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Ecological Economics 31 (1999) 287–304

ECOLOGICAL ECONOMICS

www.elsevier.com/locate/ecolecon

Limiting abuse: marine protected areas, a limited solution

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Abstract

Designation of marine protected areas (MPAs) is increasing as humans seek to combat overexploitation of marine resources and preserve the integrity of the ocean's unique biodiversity. At present there are over 1300 MPAs. The primary legal responsibility for the designation of MPAs falls to individual countries, but protection of the marine environment at large scales is also critical because ocean circulation does not honor legal boundaries and often exceeds the influence of any one nation or group of nations. There are many reasons for establishing MPAs; the papers we surveyed principally referred to scientific, economic, cultural, and ethical factors. Two approaches predominated: fisheries management and habitat protection. Although the major threat to terrestrial systems is habitat loss, the major threats to the world's oceans are fisheries overexploitation, coastal development, and chemical and biological pollution. MPAs may provide conservation of formerly exploited species as well as benefits to the fishery through leakage of 'surplus' adults (spillover) and larvae (larval replenishment) across reserve boundaries. Higher order effects, such as changes in species richness or changes in community structure and function, have only been superficially explored. Because many MPAs are along coastlines, within shipping lanes, and near human centers of activity, the chance of chemical and biological pollution is high. Use of MPAs to combat development and pollution is not appropriate, because MPAs do not have functional boundaries. The ocean is a living matrix carrying organisms as well as particles and therefore even relatively environmentally sensitive uses of coastal ecosystems can degrade ecosystem structure and function via increasing service demands (e.g. nutrient and toxics transformation) and visitation. Whether an MPA is effective is a function of the initial objectives, the level of enforcement, and its design. Single reserves need to be large and networked to accommodate bio-physical patterns of larval dispersal and recruitment. Some authors have suggested that reserve size needs to be extremely large — 50–90% of total habitat — to hedge against the uncertainties of overexploitation. On a local scale, marine protected areas can be effective conservation tools. On a global scale, MPAs can only be effective if they are substantively representative of all biogeographic zones, single reserves are networked within biogeographic zones, and the total amount of area reserved per zone is 20% or greater. The current size and placement of protected areas falls far short of comprehensive or even adequate conservation objectives.

Keywords: MPA; Network; Reserve design; Fishing; Pollution; Species introduction

1. Introduction

There is increasing recognition of the profound effect humans can have on marine systems, lead-

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0921-8009/99/$ - see front matter
PII: S0921-8009(99)00085-3

ing to a multiplicity of efforts aimed at slowing the degradation of the oceans (Weber, 1993; Norse, 1993; National Academy of Sciences, 1994). Because of the difficulties in modifying human actions directly, physical solutions to resource management problems are often adopted. For instance, rather than change natural resource management from open to limited access systems, a park or reserve might be designated to protect a percentage of the habitat necessary for survival of the exploited species. To date there are more than 8000 legally established protected areas covering over 750 million hectares — 1.5% of the earth's surface or $\sim$5.1% of the national land area (World Resources Institute, 1992). Many have been protected for decades and some have enjoyed protection for centuries. By contrast, marine protected areas are relatively recent developments, perhaps because of our belated realization that the ocean, like the land, can be degraded (Kenchington, 1990) and the fact that Western civilization did not regard marine systems as ownable until recently.

Within the last few decades, marine reserves have become a highly advocated form of marine conservation and management. By 1970, 118 marine protected areas had been established in 27 nations (Kelleher and Kenchington, 1992). By 1980, an additional 201 marine protected areas (MPAs) had been designated (Silva et al., 1986). The total currently stands at over 1300 (Kelleher et al., 1995). The Declaration of the IVth World Congress on National Parks and Protected Areas recommended that 10% of each biome in the world be included in protected areas (Kelleher et al., 1995) and 20% of the coastal zone should be in protected areas under agreed management plans by 2000 (Jones, 1994). The second Conference of the Parties to the Convention on Biological Diversity (Jakarta Mandate) included five action items, one of which was establishing and maintaining marine protected areas (de Fontaubert et al., 1996). Despite increasing numbers, MPAs account for less than 1% of the world's marine area and coverage is not even across biological or political regions. For instance, 267 MPAs have been declared in Australia alone (McNeill, 1994), while only four countries in sub-Saharan Africa have designated marine reserves (Hockey and Branch, 1994). Of the 150 marine biogeographic zones identified by Kelleher et al. (1995), over 20% lack any type of protected area designation.

In this paper we review how reserves are defined and who designates and controls them, and the reasons and values underlying reserve designation. We also examine the factors governing reserve design in marine systems. Finally, we explore the usefulness of reserves as a primary tool for marine conservation relative to the four main threats facing the world's oceans: overexploitation of biological resources, development of coastal areas, and chemical and biological pollution. The message is not hopeful. Although marine reserves can be useful in protecting habitat-specific species from overexploitation, the current size and placement of protected areas falls far short of comprehensive or even adequate conservation objectives. Highly mobile species are virtually unprotectable via reserves. In addition, several forms of human impact (e.g. species introductions) are not deterred by reserve designation. Unless serious political will is focussed on designating and enforcing well designed networks of reserve components across all marine biogeographic areas, the predominant uses of marine reserves will be limited to education, research, and political good will.

2. Definitions and designation authority

The IUCN defines MPAs as 'any area of intertidal or subtidal terrain, together with its overlying water and associated flora, fauna, historical, and cultural features, which has been reserved by law or other effective means to protect part or all of the enclosed environment' (Kelleher and Kenchington, 1992). Silva et al. (1986) list 91 categories of protected areas. In fact, MPA is a catch-all term including a range of protection from totally off limits to all forms of use (e.g. Leigh Marine Reserve, New Zealand) to restrictions of select users/multiple use (e.g. Great Barrier Reef Marine Park) to few restrictions (e.g. National Marine Sanctuaries in the United States).

The primary legal responsibility for the designation of MPAs falls to individual countries enacting specific legislation which governs use of particular areas. The legal boundaries recognized in the marine realm are the 3 nautical miles that States or Provincial governments regulate, the 12 nautical miles regulated by national governments as the territorial seas, and the 200 nautical mile area known as the exclusive economic zone (EEZ) regulated by the country owning the shoreline. Under the Law of the Sea, countries maintain sovereign rights to explore, exploit, conserve, and manage living and non-living resources within their EEZ while other nations maintain navigation, overflight, and limited transport freedoms (Gisbon and Warren, 1995). Conflicts of use and interpretation are inevitable both within the governing State as well as among nations. Is exploitation compatible with conservation? Can a foreign flagged vessel navigate through a marine protected area (Burke, 1981)? These issues, among others, make the designation of a completely off-limits MPA beyond a sovereign State's territorial waters politically impossible.

Three approaches have been used to create nationally controlled MPAs (Kenchington, 1990). First, terrestrial park legislation can be extended to cover marine areas (e.g. the marine nature reserve provisions of the Wildlife and Countryside Act of 1981 of the UK). Second, fisheries legislation can be extended to include protection of habitat (e.g. the Fisheries Conservation and Management Act reauthorization of 1997 of the United States). The third approach is to create an entirely new governing agency to oversee a designated area (e.g. Great Barrier Reef Marine Park Authority). The degree to which each of these approaches succeeds depends on the level of actual enforcement of conservation provisions, as assessed by some type of monitoring of conservation objectives. Kelleher et al. (1995) reviewed success of management objectives of 1306 MPAs worldwide and found that the vast majority (71%) were unassessed. Of the remaining 383, 31% had achieved their management objectives, 40% were assessed as moderate in achievement level, and the remaining 29% had generally failed to achieve management objectives. Alder (1996) surveyed perceived success in MPA management in 65 tropical countries, where respondents were government and nongovernmental managers and academics. Only 43% considered their MPA successful, 35% considered their MPA a failure, and 20% were undecided or neutral.

Despite the fact that the primary legal authority to construct marine conservation objectives falls to sovereign nations, protection of the marine environment at large scales is also critical because ocean circulation does not honor legal boundaries and often exceeds the influence of any one nation or group of nations. At an extreme, Mills and Carlton (1998) have suggested creating open ocean reserves, off limits to shipping, extraction, dumping, weapons testing, or floating cities. Such a mandate calls for international jurisdiction, as is currently the case with Antarctica. At present, there are few legal tools that can be used to conserve marine biodiversity over all of the oceans' biogeographic zones, particularly on the high seas.

International treaties, conventions, and regulatory organizations addressing marine conservation issues are many and varied (Table 1). However, the majority of these agreements do not address area-based conservation and hardly any specifically call for the designation of MPAs. Several conventions designate a specific area as worthy of conservation concern (e.g. the Hague Declaration on the Protection of the North Sea), without specifically recommending the area, in whole or in part, be declared an MPA. Other conventions call for the designation of a set of areas based on a particular conservation issue (e.g. the International Conference on Tanker Safety and Pollution Prevention designation of Particularly Sensitive Sea Areas relative to marine pollution) or based on a particular intersection of unique and valuable biological criteria (e.g. UNESCO Man and the Biosphere Program's Biosphere Reserves). Neither of these designations carry enforcement provisions or complete protection from all sources of anthropogenic degradation. At least two conferences called specifically for the designation of MPAs: the IUCN Conference on Marine Protected Areas in 1975 and the Jakarta Mandate on Marine and Coastal Biologi-

Table 1
Major marine environmental laws, including those which designate marine protected areas[a]

	Year	Convention/Organization
	1946	International Convention for the Regulation of Whaling; International Whaling Commission (IWC)
	1958	Geneva Conventions on the Law of the Sea (continental shelf, high seas, fishing)
(Y)	1959	International Maritime Organization (IMO) established (areas to be avoided; particularly sensitive sea areas)
Y	1971	Convention on Wetlands of International Importance (Ramsar; designate wetlands of national importance)
Y	1971	UNESCO Man and the Biosphere Program (MAB; biosphere reserves)
Y	1972	Convention for the Protection of the World Cultural and Natural Heritage (world heritage sites)
	1972	London Convention on the Prevention of Marine Pollution by Dumping of Wastes and Other Matter
	1973	3rd UN Conference of the Law of the Sea (EEZ established)
(Y)	1973	International Convention for the Prevention of Pollution from Ships (MARPOL; special areas)
Y	1975	IUCN Conference on Marine Protected Areas
(Y)	1975	UN Environmental Program (UNEP) Regional Seas Program (Regional Seas)
(Y)	1978	International Conference on Tanker Safety and Pollution Pevention (particularly sensitive sea areas)
	1979	Convention on the Conservation of Migratory Species of Wild Animals
(Y)	1980	Convention on the Conservation of Antarctic Marine Living Resources (CCAMLR)
	1982	UN Convention on the Law of the Sea (UNCLOS)
	1987	Convention on International Trade in Endangered Species (CITES)
	1989	Salvage Treaty
(Y)	1990	Hague Declaration on the Protection of the North Sea
	1991	International Convention of Oil Pollution Preparedness, Response and Cooperation
	1991	UN General Assembly High Seas Drift Net Resolution
	1992	UN Conference on Environment and Development (UNCED) Convention on Biological Diversity
	1992	UN Conference on Environment and Development (UNCED) Agenda 21
(Y)	1992	Convention on the Protection of the North-East Atlantic
(Y)	1992	Convention on the Protection of the Marine Environment of the Baltic Sea Area
	1995	UN Agreement on Straddling and Highly Migratory Fish Stocks
	1995	Global Programme of Action for the Protection of the Marine Environment from Land-Based Activities (GPA)
Y	1995	Jakarta Mandate on Marine and Coastal Biological Diversity (marine protected areas)
	1995	FAO Code of Conduct for Responsible Fisheries

[a] (Y), area-based conservation indirectly addressed. Y, Convention/Organization specifically addresses some form of area-based conservation.

cal Diversity in 1995. Neither conference carries the weight of law. Although most international conventions to date do not designate MPAs, they can have substantive effects on marine conservation issues and in some cases may be more effective than reserves. For instance, several conventions and the associated international organizations regulate pollution (e.g. MARPOL), resulting in a higher standard of environmental quality in all marine areas, including MPAs.

3. Reasons for establishing marine protected areas

In general, the reasons for establishing protected areas are varied but include scientific, economic, cultural, and ethical factors (Jones, 1994). We arbitrarily chose 30 papers to examine the stated reasons for establishment of marine reserves as well as the values underlying the designation (Table 2). Rarely was there a single motivating factor for reserve designation. Almost all (93%) of the papers reported a need for some form of protection of local marine resources. Corollary reasons included the need to maintain biodiversity (67%), the need to promote or control tourism (67%), and the need to enhance fisheries through protection or management (53%). Two major themes emerged: the MPA as a fisheries management tool — a sustainable economic approach to biomass conservation; and the MPA as a national park protecting unique habitat and resident marine communities — a biodiversity conservation approach which may contain economic overtones as in the case of catering to ecotourism.

All papers expressed the desire to maintain or increase the value of the system, although types of valuation varied (Table 2). The most common value was economic (90%), often stated as a need for sustainable development or enhancement of tourism. Related values included environmental (87%), expressed as the need to protect coastal processes or services, and research and education (80%) expressed as the need to monitor environmental change and biological resources. Closely related to environmental concerns were ecological values (70%) such as protecting rare or ecologically important species. Roughly a third of the papers mentioned socio-cultural values including the need to maintain historical sites or preserve cultural use by indigenous people. Slightly fewer papers listed maintenance of aesthetic value, such as the beauty of the landscape or some unusual geological attractions, as important. Only three papers mentioned intrinsic value as a reason for marine reserve designation. In general, the reasons for creating marine and terrestrial protected areas are similar, and both have a large share of paper tigers with little management, regulation, or modification of human behavior.

4. Reserve design

Reserve design has been a linchpin of terrestrial conservation. Issues such as where, how big, how many, how connected, and what arrangement have all been debated in the conservation literature (Meffe and Carroll, 1994). Can terrestrial paradigms translate into the marine realm? Terrestrial reserves are often isolated patches of habitat (Kenchington and Agardy, 1990). Because the majority of organisms protected by such reserves

Table 2
A general survey of the reasons for MPA designation and the underlying values

	Total ($N = 30$)	%	References [a]
Reason for MPA designation			
Maintain biodiversity	20	67	3, 4, 6, 8–12, 14, 16, 17, 20, 22, 23, 24, 26, 27, 29
Fisheries management and protection	16	53	13–20, 23–30
Promote/control tourism-recreation	20	67	2–8, 10–12, 14, 15, 17–19, 23, 24, 27–29
Protection of marine environment	28	93	1–3, 5–8, 10–30
Primary value underlying MPA designation			
Environmental (services; coastal protection)	26	87	1–6, 9–17, 19, 21–30
Ecological (rare or important species)	21	70	1, 3–5, 11–17, 19, 21–29
Economics (sustainable development; tourism)	27	90	2–8, 10–19, 21–30
Aesthetics (maintain beauty; attractive landscape)	10	33	1–3, 5, 7, 12, 13, 17, 19, 24
Societal/political (maintain history and culture)	12	40	1, 3, 6, 7, 12, 19, 21, 23–25, 27, 29
Research and education (monitoring; generic education)	24	80	1–12, 14–16, 18, 19, 21, 23–25, 27–29
Intrinsic (non-human rights)	3	10	6, 10, 15

[a] 1, Bunce et al., 1994; 2, NERC, 1973; 3, Ray, 1976; 4, Allen, 1976; 5, Ballantine and Gordon, 1979; 6, Silva et al., 1986; 7, Lien and Graham, 1986; 8, Sybesma, 1988; 9, Bohnsack et al., 1989; 10, Tisdell and Broadus, 1989; 11, Dhargalkar and Untwale, 1991; 12, Kelleher and Kenchington, 1992; 13, Wolfenden et al., 1994; 14, Hockey and Branch, 1994; 15, McNeill, 1994; 16, Bohnsack, 1993; 17, Kelleher et al., 1995; 18, Rowley, 1994; 19, Kenchington and Agardy, 1990; 20, Towns and Ballantine, 1993; 21, Breceda et al., 1995; 22, Ticco, 1995; 23, Kenchington and Bleakley, 1994; 24, Kennedy, 1990; 25, Gubbay, 1993; 26, Harmelin et al., 1995; 27, Harriott et al., 1997; 28, Jennings et al., 1996; 29, Alder, 1996; 30, Alder et al., 1994.

are either not mobile or not readily able to cross the impacted matrix surrounding reserve sites, links between reserve patches are limited (Kenchington and Agardy, 1990) and both buffer and corridor design also need to be considered (Dyer and Holland, 1991). By contrast, local endemism is infrequent in marine systems (Hockey and Branch, 1994) due to the fact that the surrounding medium — water — supports life as well as transports it (Norse, 1993; Ruckelshaus and Hays, 1998). As a result, linkages are spatio-temporally widespread and the scale of marine ecosystems is large (Kenchington and Agardy, 1990). This makes it difficult to fence a portion of the ocean and recognize the boundary as anything but political for all but the most localized species.

How large does an MPA have to be to conserve mobile species? Several models examining marine reserves as fishery management tools have suggested that effective reserve size is highly sensitive to species' mobility. However, increasing mobility may also allow for an unhindered fishery even at reserve size approaching 50% of total species habitat (Polachek, 1990). Clark (1996) and Lauck et al. (1998) suggest that reserve size needs to be extremely large — 50–90% of total habitat — to hedge against the uncertainties of overexploitation.

In general, the degree of protection an MPA affords can be assessed as a function of species' dispersal distance and site fidelity (Kenchington, 1990; Fig. 1). Species with high site fidelity and low dispersal capabilities (Fig. 1, box A), such as the sea palm *Postelsia*, are obviously protectable by even a small reserve. Species with low site fidelity and high dispersal abilities (Fig. 1, box D), such as swordfish, are unenclosable. Species with high site fidelity and high dispersal abilities (Fig. 1, box B), such as corals (larval dispersal) or seabirds (adult dispersal), may be protectable, but only at certain life history stages and/or times of season. Even during periods of relative site attachment seabirds can range well beyond the modal limits of current MPAs. Satellite telemetry tracks of foraging Magellanic penguins, *Spheniscus magellanicus*, from a colony at Punta Tombo, Argentina, show that breeding penguins forage up to 600 km away from their nest site (Boersma, unpublished data; Fig. 2). Such distances defy national jurisdiction over MPA designation. Finally, species with habitat fidelity and relatively-sized dispersal tendencies (Fig. 1, box C), such as reef fish with short larval periods, may be protectable in large reserves. However, many habitat-dependent marine species exhibit protracted planktonic larval stages (e.g. coral reef fish: range 20–130 days, Carr and Reed, 1992; temperate Pacific invertebrates: range 0–230 days, Strathmann, 1987) allowing settlement to occur hundreds to thousands of kilometers from the source population and decoupling recruitment dynamics from local larval production (Carr and Reed, 1992; Ruckelshaus and Hays, 1998). Assuming that an ecosystem will have representatives in at least three of these four categories, the current modal size of MPAs is inadequately small. In fact, the largest MPA — the Great Barrier Reef Marine Park — is not large enough to protect even two of the four types completely (Fig. 1).

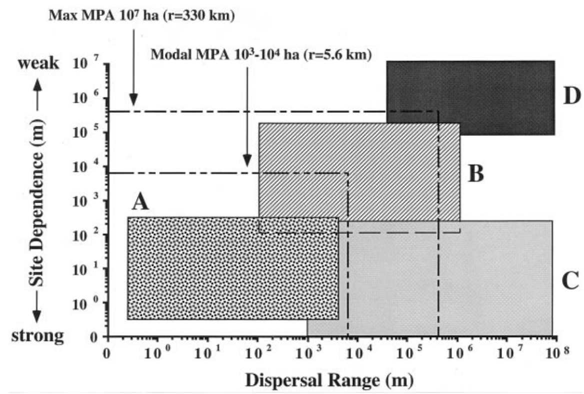

Fig. 1. A graphic model of estimated MPA size needed to contain all life history stages of marine organisms categorized by level of site dependence and dispersiveness. (A) Highly site-dependent organisms with highly restricted dispersal; (B) habitat-dependent organisms with limited dispersal; (C) site-dependent organisms with extensive larval or adult dispersal phase; (D) migratory/pelagic species. Boxes represent MPAs estimated as a circular area where the radius (i.e. longest distance an organism could travel to get to the nearest edge) is indicated by the limits of the box (modified from Kenchington, 1990; World Resources Institute, 1994).

Because the extreme dispersal capabilities of many marine organisms make them virtually uncontainable through all life history stages, many authors have suggested that MPAs be a network of sites designed to accommodate bio-physical patterns of larval dispersal and recruitment (Ballantine, 1991; Man et al., 1995; Ballantine, 1997; Allison et al., 1998). Current strength and direction vary with a range of physical parameters, including hydrography, season, and interannual climatic variation (Carr and Reed, 1992). Roberts (1997) calculated upstream and downstream transport envelopes around 18 reef sites in the Caribbean, based on an average larval competency period of 1–2 months and a detailed current map. Envelope size varied widely as a function of current speed and direction. The average distance larvae would have to travel between sites (minimum inter-reserve distance for a one month larvae) was 145 km and spanned across an average of two countries' jurisdiction. If the goal of an MPA is to protect a functioning community rather than a single species, network arrangement and component size must incorporate the dispersal pattern of all species (Allison et al., 1998).

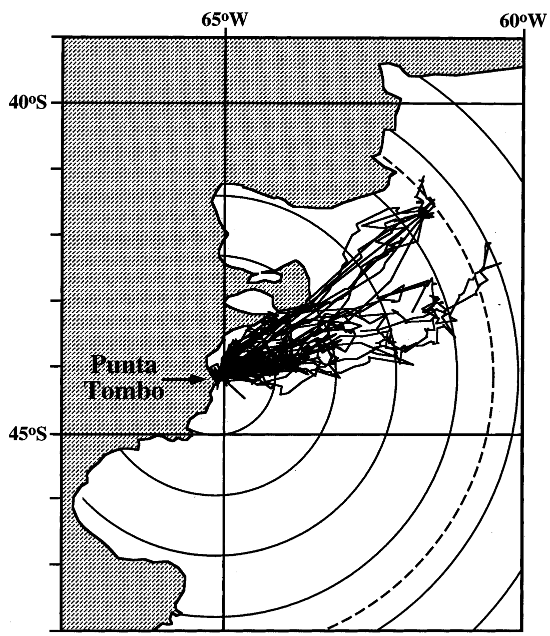

Fig. 2. Satellite telemetry tracks of 19 foraging Magellanic penguins leaving from and returning to a colony at Punta Tombo, Argentina (1995–1998). Concentric circles are 100 km annuli. The dashed circle indicates the areal coverage of the world's largest MPA (Great Barrier Reef Marine Park) superimposed as a hemisphere (radius = 467 km).

Ballantine (1997) summarized the aforementioned constraints in four basic marine reserve design principles: (1) Representation — all habitats or biogeographic zones need to be adequately represented within a reserve system, (2) Replication — each such habitat should be multiply represented, (3) Network — the multiple reserves within each such habitat should be chosen to function as a metacommunity, and (4) The reserve area needed per biogeographic zone per national jurisdiction will probably be 20–30% of the total. At present, many reserves are too small and too isolated to provide adequate population-level protection, let alone protection for ecosystem structure and function. The fourth criterion is especially difficult to envision politically as reserving a significant fraction of coastal area is likely to inhibit entrenched subsistence, economic, and societal uses. To combat this problem, multiple-use zoning has been suggested (Kelleher and Kenchington, 1992), as have locally managed sustainable harvest programs such as community development quotas (CDQs) or management and exploitation areas (MEAs, Castilla and Fernandez, 1998). These latter approaches rely on the assumption that a non-transferable investment in the resource linked to local culture will promote biologically responsible stewardship.

5. Threats to the ocean

Proponents of marine reserves argue that protected areas are important to protect diversity, provide refugia and stock for recruitment, buffer against natural and human degradation, and provide a field laboratory for scientific research. Opponents suggest that the reserves do not work, add unnecessary regulation, are costly, and may have adverse impacts on traditional users of the areas. Moreover, the vast majority of studies on MPA effectiveness have dealt selectively with the issue of fisheries, ignoring other factors which might affect, or even compromise, reserve efficacy. In the following section we review the major threats to the world's oceans — fisheries overexploitation, coastal development, pollution, and species introductions — and address whether and how marine reserves would be a useful conservation tool.

5.1. Fisheries

At present, commercial fisheries take over 3000 marine species, one-third of which are also harvested recreationally (Kaufman and Dayton, 1997). Almost 70% of fished stocks are listed as 'fully fished, overfished, depleted, or recovering' (World Resources Institute, 1996). Pauly and Christensen (1995) estimated that the biomass extracted by marine fisheries equals 8% of global aquatic primary production, and 24–35% in upwelling and continental shelf ecosystems. Estimates of world sustainable yield vary, but there is general agreement that this benchmark is a dynamic function (Ludwig et al., 1993) and that we are approaching if not exceeding it (World Resources Institute, 1994). Fisheries can also have a substantial effect on ecosystem structure and function through gear-mediated habitat alteration/degradation chiefly via removal of biogenic and/or physical structuring in benthic systems. Resultant biological effects include change in: adult abundance (beyond the direct effects of the fishery), larval recruitment, juvenile growth rate and mortality, and total ecosystem production (Auster et al., 1996).

Can marine protected areas conserve or preserve stocks of commercially or recreationally fished species? Recent opinion suggests the answer is yes (Roberts et al., 1995; Pitcher, 1997; Schmidt, 1997; Allison et al., 1998). Theoretically, creation of a no-take marine reserve would allow population growth through simultaneous decreases in adult mortality and increases in average female fecundity, and may additionally buffer the population against the cumulative effects of harvest pressure and environmental extremes (Carr and Reed, 1992). Adult biomass within the reserve would increase as both number and size of individuals expands in response to relaxed fishing pressure (Polachek, 1990). Leakage of 'surplus' adults (spillover) and larvae (larval replenishment) across reserve boundaries will create a sustainable current and future supply of fishable individuals (Polachek, 1990; Plan Development Team, 1990). Taken together, these potential effects of marine reserves hedge against growth overfishing (the tendency for fishing mortality to select against larger sizes) by providing areas in which large individuals will remain safe (Polachek, 1990) as well as recruitment overfishing (a decline in the number of individuals entering the fishable population) by providing a steady supply of larvae to become future fishable cohorts (Roberts and Polunin, 1991). These latter effects create an economic incentive for local fishers to support or even patrol reserves (Johannes, 1982; Rowley, 1994). In addition, there has been the suggestion that reserves may enhance local diversity, habitat complexity, and community stability over chronically fished/overfished areas (Dugan and Davis, 1993). Conversely, increasing size, abundance, and perhaps diversity of upper trophic-level fishes (i.e. formerly fished predators) may alter community structure through top-down regulatory changes.

Several studies have concluded that reserves increase the size and number of a variety of formerly fished species relative to surveys conducted at the same site prior to reserve designation or in equivalent, but fished, habitat (Alcala and Russ, 1990; Harmelin et al., 1995; Roberts, 1995; Jennings et al., 1996; McClanahan and Kaunda-Arara, 1996; see also reviews by Roberts and Polunin, 1993; Rowley, 1994; Table 3). When biomass of one or a very few former target species are assessed, the results are usually unequivocally positive, particularly in the first few years following reserve designation.

Although target species populations may be enhanced inside a reserve, when multiple species are assessed the pattern becomes muddied (Table 3). Roberts (1995) showed significant effects of area (fished vs. unfished), water depth, and time since reserve was implemented on the biomass and abundance of a range of coral reef species, although not all species increased significantly in all categories tested. By contrast, Polunin and Roberts (1993) found inconsistent differences in species abundance, size, and target-species biomass as a function of fishing pressure (none, light, heavy). Dufour et al. (1995) found differ-

Table 3
A summary of some of the effects of reserve enforcement on the resident species [a]

Marine protected area	A	L	B	P	R	H	SF	S	Re	Dp	E	Area	Reference
Belize													
Hol Chan	y?	y?	y?	y	–	–	–	–	–	y?	4	‘Small’	Polunin and Roberts, 1993
Hol Chan	(y)	(y)	y [b]	y	y	–	–	–	–	–	7	2.6 km^2	Roberts and Polunin, 1994
Chile													
Punta El Lacho	y [c]	y	(y)	y	nr	–	y	–	–	y	var [d]	0.5 lin km	Castilla and Durán, 1985
Mehuin	y [e]	y	(y)	y	nr	–	y	–	–	–	5	–	Moreno et al., 1986
Punta El Lacho	y [f]	–	–	y	y	y	y	–	–	–	5	0.5 lin km	Durán and Castilla, 1989
France													
Banyuls-Cerbère	y	y	(y)	y	y	–	–	–	–	y	7	10 km^2	Bell, 1983
Banyuls-Cerbère	?	y	–	y?	n	–	–	–	–	y	13	1.5 km^2	Dufour et al., 1995
Kenya													
Malindi, Watamu, Mombasa	nr	y [g]	nr	–	y [h]	y	y	–	–	–	–	–	McClanahan and Mutere, 1994
Kisite, Mpunguti	y	y	y	–	–	–	–	–	–	y	–	23 km^2	Watson and Ormond, 1994
Four reserves	y	(y)	y	y [i]	–	y	y [j]	–	–	–	4–26	6–28 km^2	McClanahan and Obura, 1995
Mombasa	y	(y)	y	–	(y)	–	–	y?	–	–	3	10 km^2	McClanahan and Kaunda-Arara, 1996
Mediterranean													
Carry	y	y	y	y	y	–	–	–	–	–	13	0.85 km^2	Harmelin et al., 1995
Netherlands Antilles													
Saba	y	?	y	y	y? [k]	–	–	n [l]	–	y	6	–	Roberts, 1995
Saba	y?	y?	y	y	–	–	–	–	–	y?	4	‘Small’	Polunin and Roberts, 1993
New Zealand													
Goat Island	? [m]	y [n]	–	(n)	y	–	–	–	–	–	12	~5 lin km	Cole et al., 1990
Phillipines													
Sumilon, Apo	y	(y)	y [o]	y	y	(y)	(n)	–	–	y	8	Small	Russ and Alcala, 1989
Sumilon	(y)	(y)	y [p]	y	–	–	–	(y)	–	–	10	0.12 km^2	Alcala and Russ, 1990
Sumilon, Apo	–	y	y [q]	y	–	–	–	–	–	–	var [r]	–	Russ and Alcala, 1996a
Apo	y	–	–	y [s]	y	–	–	y	–	–	11	Small	Russ and Alcala, 1996b

Table 3
A summary of some of the effects of reserve enforcement on the resident species [a]

Marine protected area	A	L	B	P	R	H	SF	S	Re	Dp	E	Area	Reference
Seychelles													
Four reserves	(y)	(y)	y	y?	y	–	–	–	–	–	–	22.8 km^2	Jennings et al., 1996 [t]
South Africa													
Tsitsikamma	(y)	y [u]	–	–	–	–	–	–	–	–	25	350 km^2	Buxton, 1993
De Hoop	y	n	–	y	y	–	–	–	–	–	var [v]	46 lin km	Bennett and Attwood, 1991

[a] The first set of columns represent population-level effects (A, abundance; L, individual size; B, biomass), the second set indicate community-level effects (P, density of predators and/or upper trophic-level fishery targets; R, richness; H, habitat complexity/structural relief; SF, community structure and function), the third set represent local fishery effects (S, spillover of adults; Re, replenishment from larval export), and the fourth set provide data on physical factors likely to influence the likelihood of finding an effect (Dp, depth; E, years enforced; Area, lineal or areal size). All effects are positive (better in reserve) unless otherwise designated. y = greater in reserve; (y) = inferred from higher order calculations, as in higher biomass probably indicates high abundance and size classes; y? = possibly greater in reserve, no clear effect; ? = for some species yes, for others no; – = information not taken (not available); nr = information not relevant; prob = probable; poss = possible.

[b] Only commercial fish reported.

[c] Assessed for *Concholepas concholepas* (gastropod) only.

[d] From −1 to 2 years.

[e] Assessed for *Concholepas concholepas* (gastropod) only.

[f] Assessed for *Concholepas concholepas* (gastropod) only. Densities rose initially and then declined. The same held true for diversity.

[g] Refers to hard coral size.

[h] Refers to number of hard coral species.

[i] Upper trophic level predators confined to snappers (Lutjanidae), predators on sea urchins were also significantly different.

[j] Change in sea urchin abundance as a function of increasing predation by protected fish, transitions reef to higher hard coral cover, changing habitat complexity and associated community features.

[k] Point diversity (number of species per count) increased with time, but overall diversity (total number of species) constant.

[l] Biomass increases in the fished area over time attributed to decreased fishing pressure rather than spillover.

[m] Except a clear positive trend for rock lobster.

[n] Assessed for snapper only.

[o] Assessed for *Epinephelus fuscoguttatus* (grouper) only.

[p] Assessed as both yield and catch per unit effort.

[q] Assessed specifically for large piscivorous fish which are also fishery targets.

[r] Sumilon −2 to 9 years; Apo 1 to 11 years.

[s] Assessed specifically for large piscivorous fish which are also fishery targets.

[t] Two effectively protected reserves were contrasted with two relatively unprotected reserves.

[u] The effect of smaller individuals in fished areas was to skew the sex ratio of two sex-changing parrotfish (Sparidae) towards female.

[v] From −2 to 4.5 years.

ences in abundance at the species level inside versus outside of a reserve, but no consistent pattern of positive effect of marine area protection on total species abundance.

Higher order effects, such as changes in species richness or changes in community structure and function, have only been superficially explored (Table 3) and are as apt to be negative as positive relative to the within-reserve community. McClanahan and Kaunda-Arara (1996) found that species richness in Kenyan MPAs was 25% higher than in transition zones and 50% higher than in totally unprotected areas. Harmelin et al. (1995) also found a significant difference (+16%) in species richness in a Mediterranean MPA but only when censuses were pooled. Because rare species are only observed occasionally, richness may not be a sensitive indicator of reserve efficacy. Changes in coral cover and coral diversity associated with overfishing have been linked to alterations in coral reef community structure, from topographically complex coral-diverse areas to coral-depauperate communities dominated by urchins (McClanahan and Mutere, 1994; McClanahan and Obura, 1995). One of the oldest studies of second and third order effects of marine reserves has monitored populations of the predatory gastropod, *Concholepas concholepas* (the loco) in Chile. After initial increases in target species biomass, secondary effects included increased predator pressure on herbivores, which allowed macroalgae to grow. Tertiary effects at one site included succession by barnacles of macroalgae, a benthic habitat change which resulted in a decrease in loco density (Castilla and Durán, 1985; Moreno et al., 1986; Durán and Castilla, 1989). Clearly, the long-term effects of marine reserves need to be monitored before we can concretely assess their efficacy.

Whether the increasing density of larger adults will flow sustainably across reserve boundaries into fishable areas is another matter. Spillover has rarely been quantitatively assessed (Table 3). Alcala and Russ (1990) used catch-per-unit-effort (CPUE) data just outside the Sumilon Island reserve in the Philippines to indirectly demonstrate spillover. During the time the reserve boundaries were enforced, CPUE was 33–58% (variation by gear type) higher than following reserve breakdown. McClanahan and Kaunda-Arara (1996) suggest that lack of a definitive spillover effect at an MPA in Kenya may have been due to intensified fishing along a thin margin at the park (no-take) boundary which effectively prevented dispersing adults from reaching the surrounding reserve (limited take). Whether spillover effects at small reserves are sustainable is controversial (Clark, 1996). Tagging studies have shown that home ranges of some species may be quite broad (e.g. Hunt, 1991 as cited in Rowley, 1994; Holland et al., 1993). Thus, increased capture at reserve boundaries may be an indication of deficiencies in reserve size or placement rather than a positive emergent property of the reserve per se. In general, the size of monitored no-take MPAs (Table 3) is small relative to modal MPA size (100 km^2; Fig. 1), and far smaller than sizes recommended as a long-term buffer against the uncertainties of fisheries exploitation and environmental change (Lauck et al., 1998).

Larval replenishment is even harder to document (Table 3). Tegner (1992) outplanted green abalone, *Haliotis fulgens*, adults into a reserve and subsequently documented higher than expected recruitment of juveniles outside the reserve. Obviously, the interaction between life history and reserve design plays a key role in determining the efficacy of the area as a significant generator of future fishable adults via larval transport (Carr and Reed, 1992; Roberts, 1997; Allison et al., 1998). Attempts by the Florida Keys National Marine Sanctuary to design a network of three no-take zones along the major current flow, such that larvae produced in the first area would settle down-current in the remaining reserves among other places, was defeated in favor of a more politically modest but biologically naive single area solution (Ogden, 1997).

5.2. Development

Many human activities other than fishing depend, directly or indirectly, on the sea. Peterson and Lubchenco (1997) define five broad categories of ecosystem services the world's oceans provide (other than the extractive value of the world's fisheries), three of which are concentrated in continental shelf environments: (1) transformation, detoxification, and sequestration of pollutants, (2) coastal ocean-based recreation, tourism, and retirement, and (3) coastal land development and valuation. Ironically, each of these services is dependent on the continued health of the relevant ecosystems we are using to the point of degradation. In fact, we are aware of these services because they add value to our lifestyles, whether it be sewage removal or increased property value as a function of coastline view potential.

Pressure on coastal environments, either directly through habitat alteration or loss as a consequence of usurpation by humans, or indirectly as a consequence of the cumulative effects of dense human presence, is increasing. Bryant (1995) estimates that fully half of the world's coastal ecosystems currently sustain a moderate to high risk of development-related threat. Highly developed regions, such as Europe, have an even higher percentage of threatened coastline (86% at moderate or high risk). Coastlines undergoing development are locally subject to habitat modification as wetlands are filled or dredged, river courses are channelized, tidelands are diked, beaches are armored, and jetties and seawalls are built. Other than the obvious habitat loss, these modifications can produce geographic ripples as the flow of water, sediment, and nutrients are altered. Of the 1108 coastal marine protected areas assessed by Bryant (1995), 59% occurred in areas currently sustaining a high risk of degradation due to development-related activities. White (1986) compared features of reef habitat quality, including total coral, topographic relief, noticeable structural damage, and butterfly fish (an obligate coral feeder) species richness among Indo-Pacific reef sites of varying protection. Sites with no legal or field protection close to centers of human habitation suffered the most degradation, becoming unsuitable for sustaining healthy reef communities. As the human population continues to expand, this trend will not get better.

In apparent contrast to the destructive pressures of habitat conversion and overextraction of natural resources, increasing economic value is being realized from ecosystem preservation in the form of tourist and amenity income. Surprisingly, ecotourism accounts for 5–15% of global tourist dollars and this sector of the economy is growing at approximately 30% per annum (Giannecchini, 1993; Miller, 1993). Tourism in various Caribbean nations accounted for 17–55% of gross national product (1977 US dollars, Beekhuis, 1981). In 1990, Caribbean tourism employed 350,000 people and generated 8.9 billion US dollars (Dixon et al., 1993).

Preservation of coastal systems can also add to land values as retirees and vacationers are willing to pay more for the aesthetic experiences provided by a functioning, relatively undisturbed ecosystem. For instance, land bordering the Chesapeake Bay increased 5–25% in value following designation of Maryland's Chesapeake Bay critical area and New Jersey's Pineland regulations (Beaton, 1988). Punta Tombo, Argentina houses the largest Magellanic penguin colony in the world (approximately 250,000 breeding pairs). Although the region derives its principle economic benefit from oil transport and fisheries, tourism has been growing rapidly (Fundacion Patagonia Natural, 1996). The penguin colony is an important tourist destination attracting approximately 50,000 visitors each year and generating millions of USD equivalents in primary and secondary income (Fundacion Patagonia Natural, 1996). Between 1987 and 1997, the number of breeding pairs of penguins at Punta Tombo declined by 16% (Boersma, 1997) due to a combination of factors including chronic oil pollution and potential competition with coastal shelf hake and squid fisheries. Reserve designation of the colony and surrounding waters may help protect penguins and the ecosystem of which they are a part, providing a sustainable ecotourism input to the local economy.

Because fisheries overexploitation is commonly viewed as the chief problem marine communities face, marine reserves have been focused on exclusion of this user sector over others (e.g. tourists). However, even relatively environmentally sensitive uses of coastal ecosystems can degrade ecosystem structure and function via increasing service demands (e.g. nutrient and toxics transformation) and visitation. There is gathering evidence that increasing tourist use, including boat-bound as well as diver visits, can negatively affect species abundance, species richness, as well as water and benthic habitat quality. Coral breakage rates per 30 min of scuba diving ranged from 0.6 to 1.9 (maximum 15) at four MPAs in eastern Australia and there were no significant differences between novice and experienced divers. Sites sustained 2000–20,000 dives per year (Harriott et al., 1997). Based on an analysis indicating that decreasing coral cover and reef species diversity was a function of increasing number of dives, Dixon et al. (1993) calculated a 'threshold level' of dives (4000–6000) per year beyond which reef quality in the Bonaire Marine Park would be significantly negatively impacted. Extrapolating to total divable area in Bonaire and controlling for overuse of the more accessible sites, diver carrying capacity was calculated at 190,000–200,000 dives per year. As of 1991, the island was at 95% capacity with an annual increase of 10%. By contrast, some authors have found no apparent effect of divers on abundance, diversity, or short-term stability of reef fish assemblages (Greene and Shenker, 1993). Pressure to continue expanding local tourist industries at the potential expense of the local environment are great, particularly in island nations. Restrictions on tourist numbers in the Galápagos (capped at 12,000 in 1973) have been relaxed several times to more than double the original cap. Associated effects include increases in the resident population as Ecuadorians seek to fill jobs in the industry (Broadus, 1987). Recent degradation of the Galápagos reefs due to the combined effects of fishing, tourism, and development pressures have prompted the Ecuadorian government to designate an MPA ranging to 64 km offshore of the islands (Holden, 1998).

5.3. Chemical and biological pollution

In 1994, approximately 44% of the world's population lived within 150 km of a coastline (Cohen et al., 1997). Of the 613 cities with more than 500,000 residents (1995), almost 40% are situated in coastal areas and 75% of the megacities (> 10 million residents) are coastal (World Resources Institute, 1996). Kenchington (1990) lists a wide spectrum of anthropogenic pollutants in coastal marine environments including: herbicides and pesticides, antifouling chemicals, sediments, petroleum hydrocarbons, phosphates and nitrates from sewage, heavy metals, surfactants and dispersants, and chlorine, as well as changes in salinity and temperature. The National Academy of Sciences (1994) defines toxics as one of seven major anthropogenically driven issues causing regional to global deterioration of coastal ecosystems, primarily due to bioaccumulation and biomagnification of heavy metals, PCBs and PAHs, and reproductive process inhibitors. In addition to localized pollution, several regional to global anthropogenic pollutants affect the marine environment. Global climate change, abetted by greenhouse gas emissions, can have huge impacts on coastal marine ecosystems through changes in precipitation patterns, increasing storm intensity, alteration of coastal currents and upwelling intensity, increasing upper ocean temperature, and ultimately, sea level rise (National Academy of Sciences, 1994).

On a local scale, designation of an MPA could affect point sources of pollution by regulating discharge from coastal and vessel sources (Allison et al., 1998). For instance, in the United States, penalties for illegal dumping within a National Marine Sanctuary are higher than in undesignated areas. However, waterborne pollutants can travel extreme distances, making local prevention of limited use. The *Exxon Valdez* oil spill impacted coastlines more than 750 km from the spill site (Royer et al., 1990). Chronic oil pollution may be even more widespread than oil spills. Using nine years of data from 14 coastal sites along 3000 km of Argentine coast, Gandini et al. (1994) showed that Magellanic penguin mortality from oil presumably discharged with ballast water during oil transport was approximately 20,000 adults and 22,000 juveniles each year. Airborne pollutants can travel tens of thousands of kilometers before settling in marine systems (Walker and Livingstone, 1992). Thus, the effectiveness of a well-designed network of coastal MPAs could be compromised by regional to global pollution which easily crosses reserve boundaries.

Chemicals are not the only pollutants in marine systems. Biological pollution stemming from directed (e.g. mariculture, fisheries transplants; Baltz, 1991) and inadvertent (e.g. ballast water, hitch-hiker species; Carlton, 1996) introductions can also dramatically and irrevocably alter native ecosystems worldwide (Carlton, 1989). Introduced species can deplete local native populations through predation, competition for food or space, habitat alteration, or introduction of disease. Increasing numbers of introduced species can alter food web dynamics, making the remaining community less resistant and perhaps resilient.

de Fontaubert et al. (1996) suggest that conservation objectives of marine protected areas should include protection of endangered species, maintenance of native species and relevant genetic diversity, and exclusion of human-caused species introductions. Can MPAs provide effective protection against biological pollution? Because many MPAs are along coastlines, within shipping lanes, and near human centers of activity, the chance of biological invasions is high (Carlton, 1996). Many species have been introduced multiple times (Grosholz and Ruiz, 1996). Carlton (1989) found 32 introduced marine organisms in the South Slough National Estuarine Research Reserve (Oregon). In concert, these invaders have colonized and significantly altered hundreds of square kilometers of hard and soft-bottom habitats. The same is true for other U.S. West Coast estuarine reserves (Carlton, 1979). Clearly, designation of an MPA was not enough to change the existing dominance of invaders or prevent new introductions. In fact, papers reviewing the efficacy of specific MPAs rarely mention whether introduced species are present, and if so, what their effects are.

6. Conclusions

On a local scale, marine protected areas can be effective conservation tools, but only in cases where: (1) the design of the reserve is intimately linked to the biology of the constituent species and the physics of the local environment, (2) humans can control the intensity and spread of relevant threats, and/or (3) the scale of the MPA exceeds the scale of the threat (Allison et al., 1998). Because humans are easier to exclude than chemical or biological pollutants, MPAs are most useful as a tool for managing the direct and indirect effects of resource extraction. Although MPAs cannot adequately protect marine resources in areas subject to human-mediated pollutants (e.g. nearshore systems) without additional forms of protection, they can provide recognition of important areas or problems, educational opportunities, and political attention to issues of conservation and sustainable use.

On a global scale, MPAs can only be effective conservation tools if they are substantively representative of all biogeographic zones (Ballantine, 1997). The amount of area reserved per zone has been estimated at 20–30% (Ballantine, 1997; Schmidt, 1997), with modeled estimates as high as 50–90% for adequate protection of some fishery stocks (Polachek, 1990; Clark, 1996; Lauck et al., 1998). At present levels of coastal habitation and human population growth, these numbers seem unrealistically high.

Science can and should provide guidelines to achieve conservation goals for MPAs, including protocols for reserve size, location, and network design; efficacy with respect to relevant local threats; and monitoring of goal attainment. Scientists should be clear about what a well-designed MPA can and cannot do in a given area. Finally, science should provide 'what if' scenarios should social, economic, and political pressures dictate reserve designation which falls short of recommended conservation objectives. Science cannot dictate policy. The Lisbon principles delineate an international policy framework for use of marine resources. Their adoption will likely aid governments and individuals in moving toward more sustainable marine use.

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