. Randall and Stoll 1983; Desaigues and Point 1990b; De Groot 1992; Pérez *et al.* 1996; Bateman and Langford 1997; Claeys-Mekdade *et al.* 1999; Dabat and Rudloff 1999; Desaigues and Ami 2001)[2]. Yet, only a small part of this literature is dedicated to the economic valuation of natural assets deals within marine ecosystems (e.g. Bennett and Reynolds 1993; Loomis and Larson 1994; Cesar 1996; Costanza *et al.* 1997; Le Goffe 1999).

An important topic for economists dealing with the conservation of ecosystems is the analysis of management tools, such as MPAs. Analyzing the economic impact of an MPA on the various services provided by an ecosystem is often performed with the help of bioeconomic modelling. Bioeconomic modelling provides a simplified and general view of the impacts of an MPA on the dynamics of natural resources and of their uses. It makes it possible to simulate various management scenarios, and to analyze interactions between the different services provided by the ecosystem, thus providing a basis for the estimation of the net benefit to society generated by the MPA. Bioeconomic modelling of MPAs, however, has a short history. Literature shows that it started a decade ago, but has been rapidly growing (Sumaila and Charles 2002; Grafton *et al.* 2005b).

Currently, few estimates of the global economic surplus generated by MPAs are available. Most of these estimates imply that the whole array of economic values (use and non-use) provided by the ecosystem are taken into account (Pearce and Turner 1990; Munasinghe and McNeely 1994; Hoagland *et al.* 1995; IUCN 1998). However, few studies take into account non-use values, which may result in under-recording the economic value of MPAs (Hoagland *et al.* 1995). As underlined by Pendleton (1995), some economic appraisals of tropical marine parks value the activities practiced in the MPA rather than protection provided (e.g. Post 1994; Dixon *et al.* 1995).

Considering use-values, until the 2000s, the bulk of the economic literature dedicated to MPAs was focused on fisheries management (Sanchirico 2000b). During the same period, the recreational benefits of MPAs were addressed only by a small number of studies, mainly related to the tourist potential of MPAs in tropical (especially coral reef) areas (e.g. Bell 1992; Dixon *et al.* 1993, 1995; Davis *et al.* 1995; Pendleton 1995; Leeworthy and Wiley 1997). Recently, this topic has attracted an increasing number of studies (e.g. Buerger *et al.* 2000; White *et al.* 2000; Dixon *et al.* 2001; Arin and Kramer 2002; Leeworthy and Wiley 2002; Rudd and Tupper 2002; Southwick 2002; Green and Donnelly 2003; Moscardo *et al.* 2003; Moscardo and Ormsby 2004; Ormsby *et al.* 2004; Tongson and Dygico 2004; Valentine *et al.* 2004).

Economic analysis of MPAs faces another important limitation, since most models that have been developed so far are essentially theoretical. This may be explained by the gap between empirical data that are available, and data that are required for estimating the parameters of the models. As a result, most empirical applications are confined to a limited number of studies, such as temporary fishery closures (e.g. Cheng and Townsend 1993; Watson *et al.* 1993; Somers and Wang 1997), or trawling bans in some areas (e.g. Somerton

[1] For more details on cost-benefit analysis and its application to the case of MPAs, see section 2.1. of this review.
[2] For more details on these methods, see section 2.2. of this review.

and June 1984; Armstrong *et al.* 1993; Boncoeur *et al.* 2000; Whitmarsh *et al.* 2002; Dalton and Ralston 2004). The application of such impact studies may be regarded as partial forms of cost-benefit analysis. For example, Whitmarsh *et al*. (2002) assess the economic effects of a trawl-ban on the artisanal fishing boats in N.W. Sicily; Leeworthy and Wiley (2002) calculate the losses incurred by displaced consumptive users (commercial and recreational) following the creation of the Tortugas Marine Reserve in the Florida Keys National Marine Sanctuary; and Morin (2004) assesses the economic impact of three potential scenarios of MPA sites on fishing and recreational boating. Multicriteria analysis methods are more rarely used, as in the case of Buccoo Reef Marine Park, Tobago, West Indies (Brown *et al.* 2001). There are also a few surveys of stakeholder attitudes, analysing how various groups of stakeholders have been, or might be affected by the zoning plan established within MPAs (Aussedat 1995; Suman *et al.* 1999; Dobrzynski and Nicholson 2000; Sabourin and Pennanguer 2003).

Three types of purposes may be assigned to MPAs: (i) biodiversity protection, (ii) sustainable fisheries management, and (iii) the development of non-extractive uses of the ecosystem (ecotourism and other recreational activities). Contrasting with the first two objectives, the literature on MPAs often regards the development of non-extractive uses of the ecosystem as an objective of minor importance. However, in practice, it is often the major reason that is put forward when creating an MPA, which illustrates the gap between theoretical and real-world considerations concerning MPAs (Agardy 2000; Jamieson and Levings 2001).

In many cases, a combination of objectives is assigned to a definite MPA. Some of them are complementary, while in other cases the objectives may compete with each other, inducing conflicts of interest between various stakeholders. This calls for an analysis that takes into account the complexity of the consequences of the creation of the MPA (e.g. Point 1990, 1998; Smyth 1995; Polunin *et al.* 2000; Salmona and Devardi 2001; Boncoeur *et al.* 2002; Sanchirico *et al.* 2002; Alban 2003). Some authors consider that the task of MPAs is to manage the various uses of marine ecosystems, in order to minimize their impact on environment as well as use conflicts, especially in multi-use parks. According to this view, MPAs are a tool for integrated coastal management (e.g. Barley 1993; Ehler and Basta 1993; Kelleher 1996; McArdle 1997; Costanza *et al.* 1998; Done and Reichelt 1998; Day 2002; White *et al.* 2002; Lynch *et al.* 2004).

In the following three sections, each of the above-mentioned objectives is reviewed, with a focus on the way economic literature has dealt with its value. The fourth section is dedicated to the analysis of distributional problems generated by the creation of an MPA, and to the compensatory measures that may help overcome these problems.

1.2. Ecosystem protection

There is a general agreement that a major objective of MPAs is to help protect species, habitat and biodiversity. Further, MPAs are expected to contribute to the improvement of scientific knowledge on marine ecosystems (e.g. Cognetti 1986; Bohnsack 1998; Fraschetti *et al.* 2002). However, the shortness of this part of the review reflects the limited amount of economic literature dealing with the services provided by MPAs as a tool for ecosystem protection.

From an economic point of view, benefits related to biodiversity protection are mainly related to non-use values, and more specifically to option values (Weisbrod 1964). Nowadays, the option value of biodiversity is widely acknowledged (e.g. Weitzman 1992; Polasky *et al.* 1993), and a distinction is made between option and quasi-option values (e.g. Bonnieux and Desaigues 1998). The option value *sensu stricto* is a kind of insurance premium warranting the possibility of a future use. The quasi-option value is the value of information gained by delaying a decision concerning the use of a resource that results in irreversible effects. It is acknowledged that one of the most important services provided by biodiversity is provision of information (Faucheux and Noel 1995). In the context of uncertainty and poor knowledge characterizing marine resources management (Costanza *et al.* 1998), the concept of quasi-option value is highly relevant, because it emphasizes the value of increasing information (Hanna 1983). However, very few assessments of non-use values generated by MPAs may be found in the literature (Dharmaratne *et al.* 2000; Bhat 2003).

The preservation of ecosystems is not the only reason put forward when creating an MPA. Other reasons include the preservation of historic heritage (shipwrecks, archaeological sites…) (Cuthill 1998; Breen and Forsythe 2001) and the preservation of cultural heritage by protecting traditional activities and lifestyle of local maritime communities (White 1986; Carew-Reid 1990; Lam 1998).

1.3. Resource conservation for sustainable fishing

Apart from ecosystem protection, a substantial amount of literature has been dedicated to the potential interest of MPAs for fisheries management (e.g. Bohnsack 1993; Shackell and Wilson 1995; Hall 2002; Polunin 2002; Pickering 2003; Hilborn *et al.* 2004). MPAs are supposed to help fisheries management in two ways. First, they provide direct benefits by contributing to the restoration of overfished stocks (e.g. Bohnsack 1996a; McClanahan and Mangi 2000; Roberts *et al.* 2001), and by decreasing the risk of stock collapse (Fogarty *et al.* 2000). Second, they provide an alternative to conventional fisheries management tools, especially when these tools cannot be implemented effectively (e.g. Agardy 1994a, 1994b; Bohnsack 1996, 1999; Botsford *et al.* 1997; Lauck *et al.* 1998). These different types of benefits are closely interrelated (Shackell and Wilson 1995; Parrish 1999; Syms and Carr 2001), and are also related to the benefits concerning ecosystem protection. Creating an MPA is often considered as an application of the precautionary principle, against the various sources of uncertainty[3] in the management of marine resources. For example, uncertainties arise from the natural variability of ecosystems, the impacts of various anthropogenic activities on these ecosystems (e.g. Lauck *et al.* 1998; Mangel 2000), and the socioeconomic system (Sumaila 2002).

This review of the economic literature dealing with MPAs as fisheries management tools will first focus on methodological aspects. It will then describe results of the research in this field.

1.3.1. Methodology

Bioeconomic modelling is the major analytic tool used by economic studies concerning the role of MPAs in fisheries management (e.g. Holland and Brazee, 1996; Hannesson 1998; Conrad 1999; Sanchirico and Wilen 1999, 2002; Holland 2000; Pezzey and Roberts 2000; Anderson 2002a; Holland and Sanchirico 2004; Sanchirico 2004; Schnier 2005). Several literature reviews have dealt with this topic (Guénette *et al.* 1998; Pickering 2003; Grafton *et al.* 2005b; Pelletier and Mahévas 2005).

Bioeconomic models of MPAs may be classified into two major categories.

Models that are directly derived, also known as spatially non-explicit bioeconomic models. Most of these are equilibrium models dealing with one species and one gear, and rely on the assumptions of space homogeneity. Their basic purpose is to assess the way fishing is impacted by closing an area representing a given proportion of the fishing zone. The spillover effect of the reserve is modelled with the help of a transfer function, usually based on the difference between stock density inside and outside the reserve, and on the mobility of fish. Models of this type may vary in terms of complexity. However, they usually suggest that the biological and socioeconomic consequences of creating an MPA depend on a range of factors, such as reserve size, fish mobility, and degree of control over fishing effort. For example, Holland and Brazee (1996), making use of a structural model, show that creating an MPA may help increase the level of sustainable catch in an overfished fishery, provided the spontaneous tendency to increase fishing effort can be controlled. Hannesson (1998) and Anderson (2002), use a global logistic model to address the question of the usefulness of an MPA in a deterministic context, assuming freedom of access into the fishing zone. Sumaila (1998) and Conrad (1999) depict the benefits of MPAs for fisheries in a stochastic environment.

The standard assumptions of a single species fishery and space-homogeneity are in many cases oversimplified, and may lead to inappropriate conclusions on the impact of an MPA on fisheries. The multispecies bioeconomic model developed by Holland (2000) shows that the impact of an MPA varies according to species. Introducing space-heterogeneity in his bioeconomic model under the form of “biological hot spots”, Schnier (2005) shows that the optimal size of the MPA depends not only on the productive capacity of the reserve and the surrounding fishing grounds, but also on the degree of heterogeneity between the two regions.

Spatially explicit models form the second category of bioeconomic models of MPAs. The literature on these models is more recent than the one dealing with the first category, but it has been growing fast. A recent review of the literature on spatial modelling applied to fisheries economics can be found in a special issue of the journal *Marine Resource Economics* (Vol.19 (1), 2004) (Curtis and McConnel 2004; Dalton and Ralston 2004; Hicks *et al.* 2004; Holland 2004; Holland *et al.* 2004; Sanchirico 2004; Smith and Wilen 2004; Strand 2004). The seminal paper in this category was written by Sanchirico and Wilen (1999). Further developments may be found in Sanchirico

[3] Five types of uncertainty are distinguished in the management of renewable natural resources : measurement errors, methodological errors, modelling errors, estimation errors, implementation errors (e.g. Gates 1984; Caddy and Mahon 1995; Berkes *et al.* 2001; Syms and Carr 2001).

and Wilen (2001), and Sanchirico (2004, 2005). The basic principle of these models is to represent a discrete number of subpopulations distributed in separate zones (patches), but interconnected by biological and economic relations (metapopulations). Spatially explicit models are usually multispecies. Their main focus is on the location of MPAs rather than on their size. The choice of the location needs to integrate the major oceanographic processes in the region (Carr and Reed 1993), the ecological characteristics of the habitats, the distance of larval dispersion (Quinn *et al.* 1993), and socioeconomic factors.

Developing spatially explicit models requires a realistic description of microeconomic behaviours, not only through time but also through space, in order to explain the mobility of fishing fleets (Hilborn and Kennedy 1992; Chakravorty and Nemoto 2001; Smith and Wilen 2003; Holland *et al.* 2004; Mahévas and Pelletier 2004). Several techniques may be used to this end. The most commonly used are gravity models (Caddy 1975; Seijo *et al.* 1993; Walters *et al.* 1993, 1999; Seijo and Defeo 1994; Walters and Bonfil 1999), and random utility models (Bockstael and Opaluch 1983; Eales and Wilen 1986; Holland and Sutinen 1999; Wilen *et al.* 2002; Hutton *et al.* 2004). Other analytic tools, such as game theory (Beattie and Sumaila 2002; Sumaila 2002) and multi-agent modelling (Thébaud and Soulié 2003) may also be used to simulate fishers' behaviour.

1.3.2. Results

Until recently, the literature on MPAs has provided limited empirical evidence on the socioeconomic benefits of MPAs to fishers (Crowder *et al.* 2000; Fogarty *et al.* 2000). According to Crowder *et al.* (2000), data were available for only 28 MPAs out of a total of 1300.

An increase in catches ?

A number of authors are sceptical about a possible increase in catches due to the implementation of an MPA (Schmidt 1997; Hatcher 1998; Shipp 2002; Willis *et al.* 2003).

In theory, the MPA should increase fish catches in the fishing zone neighbouring the no-take zone, thus overbalancing the negative impact of decreasing the size of the fishing zone (e.g. Polacheck 1990; Demartini 1993). Moreover, over time, the MPA should bring about stability in catches, by making fish stocks less vulnerable to overfishing (e.g. García-Charton and Pérez-Ruzafa 1999). This "buffer effect" may be considered as a net benefit for fishers provided they are risk-adverse (Conrad 1999; Carter 2003). However, this might not be the case, according to studies investigating the way fishers choose their fishing place (Holland and Sutinen 1999; Mistiaen and Strand 2000; Dalton and Ralston 2004; Smith and Wilen 2004). According to these studies, the benefits derived from the buffer-effect are not regarded as high enough by fishers to balance the loss of some of their former fishing zones. There is a need for more systematic surveys on the decision criteria of fishers, regarding the choice of their fishing zone (Hoagland *et al.* 2002; Holland 2002, 2004).

A case-study of a marine reserve in the Philippines, used by many authors, suggests the existence of a positive effect on catches in adjacent fishing zones (Russ and Alcala 1996). In fact, the proof in this case was *a contrario*, since the survey revealed a decrease in landings following the reopening of the reserve to fishing. This study provided some confirmation of the assumption of a transfer effect of adult fish from the reserve to the fishing zone. Another study showed an increase in catches per unit of effort (CPUE) in the St-Lucia islands, by comparing the CPUE of artisanal fishermen before the creation of the reserve and 5 years later (Roberts *et al.* 2001). Other studies show encouraging results in New-Zealand (Kelly *et al.* 2002), and in Africa (McClanahan and Mangi 2000; Kamukuru *et al.* 2004; Kaunda-Arara and Rose 2004). Recreational fishing is also used as an example to demonstrate the benefits of MPAs for fishing. According to Johnson *et al.* (1999), the change in the frequency of official records concerning sport fishing indicates a positive impact of the reserve that was created around the Cape Cañaveral launching base (Florida). The literature also provides evidence based on fishers declarations (Rowley 1994; Dobrzynski and Nicholson 2000), or on indirect elements such as the arrival of new fishers, or the buying of new boats (Boudouresque 1990; Binche 1992; Ramos 1992).

An impact on prices?

The implementation of a marine reserve may have different consequences on the price for fish: (i) an impact due to the variation of quantities landed (see above); (ii) a "quality" impact, due to a shift in the size and species composition of landings (Pauly *et al.* 1998), and to better marketing opportunities (Sanchirico *et al.* 2002). Fishers may also take advantage of the "ecologically correct" image of the fishing zone of the MPA to sell their fish at a higher price (Charles 2001). However, these price effects of MPAs have been seldom studied, as underlined by Sanchirico *et al.* (2002) and Carter (2003). Models usually assume constant ex-vessel prices (e.g. Pezzey *et al.* 2000).

The impacts on fishing effort and costs

The creation of an MPA is likely to result in a spatial reallocation of fishing effort (Fisheries Society of the British Isles 2001; Sanchirico *et al.* 2002). In order to maintain their level of catch, fishers are forced to maximise their effort in those areas that are left open to fishing (Gulf of Mexico Fishery Management Council 1999; Parrish *et al.* 2001). A "fishing the line" effect, i.e. a concentration of fishers close to the boundary of the reserve is frequently noted, and is likely to outweigh some of the benefits expected from the MPA (Dobrzynski and Nicholson 2000; Christie *et al.* 2002; Kelly *et al.* 2002). In the long run, an MPA may induce a shift of fishing effort towards other fisheries, which may create a number of negative effects in these fisheries (Smit 1995; Murawski *et al.* 2000; Sanchirico 2000a; Le Gallic 2001). Transfers of fishing effort may create new fisheries conflicts (Bohnsack 1996b), which result in a decrease of the socioeconomic benefits of MPAs (Holland 2000).

Sanchirico and Wilen (1999) show that the effects of reserves on fishing are highly dependent on how fishers spatially allocate their effort. Presenting an extension of the model developed by Sanchirico and Wilen (2001), Anderson (2002b) analyses vessel behaviours taking into account the distance from port. Fishers have a tendency to allocate their effort to areas that generate higher relative rents. Applying this idea to the California sea urchin fishery with a deterministic model, Smith and Wilen (2003) show that accounting for spatial behaviour of fishers is likely to deceive optimistic expectations concerning the positive effect on discounted rents of creating a reserve in a heavily fished area. After the creation of a no-take zone, fishers are likely to increase their effort in the area which is left open to fishing; if this shift is not controlled, it may undermine the expected benefits of the MPA as regards fishing mortality. The perception of expected rent may be influenced by the knowledge about resource location, measured for instance by past catches (Holland and Sutinen 1999; Holland 2000). It may also be influenced by species prices (Holland and Sutinen 1999) and harvesting costs (e.g. cost of travelling to fishing grounds). Grafton *et al.* (2005b) conclude that this type of effect generates spatial 'economic gradients' that may be quite different from 'biological gradients' generated by larval export and adult transfers.

The spatial redistribution of fishing effort induced by the MPA has both direct and indirect influence on fishing costs. If fishers are compelled to go fishing farther from their homeport, this will increase their operational costs (Binche 1992; Murawski 2000; Smith and Wilen 2004). Moreover, if they have to go fishing in places that are unfamiliar, this will increase the time they need to look for fish (Boudouresque 1990; Ramos 1992). The increase in time devoted to travelling and to search for fish results in less time devoted to catching the fish (e.g. Carter 2003), which creates an incentive for fishers to invest more money in electronic devices for detecting fish stocks (Sanchirico *et al.* 2002). Another possible consequence of the MPA is to increase the risk associated with fishing, by inducing under equipped and under experienced fishers to go fishing further offshore (Holland 2000). The costs resulting from space transfers of effort vary according to the degree of dependency of fishers on a particular fishing zone (Holland 2002; Sumaila 2002; Carter 2003). With few alternatives, small boats are usually more concerned by the closure of an area than large boats (Boncoeur 2004).

Not only is spatial distribution of fishing effort likely to be affected by the implementation of an MPA: if this implementation succeeds in increasing the CPUE in the adjacent fishing zone, and if effort is left uncontrolled in the fishing zone, it is likely to increase the total level of fishing effort, until the additional rent created by the MPA is totally dissipated (Hannesson 1998; Gulf of Mexico Fishery Management Council 1999). Under this scenario, the MPA results in increasing the total amount of fishing costs and exacerbating the tendency to overcapacity.

A possible second rank optimum

MPAs are often presented as an alternative to conventional fisheries management tools (e.g. Plan Development Team 1990; O'Neill 1993; Agardy 1994b; Davis and Dugan 1994; Bohnsack 1996; Hall 1998; Parrish *et al.* 2001). While these tools (e.g. TACs and quotas) reflect mainly a single species management, some authors consider that marine reserves favour an ecosystem approach to fisheries management (Sumaila *et al.* 2000; Palumbi 2002). In the context of inshore fisheries, generally characterized by a great complexity (multi-species, multi-gear fisheries), conventional management tools often perform poorly. Moreover, socio-political pressures often make it difficult to reduce fishing effort (Sumaila 1998b; Roberts 2000). The choice of a specific fisheries management tool is frequently more influenced by political considerations than by scientific arguments (Guénette *et al.* 1998).

Where conventional management tools result in a perfect control of fishing mortality, it may be argued that there is no strong case for creating a marine reserve, from a fisheries management point of view. Hannesson's model (1998) suggests that the first rank optimum (*maximum maximorum*) is reached when the share of the no-take zone in the potential fishing zone is zero. However, in the case where socio-political factors do not make it possible to lower the fishing effort below a given level, creating a marine reserve may correspond to a second rank optimum, because catches and rent, for a given level of effort, may be higher with the reserve than without it (Holland and Brazee 1996; Boncoeur *et al.* 2002). Such a result requires that fishing mortality is not left fully uncontrolled outside the reserve, i.e. that MPAs are considered as a complement rather than as an alternative to other fisheries management tools (e.g. Mangel 1998).

Taking uncertainty into account

According to several authors, the uncertainty characterizing fisheries management is the major factor justifying the use of MPAs as fisheries management tools (Clark 1996; Lauck *et al.* 1998; Sumaila 1998a; Conrad 1999; Mangel 2000; Grafton *et al.* 2004). Reserves act as a hedge against inevitable uncertainty (Ludwig *et al.* 1993; Botsford *et al.* 1997), especially where harvest rates and population stocks are measured with error, and harvests are less than fully controllable (Clark 1996). Using a dynamic source-sink model with two forms of uncertainty, Grafton *et al.* (2004) conclude that the key benefit of reserves is that they increase resilience, i.e. the speed it takes a population to return to a former state following a negative shock. Increased resilience due to a reserve can also increase resource rents, even with optimal harvesting. This conclusion contradicts the idea that reserves have no value if harvesting is optimal (e.g. Hannesson 1998). Simulating the economic value of a marine reserve in the case of the northern cod fishery of Atlantic Canada, Grafton *et al.* (2005a) conclude that, assuming environmental variability, an optimal-sized marine reserve offers the possibility of a 'win-win-win' outcome: it can increase the resource rent, reduce the recovery time of the stock after a negative shock, and lower the risk of a catastrophic collapse.

The problem of management costs

MPAs are often presented as a way to simplify fisheries management, and therefore to minimize cost. Three arguments are usually put forward: (i) spatial measures are easy to understand, which makes them easier to accept (e.g. Plan Development Team 1990; Roberts and Hawkins 2000); (ii) designing MPAs does not require a large amount of information, which makes them easy to create (e.g. Bohnsack 1993); (iii) checking enforcement is relatively easy (e.g. Roberts and Polunin 1993; Causey 1995).

However, these arguments are controversial, especially as regards enforcement costs. While some authors think that costs are lower with MPAs than with conventional management tools (e.g. Armstrong *et al.* 2001; Carter 2003), other authors support the opposite view (Parrish 1999; Sanchirico *et al.* 2002). Balmford *et al.* (2004) have provided an estimate of the running costs of MPAs, based on a survey of 83 MPAs worldwide, with a questionnaire sent to MPA managers. According to the results of this survey, annual running costs per unit area spanned six orders of magnitude, and were higher in MPAs that were smaller, closer to coasts, and in high-cost, developed countries. Moreover, the total annual running cost per unit area of an MPA were higher for MPAs that were fully protected from fishing.

It is widely acknowledged that the effectiveness of enforcement is a weak point in fisheries management (e.g. Jones 1994). Natural reserves (continental or marine) are often regarded as an incentive to poaching (Bromley 1991; Milner-Gulland and Leader-Williams 1992). Poaching is likely to reduce the benefits expected from the creation of a reserve (Jones 1994; Dobrzynski and Nicholson 2000), and several authors consider this problem as the major cause of failure of MPAs (Bohnsack 1993; Attwood *et al.* 1997; Murray *et al.* 1999a). According to Crowder *et al.* (2000), simplifying enforcement requires that the borders of the reserve are easy to control. Other factors such as the distance from the shore, the size and the type of MPA are also considered. According to various authors, the potential benefits of MPAs in terms of enforcement costs depend on the use of new information and communication technologies, such as geographical positioning systems (GPS) (e.g. A.A.A.S. 2001).

Support by fishers may have a favourable influence on enforcement costs (e.g. Causey 1995; Ticco 1995; Bohnsack 1996b; Parrish *et al.* 2001; Hallwood 2004), developing in some cases a phenomenon of self-surveillance by users (Sutinen and Kuperan 1999). For many authors, the success of a MPA is conditioned by the support of users (e.g. Alban *et al.* 2004; Helvey 2004; Scholz *et al.* 2004), which in turn depends on the degree of enforcement of the rules, and on their feelings regarding the equity of these rules and the way they are enforced (Bromley 1991; Crosby 1994).

1.4. Promotion of recreational non-extractive activities

MPAs provide a number of opportunities for recreational users to enjoy a good quality marine environment when practising non-extractive activities such as sailing, diving, kayaking, or marine mammal watching (e.g. Hundloe 1980; Bohnsack 1998; Irving Oxley and Brown 2003). Tourism related to these types of activities is often called “ecotourism” or “nature tourism”[4].

Though less developed than the literature devoted to fisheries management, the part of the economics literature on MPAs dealing with ecotourism is growing fast.

The methodological tools used in this literature are different from those used by studies dedicated to fisheries management. Up to now, bioeconomic modelling has been seldom used for investigating the relations between MPAs and recreational non-extractive activities. Valuation techniques of non-market benefits to visitors, such as travel cost and contingent valuation methods, are increasingly used. The geographical scope of studies dedicated to recreational activities in MPAs is mainly composed of tropical areas. Diving in coral reef zones is the paradigmatic case, where the flow of divers is analysed in relation with the increase in fish abundance and good quality of coral due to the MPA.

This section is organised in four parts: (i) the case for developing ecotourism in MPAs; (ii) evaluation of the impact of MPAs on ecotourism; (iii) negative impacts of tourism, and (iv) access fees.

1.4.1. The case for developing ecotourism in MPAs

Consumers’ surplus (more specifically visitors’ surplus) is increased by the protection of some specific components of the ecosystem (corals, marine mammals,...). In the case of diving for instance, the satisfaction, and hence the willingness to pay of divers increase with the quality of the ecosystem (Davis and Tisdell 1996; Rudd and Tupper 2002), and the greater probability of watching rare species or large-sized individuals is an incentive for divers to frequent MPAs. As a result, developing ecotourism may be regarded as a way to translate the benefits of ecosystem preservation into economic terms, i.e. to increase the value of MPAs and to reconcile ecosystem protection with economic development (Davis 1981; Tisdell 1991; Agardy 1993; Dixon 1993; Lindberg and Huber 1993; Munasinghe and McNeely 1994; Wells 1997; Anaya 1998; Salm and Clark 2000; Weaver 2002). Incomes provided by the development of ecotourism may contribute to the funding for the protection of the environment, covering a part of the management costs of the MPA. Ecotourism may also stimulate local economic development, provided local communities take part in the management and operation of activities related to ecotourism (Salm 1985; Meganck 1991; Kenchington 1993; Vogt 1998; Zabala 1999). As far as these activities rely on the quality of the environment, ecotourism may be regarded as an economic incentive to protect ecosystems and a way to promote their sustainable use (Myers 1972; Goodwin 1996; IUCN 1998, 2000).

Several other arguments are used to justify the development of non-extractive recreational activities inside MPAs:

1. Economic benefits provided by the development of non-extractive uses in MPAs could be used to compensate the costs incurred from restricting fishing and other extractive activities (Kelleher and Kenchington 1984; Dixon and Sherman 1990; Kenchington 1991, 1993; Craik 1994; Vogt 1998; Fogarty *et al.* 2000; Graham and Heyman 2000; Dixon *et al.* 2001).

2. The development of ecotourism might improve the social acceptability of the project, by providing sources of income to the populations living in the vicinity of the MPA (Robinson 1977; Salm 1985; Dixon *et al.* 1993; Post 1994; Aussedat 1995; Eichbaum *et al.* 1996; Attwood *et al.* 1997; Hockey and Branch 1997).

[4] Wells (1997) defines nature tourism as “those forms of tourism where natural attractions of ecological significance are the destination”. There is a large debate on the definition of nature tourism, or ecotourism (Goodwin 1996).

3. Non-extractive activities could contribute to the educational role of the MPA, by providing users with visible proofs of the results of protection measures, and through the development of educational programs (Spoto and Franzosini 1991; Forestell 1993; Brunckhorst 1994; Kaza 1995; Eichbaum *et al.* 1996).

4. The development of ecotourism may become a political incentive for ecosystem protection (e.g. Eagles *et al.* 2002). For instance, it has been argued that reopening whaling in a given country would be an incentive for whale-watchers to go to other countries, because they are ethically opposed to whaling (Orams 2002).

1.4.2. The impact of MPAs on ecotourism

The increasing interest for outdoor recreational activities, within a good quality natural environment, results in the development of ecotourism inside natural protected areas (Grandbois 1999; Eagles *et al.* 2002). As regards MPAs, special attention has been paid to two activities: diving in coral reef areas (e.g. Bell 1992; Richez 1992; Suman and Shivlani 1998; Shivlani and Suman 2000; White *et al.* 2000; Graham *et al.* 2001; Rudd and Tupper 2002; Shaalan 2005), and marine mammals or shark watching (e.g. Anon. 1997; Davis *et al.* 1997; Graham and Heyman 2000; Hoagland *et al.* 2000; Mazaudier and Michaud 2000; Hoyt and Hvenegaard 2002)[5].

Many authors underline the attractiveness of MPAs for tourism (e.g. Dixon and Sherman 1990; Agardy 1993; U.N.E.P. 1994; Dharmaratne *et al.* 2000; Shafer and Inglis 2000). Famous MPAs, such as the Great Barrier Reef in Australia, have become real "tourist attractions" (Kenchington 1991). In Europe, several case studies in the Mediterranean sea describe an increase in tourism within MPAs (Ramos 1992; Ribera-Siguan 1992; Richez 1992; Badalamenti *et al.* 2000). However, there are fears that creating an MPA results in decreasing tourist flows, due to regulatory restrictions imposed on tourists or their activities (e.g. Salmona and Verardi 2001). In developing countries, the economic effects of MPAs due to tourism are often disappointing (Wells and Brandon 1992). Only a small number of MPAs in these countries attract a significant number of tourists, a situation which is often due to the lack of good quality accommodation for visitors.

Where observations show an increase in tourist visits to an MPA, it may be difficult to separate what is due to the natural increase in tourism, and what is due to the "MPA effect". It is often easier to survey the relation between tourism and improved quality of the ecosystem by monitoring specific activities. The analysis of visitors experience has long been limited to terrestrial parks. Most field surveys of visitors have focused mainly on the reasons for choosing a given protected area, the influence of various ecosystem attributes and the attitude of visitors towards overcrowding (Ormsby *et al.* 2004). This type of analysis is now also used in MPAs (Shafer and Benzaken 1998; Mundet and Ribera 2001; Valentine *et al.* 2004). In the case of scuba-diving and snorkelling, methods such as travel cost and contingent valuation have been used to assess the benefits provided to visitors of MPAs by environmental quality (Davis *et al.* 1995; Graham *et al.* 2001; Rudd and Tupper 2002; Bhat 2003; Wielgus *et al.* 2003).

1.4.3. Negative impacts of tourism

Besides its positive contribution, the development of tourism in MPAs may also have some negative impacts, which have to be taken into account in a cost-benefit perspective.

First, an uncontrolled increase in tourists is likely to generate congestion, or overcrowding (Dixon and Sherman 1990; Tisdell 1991; Davis and Tisdell 1995, 1996; Goodwin 1996; Bhat 2003). Such an increase may result in a lower quality of environment, leading to undesirable consequences from a cultural point of view (loss of authenticity). There is evidence to show that mass tourism has negative environmental consequences (e.g. Davis *et al.* 1995; Shaalan 2005). In the case of MPAs, divers may damage habitats, for instance when their fins hit corals or when they trample on reef-flat communities (e.g. Hawkins and Roberts 1993; Harriott *et al.* 1997), or when they grasp hold of kelps in strong current flows (Schaeffer *et al.* 1999). They may also alter the behaviour of fish, especially fish feeding (Milazzo *et al.* 2005). Marine mammal watching may also alter the behaviour of animals, by imposing stress on them (Duffus and Dearden 1993; Mazaudier and Michaud 2000). Mooring on corals or on sea-grass beds by yachts or dive boats generates environmental damage (Francour *et al.* 1999).

These factors, which contradict the objectives of protection assigned to MPAs, may in turn decrease the attractiveness of the MPA to tourists, thereby generating a "tourism cycle" (Butler 1980; Lindberg 1991; Murray *et al.* 1999b).

[5] Swimming with whales is becoming a new "niche" for the development of ecotourism (Davis 1998; Valentine *et al.* 2004).

To avoid these negative effects, the development of tourism in an MPA should be kept within the limit of the carrying capacity of the ecosystem (Heberlein *et al.* 1986; Marion and Rogers 1994; Davis and Tisdell 1995; Weaver 2002; Zakai and Chadwick-Furman 2002; Ormsby *et al.* 2004), a biological concept that has been extended by several authors to social considerations, concerning the relations between local communities and tourists (Murphy 1983; Heberlein *et al.* 1986; Stewart 1993; Arrow *et al.* 1995).

To this end, it is necessary to control numbers and mobility of visitors inside the park, to minimize the impact that high tourist numbers have on the quality of the environment and on local communities (Agardy 1993; Carter 2003). A variety of management tools may be used for this purpose (A.B.A.R.E. 1993), e.g. zoning of activities, access fees, permits for some activities. Artificial reefs are sometimes used (Brock 1994), in order to limit the impact of diving on natural sites.

1.4.4. User fees

Introducing user fees is a way to regulate access to scarce resources. It may therefore help to prevent overcrowding and other negative impacts on ecosystems due to excessive numbers of tourists. It may also be a way to capture part of the consumers' surplus, in order to make the protected area self-sustaining, i.e. to finance management costs and conservation. User fees have for a long time been used in terrestrial parks (e.g. Wicks and Crompton 1986; Lindberg 1991; Lindberg and Huber 1993; Chase *et al.* 1998; Anon. 1999a, 1999b; Vourc'h and Natali 2000; Sibly 2001; Alpizar 2006), and are now increasing being applied in MPAs (Mitchell and Barborak 1991; Dharmaratne *et al.* 2000; White *et al.* 2000; Graham *et al.* 2001; Lindberg and Halpenny 2001; Arin and Kramer 2002; Bhat 2003; Green and Donnelly 2003).

According to Costanza *et al.* (1997), coral reefs could generate high economic returns from recreational activities. However, the actual financial returns are usually quite low since implementing access fees for recreational activities is sometimes considered unsuitable for the marine environment. Nevertheless, examples of successful user fee systems can be found in several locations, such as Bonaire (Netherlands Antilles), Sabah (Malaysia), Palau, Galápagos, and other sites in Africa, the Caribbean and Asia (Tongson and Dygico 2004). For example, in the Tubbataha Reef National Marine Park (Philippines), after two years of fee collection, the total fee collected covered 28% of the annual recurring costs and nearly 41% of the core costs to protect the reefs (Tongson and Dygico 2004).

1.5. Distributional effects of MPAs

The creation of an MPA does not only modify the global amount of welfare provided to society by natural assets, but it also affects its distribution, a consequence which is made obvious in the case where the creation of the MPA is regarded as an opportunity to reconsider the use rights of various stakeholders (Andaloro and Tunesi 2000; Milon 2000; Carter 2003).

Even in the favourable case where the global net surplus of welfare generated by an MPA is clearly positive, its distributional impact may be unfavourable to some groups of stakeholders. Analysing this impact makes it possible to set up compensatory measures. It also underlines the importance of institutional factors.

1.5.1. Distributional problems

The impacts of MPAs on various groups of stakeholders differ. This may result in conflicts and often makes local communities reluctant to support MPA projects (Emerton 1999; Dobrzynski and Nicholson 2000; Sabourin and Pennanguer 2003).

The development of tourism may help improve the welfare of local communities, provided a substantial part of its benefits accrue to these communities (Munasinghe and McNeely 1994). However, this is not necessarily the case, particularly in developing countries, where the opportunity cost of conservation for local communities is an important issue, due to the scarcity of jobs (e.g. Hough 1988; Tisdell 1991; Fauzi and Anggraini Buchary 2002; Ferraro 2002). Moreover, the development of tourism may have adverse socio-cultural impacts on the local communities (e.g. Badalamenti *et al.* 2000). Frustration may grow in these communities, particularly if they are not involved in the decisions concerning the creation and management of the MPAs (Hoagland *et al.* 1995; Sanchirico *et al.* 2002).

The real distributional impacts of an MPA vary according to circumstances. Distributional impacts depend on the size and location of the reserve compared to the fishery, the level of development of the country, and the state of the local labour market. However, the acceptability of the project mainly depends on its perception by stakeholders. People may regard the creation of an MPA as an exclusion from a space they are accustomed to consider as available for free access (Alcala 1988). Fishers, in particular, may be reluctant to accept the creation of zones where fishing is forbidden while other uses, such as scuba-diving, are authorized (Mondardini 1998). In some cases, they regard the creation of an MPA as a signal of the end of their activity (Dobrzynski and Nicholson 2000). However, in other cases, professional fishers may be an important user group supporting the MPA (Aussedat 1995). Many people feel that the MPA will benefit an elite, to the detriment of the majority (Agardy 1994). This feeling is usually stronger when people do not have an acute perception of the degree of degradation of the ecosystem, a perception that is unevenly distributed among groups of users (e.g. Chuenpagdee *et al.* 2002).

1.5.2. Compensatory measures and participation of stakeholders

If an MPA generates a positive global net surplus, it is possible, according to the Kaldor-Hicks criterion, to adopt compensatory measures in order to correct the negative distributional impact of the MPA on some stakeholders. Such measures may improve the social acceptability of the project (Burtraw 1991; Rettig 1994; Hoagland *et al.* 1995). They also help to preserve the belief in the fairness of institutions, which may improve the likelihood of compliance with the rules, and limit the transactions costs related to the change (e.g. Blume and Rubinfield 1984; Farrow 1996).

In the case of MPAs, a wide range of compensatory measures are considered: money transfers (National Park Service 2000), building of harbour facilities (Rettig 1994), assistance to the increase spatial access for boats (Carter 2003), assistance with the development of alternative fishing activities (Tunesi and Diviacco 1993; Rettig 1994), assistance with converting to or diversifying into tourism-related activities (Watson *et al.* 1997; Alban 1998; Vogt 1998; Alban *et al.* 2004), buyback programs (Sanchirico *et al.* 2002), allocation of exclusive use-rights, such as catch quotas (Rieser 2000) and territorial use rights of fishing (Andaloro and Tunesi 2000). However, some of these measures may have undesirable effects (Rettig 1994). According to Roberts *et al.* (2001), this helps to explain why direct money transfers are seldom used in developing countries. Several authors also underline the fact that compensatory measures should be transitional (e.g. National Research Council 2001).

The institutional context is often regarded as a key-factor in the success of an MPA. Numerous authors stress the importance of the participation of stakeholders in the decision making process concerning the creation and management of the MPA (e.g. Norse 1993; Stewart 1993; Hoagland *et al.* 1995; Wells and White 1995; Beaumont 1997; Brown *et al.* 2001; Pollnac *et al.* 2001; Fauzi and Anggraini Buchary 2002; Alban *et al.* 2004).

2. Methodological aspects

The second part of this literature review deals initially with cost-benefit analysis and its application to MPAs. It then focuses on the problem of valuation of non-market values.

2.1. Cost-Benefit Analysis: principles, implementation problems, and application to MPAs

As discussed above, various reasons may lead to the creation of an MPA. These can be summarised as (i) ecosystem preservation, (ii) fisheries management, and (iii) development of recreational non-extractive activities ("ecotourism"). An MPA may be created for combinations of these reasons, but simultaneously pursuing various objectives implies taking into account the complexity of the consequences induced by the creation of the MPA, and the possible conflicting interests of various stakeholder groups (Polunin *et al.* 2000).

From an economic point of view, assessing the consequences of a public project, e.g. creating an MPA, should take two criteria into account (Squire and van der Tak 1985):

1. *efficiency*: what is the global surplus generated by the project, i.e. the net additional wealth that one may expect it will generate for society?

2. *equity*: how will costs and benefits related to the project be shared between social groups, and what type of compensatory measures might be set in order to compensate the groups that might suffer from the project ?

Assessing the surplus generated by a project and the way it will be shared among members of society falls within the scope of economics. Value judgements concerning the fairness of this distribution (and, as a consequence, with the relevance of possible compensatory measures that might correct this distribution) are typically a political topic. In this field, the professional competence of the economist does not go further than characterising the possible impact of various compensatory measures.

The classical approach to assessing the global surplus generated by a public project and its distribution within society is called *cost-benefit analysis* (CBA). After a short presentation of the principles and problems of implementation of this method, we survey the question of its application to the case of MPAs.

2.1.1 Principles and problems of implementation

Briefly summarised, the philosophy of CBA (e.g. Bénard 1985) consists of:

1. making a census of all stakeholders (individuals or groups affected, either directly or indirectly by the project);

2. for each stakeholder, valuing the *market and non-market* costs and benefits induced by the project *in monetary terms*, and computing the resulting balance;

3. aggregating these balances in order to calculate the global surplus of the project.

As an aid to decision-making, the method is generally used for calculating several alternative scenarios. The scenario that is normally selected is the one with the highest global surplus, provided that it respects a set of constraints that should be specified *ex ante* (one of these scenarios may be the *status quo*).

Many problems may arise when implementing CBA, with an intensity varying according to the nature of the considered project. Part of these problems falls within the general scope of project analysis, whether these projects are private or public. In this field, difficulties may concern the determination of the variables and relations that are liable to influence the yield of the project (production functions, costs, demand), and, *a fortiori*, the anticipation of their evolution. Other problems are specific to the public character of the projects that are assessed through CBA. These include:

a) the problem of determining precisely the whole set of stakeholders

Not only direct stakeholders (producers, financing bodies, direct users) should be taken into account. It is necessary to determine the consequences of the project for the whole society (Gaudement and Walliser 1983), which implies for example taking into account upstream producers, consequences for the balance of external trade, or the state budget. The «effects method» (Chervel and Le Gall 1981), which makes use of tools such as input-output tables, aims to calculate chain perturbations generated by a public project throughout the economy of the country. However, the theoretical foundation of this method is a controversial subject (Bénard 1985), and, from a more practical point of view, the data it requires are not always available, and their accuracy or reliability may be questionable. Moreover, in some cases, the very nature of stakeholders may be hard to define precisely. For instance, in the case of a project aiming at protecting a site with a heritage character, the whole set of beneficiaries may be much larger than the population of the visitors of the site.

b) The problem of expressing non-market costs and benefits in monetary terms

Non-market costs and benefits may be related to two kinds of values:

〈 non-market use values, which are related to non-market human activities (e.g. value of fish caught by a recreational fisherman, or opportunity cost of time which he devotes to this activity);

〈 non-use values, which are essentially non-market: existence, bequest and option value (e.g. value of biodiversity).

Non-market costs and benefits are usually critical to the evaluation of a public project. However, the underlying values, unlike market-values, are not naturally expressed in monetary terms. Several techniques have been developed to address this problem. These include:

〈 hedonic pricing and travel costs, which consist of measuring indirectly the value that is attributed to a non-market benefit on the basis of the price that the beneficiaries actually accept to pay for an associated market benefit;

〈 the contingent valuation method, which aims at directly measuring the value attributed to a non-market benefit by asking concerned people what would be their willingness to pay for it, or what would be their willingness to accept in exchange of being deprived of it.

These techniques are far from perfect. Their scope is not general, their methodology may be difficult to implement, and the reliability of their results is a controversial matter (for more details on travel cost and contingent method, see section 2.2. below).

c) The problem of correctly expressing some market values

If market values, unlike non-market values, may be directly observed in monetary terms, the results of these observations do not necessarily provide results that can be considered as relevant for CBA. This problem is twofold:

〈 At a given date, observed (actual) prices may not represent correctly the social costs and benefits of scarce resources that are used and of commodities that are produced, because of various distortions on markets (non-neutral taxes, subsidies, monopolistic positions, custom rights, politically determined exchange rates, externalities...). The so-called «reference prices method» (Squire and van der Tak 1985), which consists of computing a set of shadow prices on the basis of an optimisation program (representing the objective-function and the constraints of the project) is theoretically satisfying but may be in practice difficult to implement.

〈 Costs and benefits which the project generates at various time periods may be compared only through the actualisation technique. The level of the adopted discount rate may influence heavily the results of the analysis, especially if the project is to have important long-term consequences. The choice of this rate, for public projects is difficult and controversial. The terms of the problem may be briefly described as follows: the more scarce is the saving for funding the project, the higher the discount rate should be; at the same time, a high discount rate automatically disregards long term effects of the project, which may create a «short-sight» bias in the assessment of the project, but also a problem of intergenerational equity (for a discussion of the role of the time-discount rate in the context of fisheries management, see e.g. Hannesson 1993).

In order to mitigate some of these difficulties, a less ambitious method, «cost-efficiency analysis», is sometimes used. According to this method, the objective to reach is exogenous, and the various scenarios are ranked according to their social cost.

2.1.2. Application of CBA to MPAs

There is a wide scope for the use of CBA in the field of environmental economics (Hanley and Spash 1993). The creation of a MPA typically falls within this scope (Hoagland *et al*. 1995; Sanchirico 2000; Parrish *et al*. 2001; Carter 2003).

In the following development, the problem of assessing the net benefit due to a MPA is not considered in a fully general way. Only costs and benefits related to fishing and ecotourism, i.e. use values, are considered. The objective of ecosystem protection is treated as a constraint of sustainability in the use of natural resources by fishing and ecotourism. This implies that the non-use value (existence value, option or quasi-option value) of the ecosystem is not taken into account explicitly (however, in the case of an optimal management program, this value may be revealed by the level of dual variables associated with environmental constraints). This approach is certainly too limiting, and results in under-recording of the net surplus generated by an MPA (Emerton 1999). Yet measuring the non-use value of biodiversity is still confined to theoretical grounds, and, in practice, this type of value is not taken into account when attempting to assess the net economic surplus generated by an MPA (Hoagland *et al*. 1995).

Furthermore, the analysis is limited to the direct consequences of the MPA on fishing and ecotourism, relying on the assumption that a set of economically significant prices may be used to evaluate these activities.

As regards fishing, a problem may arise due to the fact that, in some cases, a significant part of catches is not marketed. In the case of subsistence fishing, landings should be valued by using ex-vessel market prices for similar products landed by commercial fishing boats -which, in practice, may raise some problems. In the case of recreational fishing, this solution does not apply, because recreational fishers do not value their activity on the basis of the fish consumption it provides, but merely for the leisure it represents. When recreational fishing takes the form of charter fishing, its economic value is normally represented in the price paid by customers to chartering companies. In the frequent case where recreational fishing is a non-market activity, it is necessary to value the willingness to pay of fishers by appropriate methods, such as the contingent valuation method, the travel cost method, or a combination of both methods (Layman *et al*. 1996).

For tourism, valuing costs and benefits generated by the MPA raises problems which are often more complex than in the case of fishing, because tourism may cover a wide range of activities. Due to the nature of the problem under survey, it is possible to restrain the investigation to the case of “ecotourism”, i.e. a form of tourism that is generated by the quality of the protected ecosystem of the MPA. Relevant valuation of the benefit generated by each visit to the MPA is available (net income resulting from services to ecotourists, and, in where there are access fees, net income of the management body of the MPA).

Assessing costs and benefits of the MPA for fishers mainly relies on the following considerations:

〈 The prohibition of fishing in the no-take part of the MPA reduces catch, *ceteris paribus*, in the short term. This is the negative effect of the MPA for fishers. The economic importance of this drawback varies according to the degree of dependency of fishers on the zone affected by the prohibition.

〈 On the other hand, by protecting a part of fish stocks against fishing mortality, the fishing ban favours an increase in biomass in the no-take part of the MPA, which is likely to induce a net transfer from the no-take zone to the fishing zone (spillover effect), thereby increasing the catch per unit of effort (CPUE) in this zone. This favourable effect of the MPA depends on two types of factors: biological parameters of the targeted stocks (recruitment, natural mortality, space mobility), and the level of fishing mortality (hence of fishing effort) in the zone left open to fishing. This factor is critical when assessing the economic performance of the MPA: if the fishing zone is under open access, the increase in CPUE will be followed by an increase in effort, and this process will normally go on until the rent resulting from the spillover effect is totally dissipated (Hannesson 1998). If the increase in fishing effort can be prevented (e.g. by a system of limited entry licenses in the fishing zone), the rent dissipation will not happen, and the MPA may improve the situation of the fishery, both in economic and in biological terms, even though this fishery suffers from some overcapacity (Holland and Brazee 1996; Guénette and Pitcher 1999; Boncoeur *et al*. 2002).

〈 The existence of a zone where fish resources are protected from fishing mortality may also be considered as an application of the precautionary principle to fisheries management (Lauck *et al.* 1998), limiting the risk of accidental collapse of the fishery by the constitution of a “buffer stock”, or safe minimum biomass level (Anderson 2002).

〈 Benefits of the MPA for fishers will be maximized if the location of the no-take zone is such that it protects a critical zone for the stock renewal. On the other hand, these benefits may be jeopardized by the opportunistic development of predators (Boncoeur *et al.* 2002).

As for ecotourism, it is generally expected that protecting marine ecosystems inside an MPA will attract visitors. In order to measure the benefit of the MPA for the tourism sector, it is necessary to determine a quantitative relationship between some attributes of the ecosystem and the number of visitors to the MPA (Dixon *et al.* 1993). A reasonable assumption is that such a relationship is an increasing, but non-linear, function due to congestion phenomena that appear once a certain level of ecotourism activity or visits is reached. Moreover, it is necessary to take account of the negative consequences increasing tourism may have on the ecosystem, specially when the increase is left uncontrolled.

In summary, bearing in mind the simplifying assumptions presented above, the net social surplus generated by a MPA may be expressed as:

〈 The fishery rent generated by the transfer of increased fish biomass from the no-take zone to the fishing zone
〈 *Minus* the negative impact on catch due to the decrease in the zone open to fishing
〈 *Plus* the tourist rent generated by the improvement in the quality of the ecosystem of the reserve

All these benefits and costs should be estimated under the constraint of sustainable use of the ecosystem resources. This implies bio-economic equilibrium conditions, but reaching equilibrium usually involves a transitional period, which raises concerns with respect to the time-discount rate.

2.2. Valuation techniques for non-market values

Several techniques may be used to assess non-market values. For MPAs, the two most relevant ones are the travel cost method and the contingent valuation method. After presenting each one of these two techniques, we will present some attempts to combine them.

2.2.1. The Travel Cost Method (TCM)

This method is based on the observation of actual behaviours of individuals frequenting a given site, i.e. revealed preferences (Clawson 1959; Clawson and Knetsch 1966). It is increasingly used to assess the benefit associated with sites that provide recreational uses (fishing, diving, etc.). TCM relies on the assumption that time and costs associated with travel form the shadow, or implicit price for visiting the site. The relationship between this shadow price and the quantity, i.e. the frequency of visits to the site, expresses the shadow demand function for the site. This function may be estimated on the basis of data gathered by concentric zones around the studied site (zone travel cost approach) or of individual data (individual travel cost approach). The traditional application of TCM consists of measuring the consumers’ surplus associated with the visits to the site. This can be done by defining the consumers’ surplus as the surface included between the demand function and the shadow price line (Freeman 1993).

In connection with public policy concerns, TCM is often used to assess the effects of a variation in the quality of a site. Improving the quality of a site, for instance, results in shifting the demand curve towards the right. As a consequence, if the unit travel cost is left unchanged, the number of visits increases. The value of this improvement is then measured by the economic surplus variation.

However, it may be difficult to identify precisely the changes in the demand function associated with an improvement of quality. The solution is to use a model with variable parameters integrating quality in the demand function (Vaughan and Russel 1982; Smith and Desvougues 1985). This procedure includes two stages: i) sites with different qualities are taken into account, and, for each site with a given quality, a demand function is estimated; ii) these demands are compared, in order to estimate the shift in the demand curve following a quality variation.

In spite of continual improvements, the applicability of TCM still raises some problems. One of them is related to the definition of the quality variable. Quality measurements usually rely on objective data (e.g. levels of chemical or organic pollution). This procedure assumes that individuals determine their recreational choices according to known objective data. However, this assumption is highly questionable, in particular concerning medical or sanitary quality: there may be an important difference between the objective quality of a site and its perception by its users. This phenomenon was clearly highlighted, for example, by a survey aiming at determining the perception of the health risks connected with consumption of shellfish picked on the strand by recreational fishers (Appéré and Bonnieux 2003). Other problems are related, *inter alia*, to the valuation of travel time (Cesario 1976), the taking into account of people staying several days to attend the site (Cameron 1992), or of the purchase of durable recreational equipment.

Relying on actually observed behaviours, TCM suffers from a lack of flexibility, in particular in the case of prospective valuation of potential situations which will occur only in the future. As it depends on the levels of quality actually observed at the sites, the range of possible quality variations is constrained by observations, which may not correspond to the ones that are expected from an improvement project.

Some alternatives to the classical travel cost method were recently proposed to overcome some of these problems. The methods of discrete choices associated with travel costs are increasingly used, due to their flexibility and to their ability to estimate the benefit related to a change in recreational sites quality. Based on the Random Utility Model, these approaches assume that users choose the site they consider the best; this choice reveals their relative preferences between the characteristics of a site and the cost to reach it. By combining the data relating to the choices of site with the costs and the characteristics of all alternative sites, it is then possible, generally with the help of Logit or Probit models, to estimate the relation between the choice of a site and costs and characteristics of all alternative sites. However, this alternative method requires information on all the possible sites that the visitor could select, in particular on their characteristics in terms of quality, and on travel cost for each site. For some recreational activities, this condition can be rather difficult to satisfy, especially when many sites may be used as substitutes. Another alternative is to combine TCM with contingent valuation (see 2.2.2 and 2.2.3 below).

2.2.2. The Contingent Valuation Method (CVM)

This method appeared approximately at the same time as the travel cost method: the first known survey goes back to 1961 and was undertaken by Davis (1963). The essence of CVM is to collect stated preferences, i.e. to value directly the variations of the quality of a good by asking individuals their willingness to pay (WTP) for an improvement concerning this good, or their willingness to receive (WTR) in compensation for a degradation of this good. These values are obtained by interviewing users, to whom a hypothetical scenario specifying the object to be valued is presented.

The major advantage of CVM lies in its great flexibility, making it possible to apply it a large range of situations. Its success may also be explained by the fact that it meets an increasing need for expertise concerning environmental valuation, following the evolution of legislation in Western countries.

However, CVM was originally criticized due to the possible existence of biases in the method. If some of these biases (e.g. the strategic bias) are regarded nowadays as less serious than originally supposed, others continue to raise problems. To this category belong the hypothetical and instrumental biases.

Particularly in the field of environmental assets, there may be a problem with the means of payment used by the CV scenario: simple means such as a tax increase, an increase in the water invoice, or a voluntary contribution, are sometimes badly felt by interviewed people. Not only does the scenario make them imagine a situation where they would have to pay for the use of a good that, in many circumstances, was formerly regarded as free, but the means of payment used in the scenario gives them the unpleasant feeling that they are submitted to a sort of “polluter pays” principle. It often results in a high rate of “false-zero” answers (zero-WTP), which means that some respondents provide a willingness to pay equal to zero because they refuse the scenario suggested (“protest response”), not because they regard the proposed improvement as valueless.

Moreover, requesting users to provide a monetary value for the improvement of an environmental good, which generally is perceived as free, requires very developed cognitive capacities from the users. There is a risk of obtaining aberrant and/or non-consistent values (e.g. characterised by the bias of inclusion). Statistical procedures are designed to “eliminate” these values, but they often appear to be artificial and lacking theoretical bases.

More generally, CVM has always raised a strong scepticism among some economists, decision makers or experts, because it is based on stated preferences: “it would produce hypothetical answers to hypothetical questions” (Cameron 1992).

This mistrust, as well as the possible existence of biases, has generated a long series of studies. Attempts have been made to compare the values obtained by CVM with those obtained by other valuation methods. For instance, Bishop and Heberlein (1979), Heberlein and Kealy (1983) compared the estimates based on CVM and by TCM. According to these studies, data obtained by CVM seem valid and credible (Whitehead *et al.* 1999). Loomis (1993) showed that both methods do not give statistically different results when the quality of a site is changed. However, the results obtained by the CVM can sometimes be prone to certain biases such as the hypothetical bias (Cumming *et al.* 1997; Diamond and Hausman 1994).

2.2.3. Combining TCM and CVM

Considering TCM and CVM, it appears that each of these two methods suffers from drawbacks that limit their scope and credibility. This has led to suggestions that aim at combining both methods (Cameron 1992). According to this new approach, TCM provides data on actually observed behaviours, while CVM provides insights on their probable behaviour in hypothetical situations that are defined by the contingent scenario –which may substantially differ from those that are usually observed.

This procedure was later improved, and lead to what is known as the hypothetical travel cost method (Layman *et al.* 1996). The first step is to estimate the aggregated demand curve for a site using the TCM. The next step is to submit a hypothetical scenario considering different conditions (e.g. quality levels of the site) to a sample of users asking them to indicate what would be their behaviour (frequency of visits to a site) according to those hypothetical conditions. The frequency of visits to a site according to various levels of environmental quality becomes the contingent variable, instead of the traditional willingness-to-pay sum. These contingent data are then combined with the information previously obtained by TCM (in particular the distance covered), to calculate indirectly the average willingness-to-pay for the considered variation of quality.

Contrasting with traditional CVM, in the hypothetical TCM no question involving price and means of payment is explicitly asked. The respondents are questioned on variables that are familiar to them, namely the variations in their frequency of visiting a site. As a result, this procedure is more realistic and accessible than traditional CVM, and avoids some of the problems that this method usually raises: strategic behaviour, refusal to pay, and, probably, hypothetical bias. Symmetrically, the hypothetical TCM, which applies to a class of problems traditionally analyzed by TCM (restoration or degradation of environmental assets for which are already visited and requiring travel), widens the scope of this method by including potential effects of events affecting the attributes of the studied site (Appéré 2002).

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