

Marine protected areas and managing fished ecosystems

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Summary

This paper provides a synthesis of the current literature on the potential of marine protected areas (MPAs) as a management tool to limit the ecosystem effects of fishing, including biological and socio-economic perspectives. There is sufficient evidence to show that fishing can indeed negatively impact ecosystems. Modelling and case studies show that the establishment of marine protected areas, especially for overexploited populations, can mitigate ecosystem effects. Although quantitative ecosystem modelling techniques incorporating MPAs are in their infancy, their role in exploring scenarios is considered crucial. Success in implementing MPAs will depend on how well the biological concern, and the socio-economic needs of the fishing community are reconciled.

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Price: NOK 50 + postage

ISSN 0805-505X

ISBN 82-90584-45-8

Indexing terms

Ecosystems

Fishery

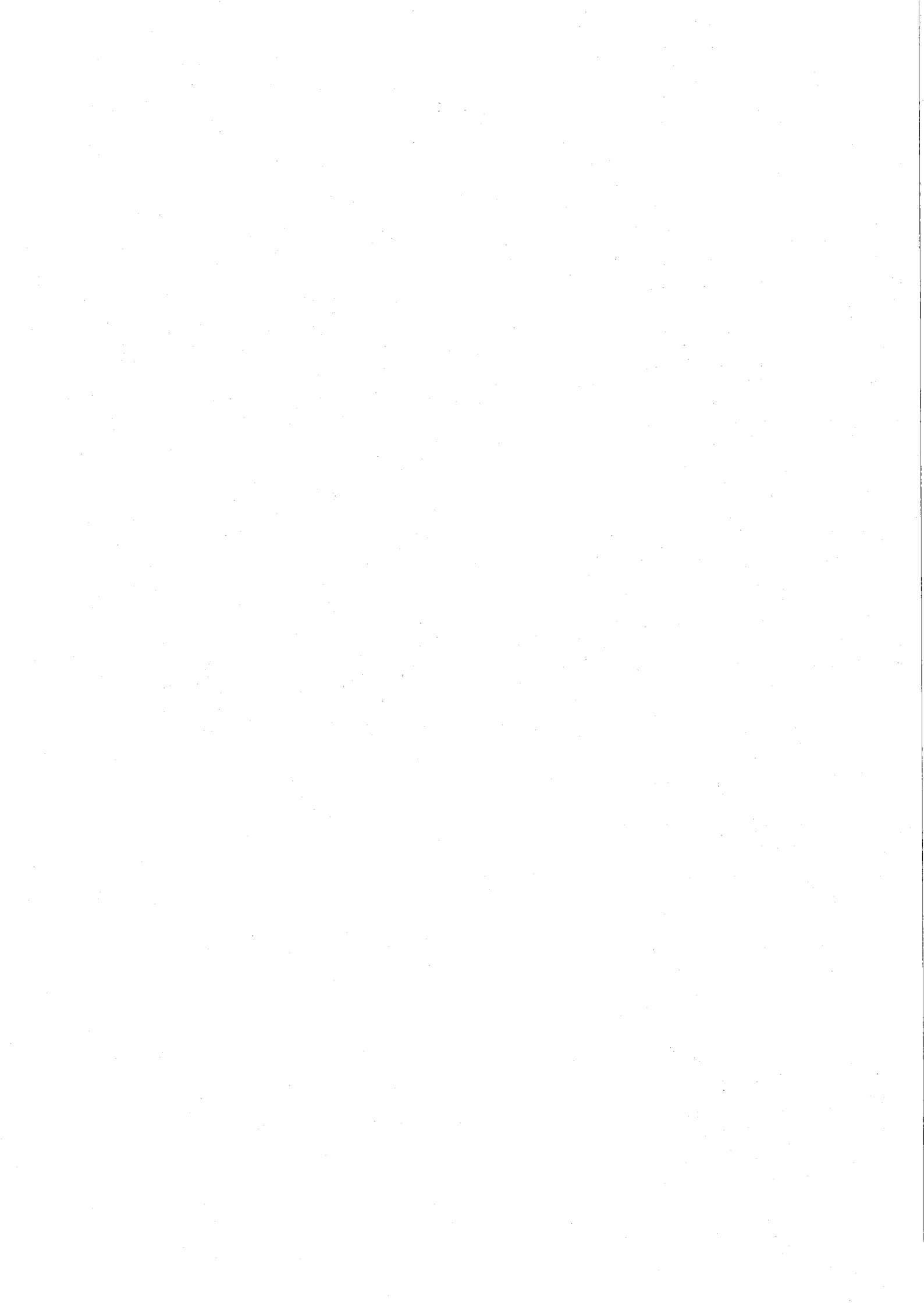
Marine reserves

Marine protected areas

Community participation

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Approaches and goals in establishing marine protected areas¹

Traditional living resource management includes the setting aside of areas from exploitation in both terrestrial and marine systems. Such areas are set aside to ensure the continuity of stocks for future generations and these practices are still being employed in developing countries throughout the world. The notion of setting aside protected natural areas solely for their scenic, natural or scientific values, however, is a relatively recent trend (MacEwen and MacEwen, 1982). Currently there are approximately 4,500 recognised protected areas (as per IUCN definitions) around the world. Of these, however, only about 850 include a coastal or marine component (Elder, 1993).

The first recorded attempts to establish marine protected areas (MPAs) were early this century. In 1906 an attempt at regulating the collection of marine organisms within three-quarters of a nautical mile of a recreation reserve at Green Island on the Great Barrier Reef, Australia, was reported (Morning Post, 1906). Fishers rejected the proposal then. It was not until 1937 that the island and the waters within one nautical mile of it were protected using the Fisheries and Oysters Act, which basically closed the area to all forms of fishing. The first marine protected area declared using dedicated legislation, including provisions for management of surrounding waters and their biota, was at Fort Jefferson National Monument, Florida, which was declared in 1935 (Randall, 1968). The legislation used to protect this area of the Dry Tortugas was primarily designed for terrestrial systems. It was not until the post-war era that more parks with significant marine areas were established (Bjorklund, 1974). These areas, however, were also primarily protected using terrestrial legislation. Indeed, even today few MPAs are declared using MPA-specific legislation (Alder, 1996).

The use of MPAs in managing marine resources such as those used for fishing was not widely applied until the 1970s. Although many nations recognised the economic potential of their marine environments, they did not perceive the exploitation of marine resources to be a threat to the sustainability of these coastal and marine areas. Their desire for MPAs was therefore based on the ideals of natural beauty, or scientific research, as in terrestrial protected areas. This trend continued until nations began to look more towards the sea for economic growth and new food sources, which resulted in heavy fishing pressure on the world's ecosystems, especially from distant water fleets (Bonfil *et al.*, 1998). One of the effects of this pressure was to galvanise political action worldwide, leading to the ratification of the Law of the Sea Convention in 1982, for instance. Once this occurred the role of MPAs became increasingly distinct from that of

¹ It should be noted that in this paper, a marine protected area (MPA) refers to a management area in which usage is regulated by zoning for different activities. It includes marine reserves, which are strictly no-take areas.

terrestrial protected areas. Today, the development of MPAs within an Integrated Coastal Areas Management Plan or resources management strategy generally takes the form of a representative systems of MPAs aimed at contributing to the maintenance of biodiversity, ecological processes, and sustainable resource usage.

Although signs of overexploitation in most of the world's fisheries (Ludwig *et al.*, 1993; Safina, 1995) raise serious concerns about the efficacy of current fisheries management strategies, we still have to formally address the effects of fishing on entire ecosystems. Marine reserves, areas closed to exploitation, are seen as an additional management tool that could control fishing mortality (Bohnsack, 1996; Guénette *et al.*, 1998a) and thus hedge against the risk of fisheries collapse (Bohnsack, 1993; Clark, 1996; Guénette *et al.*, 1998a; Sumaila, 1998c). In tropical fisheries, where numerous species prevent managers from applying single-species stock assessment techniques, closed areas may be the only available tool (Roberts and Polunin, 1993a; Williams and Russ, 1995). Fishing throughout an ecosystem exposes us unnecessarily to the vagaries of uncertainty, and to the consequences of genuine management mistakes. In effect, such fishing practices deprive us of any insurance policy against fishery collapse (Clark, 1996; Lauck, 1996; Sumaila, 1998c).

In terms of socio-economics, the following narrative captures the issue at stake: A journalist once asked the Minister of Fisheries in Namibia how he planned to handle the tradeoffs between the needs to conserve Namibia's fishery resources and the need for maintaining high levels of employment in the fishing sector of the economy. The Minister countered (we believe rightly) that the question missed the point: The issue, according to the Minister, was not "conservation vs employment" but rather "employment today vs employment tomorrow" (Namibia Brief, 1994). Given the collapses of various fish stocks around the world (e.g. Atlantic cod off Newfoundland) and the scientific evidence gathered so far (see for instance, Safina, 1995; Pauly *et al.*, 1998), it is almost certain that, at current global fishing levels, we are unnecessarily sacrificing tomorrow's employment for today's.

In broad terms, this paper is made up of two main parts: a part that provides a synthesis of what the current literature says about how MPAs may be used to limit the ecosystem effects of fishing; and one which briefly presents a number of promising quantitative modelling methods (either current or being developed) for the assessment of marine reserves as ecosystem/fisheries management tools. Integrated in these two parts are issues addressing socio-economic effects of fishing practices and how these might change as MPAs are implemented. We end the paper by presenting some considerations about the establishment of MPAs with suggestions on how to move forward. It is beyond the scope of this paper to extensively review either the use of marine reserves in fisheries/ecosystem management or the effects of fishing on populations and ecosystems; instead, the objective is to focus sharply on how marine protected areas may be employed to mitigate against the effects of fishing. Comprehensive reviews have been published recently on both marine reserves and the effects of fishing. See, for instance, Hall (1999), Dayton *et al.* (1995), Roberts (1995a), Jones (1992) and Hutchings (1990) on the effects of fishing; and Guénette *et al.* (1998a), Attwood

et al. (1997), Bohnsack (1996), Roberts (1995), Rowley (1994) and Dugan and Davis (1993) on marine reserves and MPAs. The reader is also referred to a forthcoming special issue of the *Bulletin of Marine Science*, which will contain selected papers presented at the 2nd Mote Symposium on Essential Fish Habitat and Marine Reserves, Sarasota, Florida, in November 1998.

Effects of fishing

The ecosystem effects of fishing may be classified into three broad groups, that is, if we include humans as part of the ecosystem. These are the effects on (i) the conservation of species, including maintaining fish populations above certain critical thresholds, enhancing the possibility of egg and larval exportation and adult dispersal, and hedging against natural and anthropogenic disasters and uncertainty; (ii) the conservation of marine habitats, protecting them from degradation resulting from fishing activities, and preserving marine biodiversity, healthy ecosystems and critical habitat; and (iii) the maintenance of sustainable employment and economic activity based on marine resources.

Conservation of species

Fisheries management has generally focussed on one single species at a time. It has aimed at maximising yields for the fishing industries while preserving the targeted species or stocks, and balancing the needs of different users while considering the social and economic imperatives (Hilborn and Walters, 1992). Various management reference points have been developed (Deriso, 1987; Sissenwine and Shepherd, 1987; Patterson, 1992; Mace and Sissenwine, 1993; Smith *et al.*, 1993; Mace, 1994; Myers *et al.*, 1994), but most of them still rely on accurate estimation of the stocks and adequate models, as well as efficient control of effort and catch. These strategies typically underestimate the importance of uncertainties in stock assessments, population dynamics and environmental processes, which often result in overfishing (Hilborn and Walters, 1992; Ludwig *et al.*, 1993; Rose, 1997; Lauck *et al.*, 1998). In addition, management schemes achieve only partial success in controlling effort and/or catch, in some cases inciting fishers to cheat by misreporting, discarding and upgrading (FRCC, 1996; Munro *et al.*, 1998). Reducing effective fishing effort is almost impossible in the face of improving technology (Hilborn and Walters, 1992; Parsons, 1992; Ludwig *et al.*, 1993; see also Pitcher, in press). Adequate stock assessments are often impossible because of changing spatial population distributions of fish that distort catch per unit effort data (Radovitch, 1979; Saville, 1980), sampling variance and misreporting of the catch (Larkin, 1977; Ludwig *et al.*, 1993). To be effective, the stock assessment has to be done quickly enough to be used in the quota setting process for the next year (using real time information) (Walters and Pearse, 1996). Unfortunately, there seems to be an inherent time lag between stock assessment and quota setting (Fahrig, 1993), increasing risks of overfishing in the case of variable recruitment. For all of these reasons, overexploitation is frequent, even in countries where large amounts of resources have been assigned to management science and stock assessment.

The effects of overexploitation on a species are well known: diminished biomass, decrease in mean body size and age at maturity, and an unbalanced sex-ratio in protogynous species (Heessen, 1988; Buxton, 1993b). As the stock is

depleted, the age structure is truncated, which reduces the number of larger sizes and experienced spawners capable of producing more numerous and better quality eggs (Kjesbu, 1989; Solemdal, 1997; Marshall *et al.*, 1998). Truncated age structure may also influence the potential reproductive success by shortening the spawning season for both males (Trippel and Morgan, 1994) and females (Hutchings and Myers, 1993). In some species, a critical concentration of adults is necessary to ensure successful breeding (Rogers-Bennett *et al.*, 1995) or the survival of settling larvae (Tegner and Dayton, 1977; Davis, 1995). All of these make overexploitation even more dangerous. Although other factors such as climatic variations may have profound effects on fish populations (Cury and Roy, 1989; Kawasaki, 1992; Bakun, 1996; Klyashtorin, 1997; Springer and Speckman, 1997), fishing has been found to be a decisive factor in stock collapses in many cases (Pauly *et al.*, 1987; Heessen, 1988; Parsons, 1991; Hutchings and Myers, 1994; Myers *et al.*, 1996; Orensaz *et al.*, 1998).

Fishing for one species often implies catching other species which share the same habitat and are vulnerable to the same fishing gear (Brander, 1988; Alaska Sea Grant College Program, 1996). The problem may arise from the gear and methods used, the fishing season (Alderstein and Trumble, 1998), and/or the management regime (Crowder and Murawski, 1998). It is very difficult to find a set of regulations that would be practical and efficient in reducing such by-catch on multi-species fishing grounds. Single-species management, by using size-limits, mesh-size, quotas and by-catch limits, compels fishers to discard fish when one of these limits have been reached (Brander, 1988). The effects of by-catch on long-lived species, such as rays and sharks, are likely to be high as these species are affected by even a low fishing mortality rate (Brander, 1988; Pauly, 1988; Casey and Myers, 1998; Fogarty and Murawski, 1998). Similarly, catching the unwanted juveniles of a target species may be detrimental to both the population and the fishery by augmenting mortality of these juveniles (Reise, 1982; Riesen and Reise, 1982; Garcia and Demetropoulos, 1986; Myers *et al.*, 1997; Alderstein and Trumble, 1998).

Looking at only one species at a time, we often fail to realise the significance of serial depletion, of individual stocks and fishing grounds, as illustrated by fisheries in all parts of the world including the Gulf of Alaska (Orensaz *et al.*, 1998), the Cuban shelf (Claro, 1991), the Jamaican reef (Koslow *et al.*, 1988), the California Coast (Dugan and Davis, 1993), New Zealand (Ballantine, 1991), Lakes Victoria and Malawi (Craig, 1992), Georges Bank (Fogarty and Murawski, 1998), New England (Brailovskaya, 1998), the Gulf of Thailand (Pauly, 1988) and the North Sea (Daan, 1980; Heessen, 1988). These changes are not always conspicuous as the total yield may remain the same over time while the relative composition of the catch changes.

Many world fisheries, once targeting long-lived, high trophic level piscivorous fish, are now catching more invertebrates and short-lived pelagic planktivores (Fischer *et al.*, 1997b; Caddy and Rodhouse, 1998; Pauly *et al.*, 1998; Pitcher and Pauly, 1998). The resulting ecosystems are often economically less efficient since secondary species may have less value (Claro, 1991) and a larger proportion of the total catch comes from industrial fishing destined to the typically low-value

production of fish meal (Robertson *et al.*, 1996; Fischer *et al.*, 1997b). Long-term effects on the ecosystem are often gradual and not formally documented, resulting in the absence of baselines needed to evaluate the seriousness of the situation (Pauly, 1995).

Fishing may also have an impact on community structure by altering predator-prey relationships. A good example would be the case of the cod and capelin (*Mallotus villosus*) relationship in the Barents Sea. Heavily fishing capelin, shrimp and herring led to reduced prey availability for cod, which showed decreasing growth and increasing cannibalism (Mehl, 1991; Tjemeland and Bogstad, 1993). Several studies are showing the impact of declining forage fish populations (often due to overfishing) on the survival of marine mammals (Hansen, 1997) and the breeding success of seabirds (Furness, 1982; Duffy, 1983; Anker-Nilssen *et al.*, 1997; Fischer *et al.*, 1997b; Hayes and Kuletz, 1997). The impact is not only restricted to the total abundance of prey but also its spatial distribution and the encounter rate between prey and predators (Furness, 1982; Robertson *et al.*, 1996; Furness and Tasker, 1997).

Fishing may even eliminate trophic groups or keystone species and result in a complete change to the overall community structure (Roberts and Polunin, 1991; Russ, 1991; Done, 1992; Roberts, 1995a; Jennings and Polunin, 1996; Goñi, 1998; Hall, 1999). For example, reef fishing mainly targets large predatory and herbivorous fish, among them triggerfish which feed on urchins, a keystone species of these ecosystems (Roberts, 1995a). Fishing finfish thus results in high concentrations of urchins, at the same time it controls algae (Hay and Taylor, 1985; McClanahan and Shafir, 1990; Jennings and Lock, 1996), and may even increase erosion of the coral reef substrate (McClanahan and Shafir, 1990; Roberts, 1995a). As urchins are more efficient than herbivorous fish, they may suppress the densities of these fish. Conversely, in Jamaica, in the absence of urchins, high fishing pressure on finfish still prevents the herbivorous fish population from recovering and thus helping to control algae (Hugues, 1994). Other examples of disturbance of top-down controls can be found in Botsford *et al.* (1997) and Parsons (Parsons, 1992). Once again, such ecological shifts may also be caused by a combination of environmental factors (e.g. McClanahan and Muthiga, 1998).

Conservation of habitats

Trawls and dredges may modify the sea bed by ploughing, scraping, resuspending sediments, and destroying non-target species (Jones, 1992; Goñi, 1998). Evidence that the use of dredges and bottom trawls is detrimental to demersal habitats and their fauna is however, difficult to gather because of an array of reasons. Often, studies are limited by the lack of unexploited habitats of similar type which can be used as controls (Jones, 1992; Kaiser, 1998). On top of this comes the lack of knowledge about the previous levels of fishing intensity on the studied grounds (Hutchings, 1990). Most studies are carried out over short periods of a few months and thus do not account for cumulative effects (Jones, 1992; Thrush *et al.*, 1995). Also, dredging may have less impact on high energy shallow grounds, which are regularly disturbed by storms, than on deeper-water fishing grounds

which would be more likely to suffer long-lasting effects (Eleftheriou and Robertson, 1992; Jones, 1992). Clear impacts are also difficult to detect over long periods of time because the distribution of fishing effort is patchy, highly concentrated (Rijnsdorp *et al.*, 1996) and mobile (Allen and McGlade, 1986; Hutchings and Myers, 1994). This is further complicated by concomitant possible effects of pollution, eutrophication and variations in climate (Samoilys, 1988; Kaiser, 1998). Finally, impact studies are hampered by the lack of knowledge on these epibenthic communities, beginning with their taxonomy (Hutchings, 1990).

Despite these uncertainties, there are indications that the use of bottom gears may change the structure of the benthos and especially its species composition (Saxton, 1980; Bradstock and Gordon, 1983; Hutchings, 1990; Jones, 1992). It may also decrease bottom complexity by removing those macro-benthic organisms which provide shelter (Auster *et al.*, 1991; Sainsbury *et al.*, 1993; Auster and Malatesta, 1995; Auster *et al.*, 1996). A few dredges or bottom trawl passes are often sufficient to reduce habitat complexity (Auster, 1998; Watling and Norse, 1998), and kill or damage infaunal and epifaunal organisms, while sediments and small infauna such as polychaetes seem to recover after a few months (Peterson *et al.*, 1987; Eleftheriou and Robertson, 1992; Currie and Parry, 1996) or several years (Watling and Norse, 1998). However Poiner *et al.* (1998; 1998) found that each consecutive trawl removes 9-13% of the sessile and mobile benthic invertebrates, and fish communities. Repeated disturbance and mortality of large benthic organisms are likely to prevent any recovery of the vulnerable species, especially the long-lived ones (Dayton *et al.*, 1995), and the species associated with undisturbed highly structured bottoms (Watling and Norse, 1998). Fishing with mobile bottom gears has contributed to tremendous benthic changes in the Wadden Sea (Reise, 1982; Riesen and Reise, 1982). By modifying habitat structure and siltation processes (Jones, 1992; Rothschild *et al.*, 1994), or destroying seagrass habitat, mobile gears have been shown to hamper juvenile settlement for some species (Peterson *et al.*, 1987). Additional hints are provided by new fishing grounds in Australia (Saxton, 1980; Sainsbury *et al.*, 1993) and Arctic Canada (McAllister and Spiller, 1994) where trawls collected massive quantities of large epibenthic organisms that decreased rapidly with time.

In a few cases, the effect of dredging and the subsequent reduction in epibenthic organisms has been linked to changes in relative fish community composition. For instance, Sainsbury (1993) showed that in trawled areas of the Northwest Shelf of Australia, *Nemipterus* and Sauridae were abundant while unfished areas were dominated by *Lutjanus* and *Lethrinus*, which preferred complex bottom structures for cover. Destruction of coral cover on reefs has also been linked to impoverished marine resources and fish stocks (Jennings and Lock, 1996; Vincent and Pajaro, 1997). Temperate demersal fish such as cod, which seek cover to reduce predation, may also be impacted by systematic decreases in bottom complexity (Lough *et al.*, 1989; Fraser *et al.*, 1996; Gregory and Anderson, 1997). Rao (1988) attributed the decline of a marine catfish (*Arius tenuispinis*) to the disappearance of its principal prey, a polychaete, due to incessant trawling of its feeding grounds.

Socio-economic considerations

If we consider fishing communities to be part of the ecosystem, then we can talk about the "socio-economic ecosystem effects of fishing". Many social scientists have convincing arguments to the effect that fishing communities ought to be seen as part of the ecosystem (see for instance, Coward *et al.*, in press). These scientists argue that the fact that fisheries are managed under multiple, usually conflicting objectives, should not be lost sight of. Apart from resource conservation and food supply, ecosystem management goals include generation of employment and economic wealth, income for fishers and the maintenance of viable fishing communities (Charles, 1989; Behnken, 1993).

Few studies have examined the socio-economic impacts of fishing, though a number of studies have quantified the cost of habitat changes on fisheries. However, the economic impacts of destructive fishing practices such as trawling, cyanide fishing and blasting are poorly understood. Cesar *et al.* (1997) studied the economic impact of destructive fishing practices, including poisoning and blast fishing, around Indonesian coral reefs. They found that the benefits to private individuals were high, but the social costs were much higher, up to 50 times larger in the case of blast fishing in tourist areas. Johannes and Riepen (1995) investigated the socio-economic implications of the live reef fish trade in Asia and the Western Pacific where cyanide is used extensively. The profits generated by this form of fishing were high for individual fishers, but only in the short-term. They noted the social costs of cyanide fishing to local communities but they did not put a dollar value on it.

The long term effects of fishing on the economic and social well-being of the fishing communities may be either negative or positive. They tend to be positive if the interaction between the fishing community and the fish is such that the ecological base of the resources remains intact through time; in other words, if sustainably managed. On the other hand, if this interaction degenerates into the destruction of the resource base, as it usually does, then the negative ecosystem effects of fishing hit hard on the community. This negative effect can result in huge dislocation in the economic and social life of the fishing community dependent on the resource. An often-cited modern example of this is the huge economic and social pain that followed the collapse of the North West Atlantic cod fishery off the coast of Newfoundland, Canada.

Several factors contributed to the fishery crises in Newfoundland, such as policy problems of the welfare state, socio-economic crisis of the fishing communities (Ommer, 1994), and inappropriate fisheries policies, resulting from overestimation of the stock (Steele *et al.*, 1992). A moratorium was imposed in 1992, in response to the overfishing situation of the northern cod stock. At that time, the fishing industry was already over-capitalised, both in vessels and in processing plants. It was suggested then that 19,000 fishers and plant workers plus 20,000 others would be directly affected (Steele *et al.*, 1992). This does not take into consideration, however, the effects on social and cultural identity and values of communities with such a long tradition of fishing.

Marine protected areas as a management tool

Ecological factors

From the single species point of view, a marine reserve would be expected to help control fishing mortality and by so doing restore, at least partially, pre-industrial exploitation patterns, when less efficient fishing techniques and lower boat power prevented the exploitation of portions of the fishing grounds. Such reserves would increase resilience against both overexploitation and uncertainties, and may even prevent resource collapses (Ballantine, 1989; Ballantine, 1995; Bohnsack, 1996; Guénette *et al.*, 1998a). Mistakes in stock assessment would have less impact in the presence of adequate protected areas. In the absence of exploitation, the spawner biomass is likely to increase, improving the reproductive potential, and eventually rebuilding the stocks. The presence of large individuals would also reduce the risk of sex imbalance in protogynous species (Buxton, 1993b).

Increases in density and biomass of various species and especially those targeted by the fishery have been reported in several reserves (Plan Development Team, 1990; Roberts and Polunin, 1991; Dugan and Davis, 1993; Roberts and Polunin, 1993a; Rowley, 1994; Bohnsack, 1996; Guénette *et al.*, 1998a) (see also Appendix 1). It should be noted, however, that the presence of even limited exploitation within the protected area diminishes expected benefits (ICES, 1994; Jennings *et al.*, 1996; Attwood *et al.*, 1997; Wantiez *et al.*, 1997; Goodridge *et al.*, in press). These benefits decrease rapidly after exploitation resumes in previously unfished reserves (Roberts, 1986; Alcala and Russ, 1990; Russell, 1997; Robertson, 1998). Generally, marine reserves have not been shown to swell the fish population in the unprotected parts of the habitat (Roberts and Polunin, 1993a; Schmidt, 1997; Guénette *et al.*, 1998a). However, in some cases reserves have been shown to sustain yield by adult migration into the neighbouring fishing grounds in the Philippines (Alcala and Russ, 1990; Russ *et al.*, 1992; Russ and Alcala, 1996a), South Africa (Bennett and Attwood, 1991) and Spain (Ramos-Espla and McNeill, 1994). In addition indirect evidence coming from modifications in fishers' behaviour should also be considered (Rowley, 1994). Reserves may also be a suitable tool for indirectly reducing by-catch, when it is possible to protect critical habitats of the species or age group at risk. For instance, spatial closures, both temporal and permanent, were successful in cases where juveniles migrate towards adult habitat, such as plaice (ICES, 1994) and red mullet (Garcia and Demetropoulos, 1986; Caddy, 1990). Such reserves would be more efficient than gear modifications, as well as easier to regulate and enforce than single-species oriented regulations, which can often be contradictory.

The effects of fishing on benthic structure and community structure underline the importance of creating permanent reserves. By eliminating mobile gear fishing, the bottom complexity as well as the benthos and fish species composition are likely to change from disturbed to mature ecosystems (Watling and Norse, 1998). Species vulnerable to fishing and perturbations are likely to increase while their prey may decrease. Similarly, long-lived species and those

requiring highly structured habitat would be expected to thrive. However, we do not know if damage done to benthic communities is reversible, and if so, reconstruction could occur through switches of communities (Hall, 1994). In addition, responses of individual species may be dampened through competition (McClanahan and Obura, 1995) or global recruitment conditions (Jourde, 1985; Wantiez *et al.*, 1997). Evidence that closed areas may result in community structure modification have been shown in Kenya (McClanahan and Obura, 1995), California (Engel and Kvitek, 1998), Sicily (Pipitone *et al.* 1996 in Engel and Kvitek, 1998), and Zimbabwe (Sanyanga *et al.*, 1995). Since some epibenthic species are slow-growing and long-lived (around 100 years, Watling and Norse, 1998), rebuilding the habitat structure may be a long process.

Both larval dispersal and migration patterns will define the location, size and number of reserves necessary to protect a particular species (Carr and Reed, 1993; Quinn *et al.*, 1993; Attwood and Bennett, 1995; Allison *et al.*, 1998). The patterns of larval dispersal, the location of their settlement and the presence and contribution of neighbouring populations will be crucial to the efficacy of the reserve and its ability to sustain a population (Quinn *et al.*, 1993; Allison *et al.*, 1998). A few cases convincingly point out the importance of accounting for larval dispersal in sustaining or rebuilding fished patches (Tegner and Dayton, 1977; Tegner, 1992; Tegner, 1993; Rogers-Bennett *et al.*, 1995; Stoner and Ray, 1996; Dye *et al.*, 1997; Orensanz *et al.*, 1997). Although successful, some reserves would not be sufficient to sustain their own population. For example, in the Bahamas, the queen conch is thought to depend on unfished deep waters outside the reserve for a part of its recruitment (Stoner and Ray, 1996). Reserves would also be especially useful when adult density is an important factor for successful reproduction.

A fast rate of adult migration outside the reserve is likely to decrease the efficiency of the reserve since a large proportion of individuals would still be vulnerable to exploitation (Gu nette *et al.*, 1998a). In consequence, the need for knowledge of home range and migration patterns becomes crucial, and this has already been addressed by several authors (Bennett and Attwood, 1993a; Holland *et al.*, 1993; Attwood and Bennett, 1994; Zeller, 1997). When the objective is to control fishing mortality for targeted species, it may be possible to design reserves that would help protect the stock when combined with other management measures. Possible solutions include permanent and/or temporal closures to include critical habitats such as nurseries, spawning and feeding grounds or to protect the stocks during crucial life history events such as migrations and spawning aggregations (Gu nette *et al.*, 1998a). Some closed areas used as part of fishery management regimes (for single species) produced positive results for crabs (Yamasaki and Kuwahara, 1989), shrimps (Roberts, 1986), spiny lobster (Davis and Dodrill, 1989) and plaice (ICES, 1994). In other cases, poor results have been shown when the protected area is located in unfavourable habitats (Heslinga *et al.*, 1984; Tegner, 1993), or is not protecting a sufficient portion of critical habitats (Armstrong *et al.*, 1993; Shepherd and Brown, 1993; Cadrin *et al.*, 1995). In such situations, the establishment of marine reserves could lead to a

false sense of security while antagonising fishers and other stakeholders (Carr and Reed, 1993; Dugan and Davis, 1993).

Based on the minimum spawning biomass that should be preserved in exploited stocks, the Plan Development Team (1990) suggested that 20% of the total habitat be protected. Modelling using species with different life histories suggested that a large proportion of the total habitat (up to 50%) should be included in reserves to efficiently protect both the habitat and the animals contained therein from the negative impacts of extractive use of the resources (Attwood and Bennett, 1995; Man *et al.*, 1995; Holland and Brazee, 1996; Sladek Nowlis and Roberts, 1997; Guénette and Pitcher, 1999). Compared to one single reserve, a network of reserves would increase their buffer function against environmental variation and local catastrophes (Ballantine, 1995; Ballantine, 1997). A network of reserves would also be more suited to species with a low site fidelity or with a poorly understood life history. A good example would be provided by a squid (*Loligo vulgaris reynaudii*), a species that sporadically uses the undisturbed spawning habitats within the Tsitsikamma National Park (South Africa), depending on environmental conditions (Sauer, 1995).

Socio-economic factors

Economic factors are generally not taken into account in the planning of MPAs (Tisdell, 1986), probably because MPAs are usually created either in anticipation of biological and ecological benefits, or in response to public pressure, in particular that from conservation groups. Arguments have been put forward for the inclusion of both social and economic variables in the decision to establish marine reserves (Sumaila, 1998c). Economic justification for establishing marine reserves usually takes two broad forms. First, it is argued that economic benefits may follow the establishment of marine reserves in the form of creating employment through non-consumptive activities such as tourism and recreation. Second, it is expected that marine reserve creation can protect future jobs by increasing the chances of managing the stocks sustainably.

Quantitative modelling for assessing marine reserves

Biological

Single species

Single-species modelling has been useful in showing how marine reserves could help rebuild over-exploited populations by increasing population abundance, survival and the numbers of older individuals, thus serving as a hedge against stochastic recruitment failure (see Guénette *et al.*, 1998a). Equilibrium models were useful to explore the influence of population dynamics and basic mechanisms behind marine reserves, such as the impact on fishing mortality, yield, body size, mean age, and the implications of high exchange rate between protected and unprotected areas (Polacheck, 1990; Die and Watson, 1992; Russ *et al.*, 1992; Daan, 1993; DeMartini, 1993; Watson *et al.*, 1993). The stage-based model built by Crouse *et al.* (1987) has shown how young adult mortality of loggerhead sea turtles (*Caretta caretta*) was more important for the population size and productivity of this species than were its nesting beaches. From this, it appears that the use of Turtle Excluder Devices in trawls (Crouse *et al.*, 1987) and more importantly, the reduction of turtle exploitation in Asia (Poiner *et al.*, 1990) would efficiently decrease the total mortality of adults. Using life tables, Walker (1996) showed how different species of skates are vulnerable to fishing at various degrees of intensity.

The addition of stock-recruitment relationship and reproductive potential led us to consider resilience to exploitation induced by the increase in the number of large spawners in closed areas (Quinn *et al.*, 1993; Man *et al.*, 1995; Holland and Brazee, 1996; Sladek Nowlis and Roberts, 1997; Guénette *et al.*, 1998a; Guénette and Pitcher, 1999; Sladek Nowlis and Roberts, 1999). The balance between stock rebuilding and yield improvement depends on the rate of biomass exchange between protected and unprotected areas. Also, larval dispersal is shown to be a possible mechanism for rebuilding the stock (Quinn *et al.*, 1993; Man *et al.*, 1995; Sladek Nowlis and Roberts, 1997; Sladek Nowlis and Roberts, 1999).

Spatial modelling

Since the marine environment is not homogenous, spatial structure of the species habitat should be included in modelling to help understand the influence of larval dispersal, adult migration and age-specific habitat needs. In addition, explicit spatial models summarise better the interaction between migration and the size and shape of the reserves. To date only a few spatial studies have incorporated marine reserves. Attwood and Bennett (1995) used simple spatial structure to compare three species with different life histories (longevity, reproduction,

migration). They show how migration influences the size of the reserve necessary to rebuild the population. Quinn *et al.* (1993) built a two-dimensional model to evaluate the importance of larval dispersion. He suggested the use of a network of reserves strategically located to insure sufficient larvae production and dispersal. St. Mary *et al.* (1998) included density dependent mortality in a two-stage model where adults and juveniles occupied different habitats. They found that density dependence and site fidelity were crucial factors in determining which stage should be protected.

Since fishing effort is spatially and temporally variable, spatial dynamics of fish distribution and fishing effort should also be included if the goal is to limit fishing mortality and compare benefits emerging from different management strategies. Rijnsdorp (1995) used a spatially-explicit model that takes into account the distribution of plaice (by age group) and of fishing effort and quantity of discards, both by season and area in the North Sea. The movement of fish was based on historical data obtained from surveys and commercial CPUE. They assumed, and rightly so (ICES, 1994), that fishing effort would redistribute around the boundaries of closed areas. The authors concluded that a closed area located to protect undersized juveniles would be beneficial for plaice populations. Guénette *et al.* (1998b) used an age and spatially-structured model that included explicit migration of northern cod, contraction of geographic distribution with decrease in abundance (Atkinson *et al.*, 1997), and fishing with three gears. The results show that marine reserves by themselves may not be sufficient to control fishing mortality of a migrating species subjected to extreme fishing effort. Only very large marine reserves (60-80%) covering the aggregation areas would help rebuild the stock. In addition the model underlines the competition between fishing gears that compels the use of additional management measures to control effort. Hutchings (1995) suggested closing the winter offshore fishery for cod in Newfoundland where trawlers are targeting the pre-spawning aggregations, thus returning to a pre-industrial spatial and temporal refuge for this species. However, this closure should be accompanied with some sort of effort control for the inshore fleet which has increased tremendously since the 1950s and especially during the 1980s.

Ecosystem modelling

The recognition that the exploited stocks are parts of ecosystems and the fact that there are usually natural interactions between species, such as predator-prey relations, has compelled fisheries scientists to come to the conclusion that models that aim to contribute to the sustainable management of marine resources must take the ecosystem approach. Hence, in recent times, we have seen the development of several generic approaches to multi-species and ecosystem analysis. One can identify at least four different approaches to ecosystem management in the literature (Walters *et al.*, 1997): (i) multi-species virtual population analysis (see Sparre, 1991; Tjemeland and Bogstad, 1998), (ii) differential equation models for biomass dynamics (see Larkin and Gazey, 1982), (iii) bio-energetic models (see Stewart *et al.*, 1981; Kitchell *et al.*, 1996), and (iv)

the ecosystem model known as Ecopath, proposed by Polovina (1984), and vigorously pursued recently by Christensen and Pauly (1995).

All four approaches mentioned above appear to have the potential of being extended to allow for the analysis of the effect of establishing marine reserves. It would be useful to undertake such extensions so as to provide plausible alternative approaches for comparisons purposes. For example, the multi-species virtual population model published by Tjemeland and Bogstad (1998) for the Barents Sea could be extended to assess the possible impact of marine reserves. The model is spatially structured and includes sea temperature, feeding related growth, migrations and trophic interactions between cod, capelin, herring, harp seal and minke whale. At this stage, only the Ecopath family of models includes non-exploited species, and allows for the assessment of the impact of marine reserves on ecosystem management. The Ecopath family includes Ecosim, a dynamic version of Ecopath (Walters *et al.*, 1997), and Ecospace, a spatial version of Ecopath/Ecosim (Walters *et al.*, 1998). From now on this section will describe how the Ecopath/Ecosim/Ecospace frameworks have been adopted to incorporate marine reserves.

Ecocosim and the quasi-spatial modelling framework

In the Ecopath/Ecosim approach, ecosystem components are first arranged in a number of functional groups, for each of which the production, consumption and diet is quantified and a possible model with a set of mutually compatible trophic fluxes is constructed (Pauly, 1996; Walters *et al.*, 1997). Ecopath is a static modelling approach, Ecosim, however, introduces dynamism into the static model by (i) using Ecopath to estimate parameters based on the assumption of mass-balance; (ii) representing biomass and size structure dynamics by a mix of differential and difference equations, (iii) using variable speed splitting to model the dynamics of both 'fast' (e.g. plankton) and 'slow' (e.g. top predators) groups, and (iv) representing micro-scale behaviour by allowing differentiation between top-down, intermediate and bottom-up control of predation.

Ecosim is then extended to provide a means of describing the spatial relations of biomass and fishing mortalities which are required to examine the potential impacts of marine protected areas. To do this, Watson and Walters (1998) devised a simple modification to Ecosim which allows the biomass of Ecopath groups to be partitioned into two portions with exchange processes operating between them (Figure 1). One biomass portion is then assumed to be within a marine reserve and subject to different levels of fishing mortality (at least for some groups) than the other portion. If the biomass of Ecopath groups is assumed to be uniformly distributed in space then the proportion of the biomass assumed to be in the marine reserve is also the proportion of the area of the marine system described by the Ecopath model included in the marine reserve. This assumption allows us to observe the impact of marine reserve 'size' and biomass exchange rates on the calculated biomasses and catches of Ecopath groups.

The Ecospace modelling approach

Ecospace is a spatially explicit model for policy evaluation which allows consideration of the impact of marine reserves in an ecosystem (i.e. trophic) context, and which relies on the Ecopath mass-balance approach for most of its parameterisation (Walters *et al.*, 1998). Additional inputs to the framework are movement rates, used to compute exchanges between grid cells, the settings (top-down vs. bottom up control) also required for Ecosim, the dynamic simulation routine derived from the system of linear equations in Ecopath, and habitat preferences for each of the functional groups included in the model.

Convergence from the homogenous distribution assumed in the Ecopath base model to highly patterned distributions, simultaneously accounting for the habitat preferences and food requirements of predators and prey, the distribution of fishing effort (driven by local abundances and fishing costs) and the existence of marine reserves, is extremely rapid, due to an integration scheme with different step sizes for the 'fast' and 'slow' groups, allowing the former to track the population changes of the latter.

Like all modelling approaches, the Ecopath/Ecosim/Ecospace frameworks are processes in successive approximation. The methods are in continuous development. Clearly, they face a number of limitations; for example, they cannot capture sudden changes of ecosystem state, there is no explicit consideration of seasonal changes or directed migration, and there is a lack of adequate knowledge of the interactions between different species and their habitats. Despite these and other limitations, the outward simplicity of these frameworks, and the information-rich graphs they generate, coupled with the increasingly global availability of the required Ecopath files, should ensure a wide use for this approach, both for generating hypotheses about ecosystem function and evaluating policy choices.

Bio-economics

Although the number of MPAs has increased significantly within the last two decades, the number of applications of economic analyses to MPAs are few (Hoagland *et al.*, 1995), partly because of the difficulty in performing such analyses. In their review of net benefit evaluation for marine reserves, Hoagland *et al.* (1995) compared 62 studies related to this issue which have been published between 1980 and 1995. The result showed that only about 18 per cent of these provided dollar estimates of benefits and costs based on empirical analysis. Only two studies included both market and non-market values of marine reserves in the estimate of cost and benefits. The difficulties of estimating the economic benefits of marine reserves, as pointed out by Laist *et al.* (1986) in the case of coral reefs, lies for example in the lack of information on species composition and in determining pollution effects. Despite these difficulties, Dixon and Sherman (1990) demonstrated that in many cases 'market' benefits alone can justify the creation of a MPA.

Economic analyses have tended to adhere to the following dichotomy. First there are the cost-benefit types that seek to determine the net economic benefits

that can be expected from the establishment of MPAs, considering the possibility that non-consumptive activities will increase as a result. In this case, methods such as contingent valuation, hedonic pricing and travel cost are commonly used to evaluate the benefits of marine reserves (Dixon, 1993; Sobel, 1993). On the other hand, there are analyses that are bio-economic in nature, which seek to isolate the usefulness of marine reserves as tools to support and enhance sustainable management (Holland and Brazee, 1996; Lauck, 1996; Hannesson, 1998; Sumaila, 1998b).

So far most bio-economic models are based on single species biological models. The main aim of this class of models, which are usually quantitative in nature, is to assess the bio-economic consequences of establishing marine reserves under different assumptions and scenarios. In addition to computing the catch and stock levels (as with biological studies of marine reserves), these models determine the present value of the stream of economic rent. Single species bio-economic models of marine reserves have been published by Holland and Brazee (1996), Sumaila (1998b) and Hannesson (1998). Holland and Brazee developed a multi-age dynamic model for red snapper of the Gulf of Mexico in which the effort is assumed to be fixed. They conclude that reserves will sustain or increase harvests in heavily fished inshore fisheries.

The next two models assume that fishing effort is variable from year to year to insure optimal economic benefits to the fleet. Sumaila (1998b) uses data on the North-east Atlantic cod to determine the bio-economically optimal size for the Barents Sea fishery. This model considers uncertainty in the form of a shock to the system through recruitment failure in the fished area of the habitat. The study found, among other things, that the establishment of a marine reserve in the context of a sole owner fishery is bio-economically beneficial when net exchange rates for cod are reasonably high and reserve sizes are large. Large reserves provide good protection for the stock in the face of the shock, while high transfer rates make the protected fish available for harvesting after the shock has occurred. Hannesson (1998) used a single age hypothetical model to investigate what will happen to fishing outside the reserve, and to the stock size in the entire area as a result of the establishment of a marine reserve. As in Sumaila (1998b), this analysis finds that the conservation effect of a marine reserve depends critically on the size of the reserve and the migration rate of fish.

The only spatially based bio-economic models of marine reserves we are aware of have been published by Sanchirico and Wilen (1998) and Holland (1998). Sanchirico and Wilen (1998) found that the directional flow of biomass could stem from special economic circumstances rather than special biological characteristics. They found that in many cases the industry might benefit from closing areas which are less profitable rather than areas that are biologically unique. Holland (1998) added fishers' choice of fishing grounds based on interviews, to a spatially structured, multi-area and multi-species model. His model demonstrates, among other things, that (i) it is unlikely that area closures will increase fishery profits significantly when effort is already at optimal levels, but they may allow for the maintenance of higher levels of spawning biomass; (ii)

that area closures can impact various groups of fishermen differently, that is, there may be losers and gainers in the fishing community.

Ecosystem based bio-economic models of marine reserves are rare. The only one we are aware of is the extension to the quasi spatial modelling framework described earlier. The catches determined using this framework are evaluated to find the economic benefits that are achievable for different sizes of marine reserves (Pitcher *et al.*, 1998; Sumaila, 1998a).

The way forward

Difficulties of creating marine protected areas

The difficulties in establishing MPAs are a common feature irrespective of the country - developing or developed, tropical or temperate. Establishing MPAs in Canada, for instances, has been a long and controversial process. There has been no lack of initiatives, yet by 1998 only two marine conservation parks had been established (National Research Council, 1997) within the 28 marine regions of Canada (Lien and Graham, 1985). In Florida (USA) the establishment of the Florida Keys Marine Sanctuary was delayed for several years as issues between state and federal authorities were negotiated. This delay also intensified the conflicts between fishers, managers and conservationists (National Research Council, 1997). Similar sentiments have been expressed by Western Australian rock lobster fishers in relation to the establishment of a marine park north of the Perth Metropolitan Area.

Notwithstanding the progress that has been made so far, we still face difficulties and challenges with respect to adequately understanding the ecological basis of ecosystem protection. How do we deal with the lack of knowledge on fish behaviour, migration patterns, larval dispersal? The lack of satisfactory answers to these questions makes it difficult to both select the best areas, and to determine appropriate sizes of reserves. In addition, our knowledge of sources and sinks habitats is tainted since most of these are already overexploited and because inter-annual environmental variations may introduce additional uncertainties. We cannot wait to have all the answers to undertake actions (Ballantine, 1991; Ludwig *et al.*, 1993; Pollard, 1993). Furthermore, no single habitat will act as a primary habitat for all species (Roberts, 1998). The most reasonable choice would be to close a wide variety of common and ordinary habitats, along with exceptional sites, which together would likely cover a large variety of organisms (Ballantine, 1991; Russ and Alcala, 1994). As our knowledge increases, refinements and additions will improve the network (Ballantine, 1991; Ballantine, 1995; Roberts, 1997; Gu nette *et al.*, 1998a).

Because we do not generally have good baseline data (Pauly, 1995), we do not know the real extent of impacts of fishing on ecosystems. However, we have indications that fishing may be detrimental to the ecosystems and that we are rapidly mining the sea bounty. We do not know how ecosystems rebuild and how much time it would take. Permanently closing an area would help us understand the process. We already have evidence that closing an area to fishing has led to modifications in coastal ecosystems, and that under certain conditions, marine reserves can help adjacent fisheries by increasing target populations. We also know that several long-lived species cannot survive under frequent fishing disturbance, and those which did survive, did so because the habitat was inaccessible to fishers (Beverton *et al.*, 1984; Walters, 1998). In the same way,

reserves could work by rendering a significant proportion of the habitat inaccessible (Shackell and Lien, 1995).

Another major issue in the establishment of marine reserves as in any resource allocation exercise, is the "conflicting interest problem". First, the establishment of reserves will likely stir competition between fishers using different gears. Second, areas set aside for conservation versus access to resource exploitation, recreational fishing versus commercial fishing (Kay and Alder, 1999), conservationists versus urban and tourism developers (Tisdell, 1986), and economic objectives versus socio-cultural considerations, are all common conflicts. For this reason, resource use analysis is needed for zoning and management planning of MPAs (Rigney, 1990). This includes a definition of the uses, their value and socio-economic importance, and a measure of their intensity and effects. Related to the foregoing are issues pertaining to equity and existing use rights. In marine reserves where all consumptive uses are prohibited, reserves could represent fairness and equity. However, in the case of multiple-use MPAs, although all uses may be non-consumptive, groups denied access to commercially valuable resources by zoning or exclusion could consider management by MPAs to be largely unfair. Thus, it is important to recognise that political pressure and opposition from interest groups is a common theme in the establishment of MPAs, even with general public acceptance (Tisdell and Broadus, 1989; Plan Development Team, 1990; Fiske, 1992). As a consequence, they might want to use political campaign for changes in boundaries, regulations, etc., in order to obtain more benefits (or to avoid greater loss) from the management plan.

In economic terms, stakeholders are willing to embrace the marine protected area concept if it is at worst economically neutral and their potential to increase their economic gains are not unduly constrained. A good example of this attitude is the rock lobster fishers on the Central Coast of Western Australia, who are willing to accept the declaration of a MPA in their fishing areas subject to guarantees on their access to the resource now and in the future. Similarly, but at a larger scale, the development of Australia's Oceans Policy, which is based on an ecosystem approach, including a representative system of MPAs, has been controversial because many stakeholders are concerned with their future access rights. In sum, it is helpful to consider the benefits of MPAs in terms of the trade-offs between long-term protection of rich ecological resources and the more immediate use of resources for economic gain. These trade-offs are in no way easy to administer, as they involve uncertainty associated with the ecological benefits of MPAs, non-monetary values that people put on resources, intra- and inter-generational equity considerations, and the socio-cultural preferences of local communities. As stated by Dixon *et al.* (1993), in some instance, it may be more important to consider a balanced use of natural resources for both economic and ecological functions than to strictly preserve the resources in the area.

Keys to success

There are calls by some scientists that the currently low area of ocean surface that is reserved should be boosted to 20% by the year 2020. The big question is what

can scientists do to help achieve this? It is our view that the answer to this question probably lies in the following statement by Tony Pitcher of the Fisheries Centre (University of British Columbia, Vancouver), quoted in Schmidt (1997): "The country that has the courage to set up no-take areas now is the country that will have a thriving fishing industry in 20 to 30 years". We believe the role of science is to help countries come to terms with this statement by undertaking interdisciplinary research to show that this quotation is indeed true for most countries.

To help the process, apart from addressing some difficult issues in relation to the creation of marine protected areas, we need to consider those factors that may contribute to their success. First, at the same time that we are trying to learn more about the ecological and socioeconomic impacts of MPAs, we have to acknowledge that establishing MPAs is like any other public policy decision. That is, it is a political process where scientific knowledge may inform the debate and influence the outcome, but it will not make the decision (Sobel, 1996). Following Ludwig *et al.* (1993) who suggest that we should not wait for scientific consensus before we react in such situations, we should now create marine reserves as an insurance against management failure and as a common sense precaution. It is more effective and has more general appeal to the public to consider establishing MPAs because it is the "right thing to do" (Ballantine, 1995). Although this may seem as if the MPAs are being established on an opportunistic basis (Brunckhorst and Bridgewater, 1995; Hockey and Branch, 1997), it may still be successful given that the objectives are precise.

Secondly, from the social, organisational, regulation, control and implementation points of view, we need to better understand fishing patterns and fishers' reactions to marine reserves. To enhance the potential for establishing and thus reaping the expected benefits of such marine reserves, fishers must be involved early in the decision-making process (Fiske, 1992; Alder *et al.*, 1994; Neis, 1995; Vincent and Pajaro, 1997). This is important because fishers possess detailed knowledge of their fishing grounds (Neis, 1995; Younger *et al.*, 1996; Fischer *et al.*, 1997a; Neis *et al.*, in press) which could be used to define acceptable and efficient reserves. In addition, fishers' reactions to the establishment of area closures, temporal or spatial, should also be taken into account. The "plaice box" in the North Sea is a good example showing the importance of this consideration. Although fishing effort had decreased following the exclusion of big trawlers, small boats increased their total effort within the box (ICES, 1994; Piet and Rijnsdorp, 1996). At the same time, the trawling activity concentrated along the borders of the closed area (Rijnsdorp *et al.*, 1996). As involving fishers implies that part of the fisheries management function is controlled locally, scientists and policy makers need to improve their communication with fishers, to eliminate mutual distrusts and to truly share responsibilities.

Thirdly, well designed monitoring programs will be necessary to assess benefits, to increase knowledge of both fishers and scientists and to improve the level of protection. We need to gather data about the pathways of population and ecosystem rebuilding. Fished and unfished areas should be compared to assess the

extent of the rehabilitation. Catch statistics in relation to the reserve should be gathered to detect yield improvement. The monitoring programs should be planned on a long-term basis as many processes will take a long time and the more impacted species are long-lived. Keeping track of fishers' behaviour and fishing power will also be essential to maintain the protection conferred by reserves.

Finally, it has been widely recognised that public participation and local community involvement is an essential factor contributing to the success of establishment and implementation of MPAs (Kaza, 1988; Tisdell and Broadus, 1989; Plan Development Team, 1990; Rigney, 1990; Fiske, 1992; Wolfenden *et al.*, 1994; Walters and Butler, 1995; Gilman, 1997). Several examples show that involvement of stakeholders and local communities contribute to the successful implementation of MPAs (see Appendix 1). In the absence of strong community support, the integrity of MPAs relies more on efficient enforcement. The community itself could also initiate the process. For instance, Bonavista Bay, a small coastal community in Newfoundland Canada, is formulating its own local management measures using no-take marine reserves to maintain lobster stocks. The fishers have been encouraged and assisted by Parks Canada, the Department of Fisheries and Oceans, a local University (Memorial) and the Fishermen's Union (Lien, 1998). This "bottom up" initiative is from stakeholders who have recognised the need to pro-actively manage their own resources. Fortunately, the management proposals here have not conflicted with national regulations. The outcome of this initiative may have been less successful if such conflicts had arisen. Involving the public also means taking into account the social, cultural and political importance of the communities. The marine sanctuary in Fagatele Bay, American Samoa, is a good example showing that successful implementation depends largely on acknowledging these issues (Fiske, 1992). Good examples can also be found in the Philippines where, apart from facilitating implementation, community involvement yielded positive ecological and socio-economic benefits (White, 1988; Christie *et al.*, 1994). It should be noted, however, that co-management and community involvement require a great deal of commitment and energy from all parties. As reported by several authors, despite its potential benefits and success, community involvement is not without difficulties and pitfalls (McCay, 1988; Attwood *et al.*, 1997; Cocklin *et al.*, 1998; Goodridge *et al.*, in press).

New directions

Acknowledging our limitations in understanding the ecosystem, one might try to use a precautionary approach in creating a network of marine reserves. At this point we should not aim at sustaining the present state of ecosystem health (or misery?) but to rebuild ecosystems (Pitcher and Pauly, 1998). Rather than articulate our strategies around exploited species, we should also try to protect non-exploited species. MPAs should also be used, in combination with other management measures, as part of an adaptive management scheme. Rather than solely controlling fishing mortality for targeted species, reserves should be designed to allow permanent and/or temporal closures to cover critical habitats

such as nurseries, spawning and feeding grounds or to protect the stocks during crucial life history events such as migrations and spawning aggregations. MPAs should be seen as tools to learn and experiment with target species recovery, ecosystem management and co-management.

Research should also be directed towards the evaluation of marine reserves to determine their success and potential benefits. Up to now, bio-economic analysis of the potential benefits of such reserves have concentrated on their impacts on the conservation of species in the protected area, and of the potential effects of spill over to the unprotected areas on catch levels and discounted economic rent (Holland and Brazee, 1996; Hannesson, 1998; Sumaila, 1998b). Clearly, protecting the marine habitat is bound to lead to higher productivity in the future, which at the next level will benefit catches and economic gains. Capturing these types of benefits of marine reserves in the next generation of bio-economic models will be crucial. Future bio-economic models will have to incorporate the fact that in most cases habitat loss or disturbance results in decline of species of commercial value with time. Another important contribution that can come from economic modelling is designing incentive regimes that will ease the regulation and control functions, and reduce poaching in protected areas. In addition to bio-economic models, an objective-based assessment model could be used to evaluate the success of marine reserve. For example, a scoring system called COMPARE (Criteria and Objectives for Marine Protected Area Evaluation), developed by Hockey and Branch (1997), can be used to measure the effectiveness of MPAs, in terms of their scientific, socio-economic and legal performance.

Planning and implementation of MPAs could benefit from a use of conflict resolution tools, such as Decision Support Systems (Bruce and Burt, 1997) and ecological models (Watson *et al.*, submitted). These models are initiated to answer questions regarding the ecological and economic impacts of management decisions. Sophisticated decision support systems may incorporate ecological models to enable managers to test "what if" scenarios and therefore refine decisions or provide a range of decisions. Current decision support systems (DSS) show potential but inherent problems such as a lack of information required to test and run the decision support system, the level of detail and quality of the data, and the high level of uncertainty of the models which constrain their use. These decision systems are applicable to only a few situations where there is sufficient information, or resources to gather the information or where the issues are few and simple. As ecosystem models are further developed and the interface of geographic information systems and DSS improves, the application of these methods to establish MPAs will widen.

Finally, one must not forget that fishery resources are not separate from the rest of the ecosystem, and thus an ecosystem-based approach must be taken as a new direction for establishment and management of MPAs. We ought to recognise that threats and damage to MPAs come also from the adjacent land, such as siltation, sewage, coastal pollution, river run-off, etc. (Rowley, 1994), and that MPAs are part of the interface between coastal and marine areas. MPAs alone may not guarantee the long-term persistence of the targeted species, as catastrophic events, pollution and climatic changes may impact the habitat and its

biota in an uncontrollable manner (Lawton, 1997; Allison *et al.*, 1998). Uncontrolled development and tourism activities may continue to alter the habitat as much as fishing (Carter *et al.*, 1994), and thus undermine the effects of MPAs. In other words, the management and objectives of MPAs must be closely linked with and support overall planning for the coastal zone. This is, of course, not an easy task, as coastal zone management is a rather complicated issue, considering that various sub-systems constituting a coastal area are interdependent. However, one could get away from attempting to quantitatively measure bio-physical changes that take place in coastal areas as a result of certain activities or events. Rather, an index could be developed to provide relative measures of the importance of such changes. A damage schedule approach presented in Chuenpagdee (1998) is one example of such an index.

Acknowledgements

We thank Daniel Pauly and Tony Pitcher for commenting on an earlier draft of the manuscript, and George Branch and Carlo Moreno for providing references. We also thank Michael Sinclair for inviting us to write the paper and for arranging the funding through the Canadian International Development Agency (CIDA) for the first two authors to attend and present the paper at the ICES/SCOR Symposium on the Ecosystem Effects of Fishing meeting in Montpellier. Ussif Rashid Sumaila acknowledges the kind support of the Research Council of Norway's Marine Resource Management Program. Sylvie Gu enette was supported by the University of British Columbia Grant Fund.

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Appendix 1

Examples of MPAs including the type of habitat, type of protection, level of community involvement and results.
 (Result: good = evidence of recovery for species; medium = problem with control and/or recovery not complete; poor = no sign of recovery)

Country/ Reserve	Habitat	Type of protection	Community involvement	Result	Reference
Australia The Great Barrier Reef Marine Park	Coral reefs	Three zones: A. preservation and scientific research B. National Park (educational, scientific and recreational use) C. general use (limited commercial and recreational fishing) Only 5% closed to fishing	- Involved in the planning phase, and management plan - Include diverse interest groups	1. Good 2. Difficult balance between conservation and users demand	(Craik, 1981; Tisdell and Broadus, 1989; Rigney, 1990; Shorthouse, 1990; Cooks, 1994; Rowley, 1994; Ferreira and Russ, 1995; Beaumont, 1997)

New Zealand Leigh Marine Reserve	Rocky reefs, kelp forests	No fishing or collecting, no disturbance	<ul style="list-style-type: none"> - People not involved at the beginning except large campaign of information - Management by committee composed of representatives from the University, local county, Underwater Association, Fisheries Management Division 	Good	(McCormick and Choat, 1987; Cole <i>et al.</i> , 1990; Ballantine, 1991; Cocklin <i>et al.</i> , 1998)
Seychelles Cousin Island Nature Reserve	Coral reefs	No fishing, no habitat disturbance	<ul style="list-style-type: none"> - Management by Bird International Enforcement effective 	Good	(Jennings <i>et al.</i> , 1996)
Sainte-Anne, Baie Ternay, and Curieuse Marine National Parks	Coral reefs	No fishing, no habitat disturbance Consumptive fishing allowed in Sainte-Anne	<ul style="list-style-type: none"> - Managed by the Government - Enforcement more or less efficient 	Poor (poaching is a problem)	(Jennings <i>et al.</i> , 1996)

New Caledonia Southeast Lagoon	Islands and their coral reefs	Close to fishing and collecting Islands not equally protected.	N/A	Medium Islands with the best protection have the highest benefits	(Jourde, 1985; Wantiez <i>et al.</i> , 1995; Wantiez <i>et al.</i> , 1997)
Ile de Mayotte (Indian Ocean) Langogori Marine Reserve	Coral reefs	Closed to fishing	N/A	Good	(Letourneur, 1996)
Republic of Palau Reserves for Trochus shells	Coral reefs	No fishing areas implemented along with seasonal closure, size limit and total closures in some years	N/A	Medium Some closed areas were not good habitats for Trochus	(Heslinga <i>et al.</i> , 1984)
Philippines Sumilon Island	Coral reefs	Closed to fishing Displaced 100 fishers	Education of the community Administration and enforcement by Silliman University	Good reserve respected but violations occurred in the 1980s because of resentment and change of political leaders	(Russ, 1989; Alcalá and Russ, 1990; Russ <i>et al.</i> , 1992; Russ and Alcalá, 1994; Russ and Alcalá, 1996b; Russ and Alcalá, 1996a)
Apo, Pamilacan, Balicasag Reserves	Coral reefs	Two zones: sanctuary (closed to fishing) and restricted fishing zone (non-destructive methods only)	Community-based management	Good	(Savina and White, 1986; White, 1988; Russ and Alcalá, 1996b; Russ and Alcalá, 1996a)

Handumon Reserve	Coral reefs	Two zones: sanctuary (closed to fishing) and restricted fishing zone (non-destructive methods only)	Community-based management	Good	(Vincent and Pajaro, 1997)
San Salvador Island Reserve	Coral reefs	Two zones: sanctuary (closed to fishing) and restricted fishing zone (non-destructive methods only)	Community-based management	Good	(Christie <i>et al.</i> , 1994; McManus, 1996)
South Africa de Hoop Reserve	Rocky and sandy shoreline	Closed to all activities except research Little enforcement; poaching frequent	Little input from users except public hearings and written inputs	Good Pressure to re-open the fishery	(Bennett and Attwood, 1991; Bennett and Attwood, 1993a; Bennett and Attwood, 1993b; Attwood <i>et al.</i> , 1997)
Tsitsikamma National Park	Rocky and sandy shoreline	Closed to fishing except 3 km stretch where shoreline fishing is permitted	- Very little input from users except public hearings and written inputs - Managed by national government	Good Poaching frequent	(Buxton and Smale, 1989; Buxton, 1993a; Sauer, 1995; Attwood <i>et al.</i> , 1997)
Kenya Kisite National Park (MNP)	Coral reefs	Closed to fishing	Managed by Kenya Wildlife	Good	(Watson and Ormond, 1994; McClanahan and Obura, 1995; Watson <i>et al.</i> , 1996)

Mpungiti Reserve (adjacent to Kisite MNP)	Coral reefs	Traditional fishing only	Managed by Kenya Wildlife	Medium density commercial species than Kisite MNP	(Watson <i>et al.</i> , 1996)
Mombasa Marine National Park	Coral reefs and lagoon	No fishing Contiguous to a restricted fishing zone (traditional gears only)	- Declared and managed by government - No community involvement - Fishers displaced by the reserve N/A	Good	(McClanahan and Obura, 1995; McClanahan and Kaundra-Arara, 1996)
Zimbabwe Lake Kariba reserves	Closed areas created in 1972 Lake created in 1959 by damming Zambezi River	Closed to fishing Implemented along with other management measures (gillnets number and mesh-size restrictions) outside the reserve	N/A	Good Because of high fishing pressure, fishers argue for opening reserves	(Sanyanga <i>et al.</i> , 1995)
Egypt Ras Mohammed Marine Park	Coral reefs	Closed to fishing	N/A	Good for size structure, medium for biomass due to low fishing pressure outside the reserve	(Roberts and Polunin, 1993a; Roberts and Polunin, 1993b)

<p>France Scandola Natural Reserve</p>	<p>Rocky and steep shoreline</p>	<p>Two zones: sanctuary (closed to fishing and scuba diving) and fishing zone (commercial fishing permitted)</p>	<p>Management organization local, includes regional and national government</p>	<p>Good in rocky reefs; undetectable effect in seagrass High disturbance in the fishing zone by boat anchors and fishing</p>	<p>(Leenhardt, 1990; Francour, 1994)</p>
<p>Carry-le-Rouet</p>	<p>Rocky bottom with mosaic of sandy, rock and seagrass patches</p>	<p>No fishing, scuba diving or anchoring</p>	<p>N/A</p>	<p>Good</p>	<p>(Harmelin <i>et al.</i>, 1995)</p>
<p>Spain Tabarca Island Reserve</p>	<p>Rocky reefs, seagrass beds, and several islets</p>	<p>Three zones: A. sanctuary (no activity except research), B. restricted zone (controlled scuba diving and selective commercial fishing), C. peripheral (selective commercial fishing, sport fishing, swimming, controlled scuba diving, vessel moorings in marked section)</p>	<p>- Management by an organization of representatives of all levels of government (national, regional, municipal) - Other stakeholders may have representatives but no decision-making powers</p>	<p>Good</p>	<p>(Ramos-Espla and Bayle-Sempere, 1989; Ramos-Espla and McNeill, 1994)</p>

Medes Islands Reserve	Islets, rocky reefs, algae beds	restricted fishing	N/A	Good	(Ramos-Espla and Bayle-Sempere, 1989; Garcia-Rubies and Zabala, 1990; Ramos-Espla and McNeill, 1994)
St. Lucia, West Indies Maria Island Marine Reserve	Reefs	No fishing	<ul style="list-style-type: none"> - Community involved in planning and decision making - Fishers informally enforce the boundaries although they may not personally agree with it 	Good	(Smith and Berkes, 1991)

Soufrière Marine Management Area	Coral reefs	Multiple use zoning: reserves (no fishing), fishing priority areas, Yacht Mooring Areas	Commercial fishers, operators, NGOs and government took part in the planning process but artisanal fishers not adequately represented - Co-management under Soufrière Foundation including resource user groups, NGOs, government agencies	Poor (artisanal fishers continue fishing in reserves because these are their only fishing grounds)	(Goodridge <i>et al.</i> , in press)
Barbados Barbados Marine Reserve	Fringing reefs, hard-bottom and sandy patches	No fishing except cast-netting for clupeids	Managed by government	Good	(Rakitin and Kramer, 1996)
Bahamas Exuma Cays	Shallow platform covered with sand and seagrass	No collection or fishing Enforcement by one warden	N/A	Good Success of the reserve for queen conch also due to un-fished areas outside the reserve	(Stoner and Ray, 1996)
Netherlands Antilles Saba	Mixture of coral reefs and gorgonians	No fishing in 15% of the Park Enforcement efficient	N/A	Good	(Polunin and Roberts, 1993; Roberts, 1995b)

Belize HolChan Marine Reserve	Coral Reefs	3 zones: A. No fishing, no collecting, no anchoring except in provided mooring, regulated and controlled diving B. local fishers only, no trawling, netting spear fishing; sports such as water skiing and sailing permitted C. fishing under license, no mangrove clearing	Community totally involved in planning and management	Good	(Polunin and Roberts, 1993; Carter <i>et al.</i> , 1994)
USA Looe National Marine Sanctuary	Coral reefs	Ban on coral collecting and damage. No spear fishing, use of fish and lobster traps, live collection of small tropical fish, and other damaging activities	Community not involved but high compliance because of good information program	Good for several target fish, insufficient for lobster (migrate outside the reserve at night)	(Clark <i>et al.</i> , 1989; Rowley, 1994)
Everglades National Park	Coastal and estuarine habitats	Only recreational harvest by net, trap and line fishing permitted	N/A	Medium	(Davis, 1981; Davis and Dodrill, 1989)
Tortugas Shrimp Sanctuary	Shrimp nursery grounds	Closed to shrimp trawling	Community not involved	Poor (poor compliance)	(Gitschlag, 1986; Klima <i>et al.</i> , 1986a; Klima <i>et al.</i> , 1986b; Roberts, 1986)

Chile Mehuín and Las Cruces Marine Reserves	Rocky shoreline	No collecting	N/A	Good	(Branch Moreno, Davis, 1995) and 1994;
Venezuela Archipiélago de Los Roques National Park	Reefs, islands, sandbanks lagoons,	Queen conch fishing prohibited in some areas	N/A	Good	(Weil and Laughlin, 1984; Silva and Desilvestre, 1986)

Figure 1

Exchange of biomass between reserve and non-reserve biomass portions. Parameter X is a user-supplied migration value and P is the proportion of the Ecopath group's biomass within the marine reserve (redrawn from Watson and Walters, 1998).

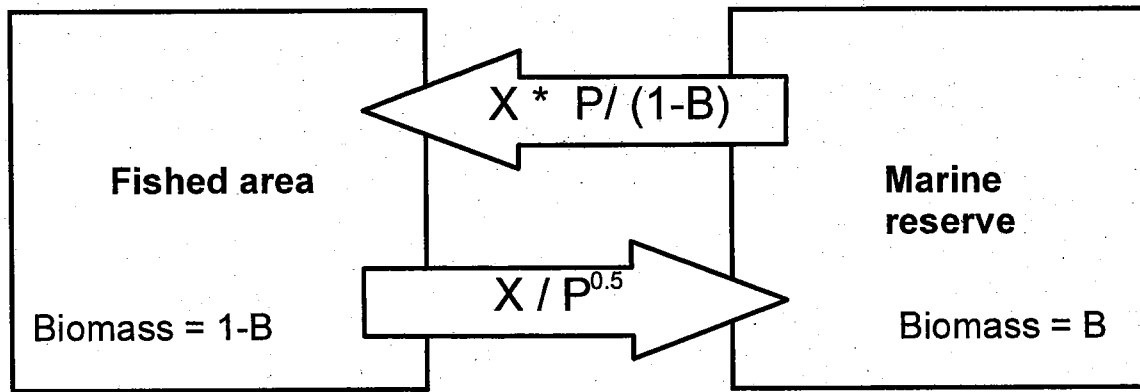


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