Chapter 5

ECOLOGICAL AND OTHER SCIENTIFIC IMPERATIVES FOR MARINE AND ESTUARINE CONSERVATION

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ABSTRACT

Declaration of marine reserves has been opportunistic rather than systematic. Setting aside marine and estuarine areas for conservation purposes will not, however, ensure the continued survival of either species or ecosystems unless the dynamics of what is being conserved are understood. We highlight the differences between terrestrial and marine ecosystems and discuss the consequences of these for marine and estuarine protected areas. These differences mean that well-established and tested land-based theories of conservation cannot be applied uncritically to marine systems. Declaring boundaries, as in the terrestrial park system, does little to preserve most marine organisms because of their size, mobility and life history, and our considerable ignorance of how marine ecosystems function. Consequently, the design of marine reserves must proceed under different guidelines to those used for terrestrial reserves. Actual practices of marine reserve selection, design and management may stray from theory, and this tendency is usually away from any scientific basis.

We discuss design strategies for marine and estuarine protected areas and assess their success by reviewing research done comparing protected areas with non-protected areas. Monitoring of established protected areas is needed to assess the ability of these areas to provide protection of resources into the future. Only monitoring before and after establishment as a protected area will allow assessment of regulations imposed on habitats and species within protected areas and the success of these measures. We critique research that has been done in marine and estuarine protected areas. The ecological integrity and protection of marine resources depends on our ability to monitor the effect of human usage on biota.

INTRODUCTION

Worldwide there is a well-established system of terrestrial national parks, nature reserves and conservation areas (Fenner 1975). There are also well-established methodologies (using algorithms to select sites that are most diverse, unique, pristine etc.) for the designation and management of these areas (Smith & Theberge 1986; Margules 1989). In contrast, there exists no parallel system nor theories for preservation of marine habitats as reserves (Fenner 1975; Ballantine 1987; McNeill 1991). Conservation in marine habitats lags a long way behind the terrestrial scene. Roughly about 4.1% of our marine territories is presently in reserves (compared with approximately 5.3% of the land), but nearly 90% of this is in the Great Barrier Reef Marine Park, leaving less than 1% represented in other geographic regions (Ivanovici 1987; McNeill 1991; Bridgewater & Ivanovici 1993). Most existing marine and estuarine protected areas have been chosen in a haphazard manner, relying mainly on the advocacy of individuals and user groups (e.g. divers) to make the case. This great difference may be due, at least in part, to the relative inaccessibility of the marine environment to humans, the view of the marine environment as common property (Hardin 1968) and the common misconception that resources in the sea are inexhaustible.

This neglect of the marine environment has led to detrimental effects of overexploitation of ocean resources, increased human populations, and terrestrial influences (Ray 1976; Salm 1989; Hatcher, Johannes & Robertson 1989). Past conservation efforts have stopped at the edge of the sea, and the ocean and its resources have been treated both as a productive untapped reservoir and as a sink for many terrestrial wastes. Protection of marine resources is therefore urgent.

OBJECTIVES OF MARINE AND ESTUARINE CONSERVATION

Objectives of marine and terrestrial conservation areas are the same - to protect ecosystems and habitats of particular areas, species or communities. Marine protected areas should maintain a range of representative habitats and biota for study and recreation, and replenish adjacent sites subject to extractive exploitation (Salm 1989). Objectives for marine protected areas were suggested by Ray (1976). Those with a largely ecological focus include habitat preservation, species preservation, conservation of genetic resources, research, education and, possibly, special and multiple uses.

CRITERIA FOR SELECTION AND DESIGN OF MARINE CONSERVATION AREAS

A rich literature exists on establishing terrestrial protected areas (Smith & Theberge 1986; Margules 1989) and many of these criteria have been used in the selection of marine protected areas (e.g. Salm 1989). The most widely used classification scheme for marine protected areas is that proposed by Ray (1975). This scheme allows for development of a system of protected areas which will include a range of habitat types and reflects ecosystem processes. The criteria formulated by Ray for selection of protected areas are a combination of ecological, cultural, recreational and educational criteria but also involve the feasibility of physically and economically preserving and managing the area (Ray 1975). The ecological criteria include representativeness, uniqueness, diversity, naturalness, natural unity and inclusiveness. Educational, cultural and recreational criteria consider diversity, types and abundance of organisms, physiography and topography, uniqueness and rarity, climate, weather and oceanographic conditions, cultural value and scientific value. The last set involves pragmatic criteria including the value for research and monitoring, degree of threat or fragility, feasibility, redundancy, national and international value, and educational, recreational and economic value.

Although these criteria for choosing a site as a marine protected area are extensive, it is impossible to be entirely objective in choosing one area over others as worthy of preservation. A systematic selection of marine protected areas is also needed to ensure that areas selected are representative. We are concerned with how ecological science can aid the design of marine reserves.

Ignorance and the need for a better inventory

There is no central database concerning coastal or oceanic ecosystems. Apart from an Australia-wide inventory of marine and estuarine protected areas (Ivanovici 1984), there has been no Australia-wide inventory of marine organisms, landscapes or sites of special scientific interest. This contrasts with the state of terrestrial species conservation where national atlases and listings of rare and endangered species exist for birds, mammals, butterflies, flowering plants and herptiles. Enormous gaps in our knowledge exist for a number of marine habitats (Fairweather 1990a), disciplines such as microbiology, taxonomic groups like the fungi and the plankton, and ecological processes like decomposition and mutualism. The study of ecosystem functioning is patchily distributed across habitats (Fairweather 1990a). Thus, the criteria listed above cannot be applied in Australia.

For example, there is no real system of marine protected areas, natural heritage sites or wilderness in Australia (outside of Queensland coral reefs) to rival the conglomerate system of terrestrial national parks. In NSW, there are 22 marine and estuarine protected areas that make up but a fraction of the coastline and are subject to numerous insults from outside their boundaries. Most, if not all, are not policed in any way.

It is difficult for marine biologists and ecologists to plug into endangered species research programmes because either the state or federal management agencies providing funds do not cover marine waters or the biota of many marine habitats and locations are not well known. Most marine research has, by necessity, been very localised even on the Great Barrier Reef, and many habitats or groups of organisms are not well understood (Fairweather 1990a). We need to catalogue an inventory of most habitats along many parts of our coasts and oceanic waters. We should support the following series of initiatives for all parts of our coastline:

1. a nationwide inventory of marine ecosystems, habitats and organisms which incorporates oceanographic, geomorphic, ecological, genetic and other information (rather than just species lists!). This should be built up over a period of five years or so, and then updated regularly via state of the environment reporting or a similar

procedure. (It is notable how sparse was the marine content of recent federal and state attempts at state of the environment reports.) Cooperation is needed between the relevant divisions of the Commonwealth Scientific and Industrial Research Organisation, Australian Institute of Marine Science, Australian National Parks and Wildlife Service, museums, herbaria, State agencies, universities and other relevant agencies;

2. a clearly identified role for the above authorities identifying and notifying of endangered marine species;
3. clear identification, notification and focus upon endangered or poorly understood marine habitats. This might be done by a dedicated research grant scheme; and
4. more emphasis and action concerning realistic marine habitat conservation. Selection of new areas to be protected could be based on points 1, 2 and 3.

COMPARISON OF MARINE AND TERRESTRIAL ECOSYSTEMS

The difference between land and sea suggests that marine conservation and other environmental issues require a different approach from that used on the land. Characteristics of the marine environment and its organisms have important consequences for the design of marine protected areas. These characteristics differ from those of terrestrial systems, making impossible the uncritical application of well-established land-based theories of reserve design to marine environments. These characteristics are generally not unique to marine systems but are more pronounced (e.g. the habitat as an open system).

The first and most fundamental difference is the greater degree of connectedness among habitats and places in the sea. Complex and dynamic currents, waves and tides move water in the ocean, allowing considerably greater transport of nutrients, material, conditions and species than do the passive media on land, air or freshwater flows.

Despite the three-dimensional nature of the marine environment, the oceans are not homogeneous. Major current patterns create biogeographic provinces within the ocean (see Ray 1975) similar to those found on land. The strength and timing of circulation patterns may vary from year to year (Holloway & Nye 1985), influencing biological patterns. Such an event is the El Nino Southern Oscillation phenomenon that varies in intensity and timing and has massive effects on the Peruvian anchoveta fisheries.

There are similarities between the circulation of water in the ocean and in the atmosphere on land (Drake & Farrow 1989). For example, convergence is a circulation pattern found in both systems, but the biological processes of species trapped in convergences differ between systems. Organisms in an atmospheric convergence are only temporary inhabitants (usually reproductive propagules), not feeding or developing (Drake & Farrow 1989). A marine convergence, in contrast, is a highly concentrated area of biological activity (i.e. species feed and are capable of morphogenetic change), marked by aggregations of plankton and predatory fish (Kingsford 1990).

Despite similarities between circulation in the ocean and atmosphere there is one circulation pattern in the ocean that has no equal in the atmosphere - the phenomenon of upwelling, where circulation brings bottom nutrients to the surface where biological activity is limited by otherwise sparse nutrients. The Leeuwin Current is responsible for upwelling events in summer and winter off Western Australia and results in nutrient enrichment of surface layers (Holloway & Nye 1985).

Perhaps because of these physical properties of the oceans, many marine organisms have a dispersive phase in their life history that allows larvae to be transported large distances from their parent populations (Underwood & Fairweather 1989). This dispersive phase involves ontogenetic morphological change. Such a stage is, however, not unique to marine species and is shared by insects, amphibians and plant seed banks (Fairweather 1991a). For some species, dispersal of larvae may only be a few metres - for example, ascidians (Scheltema 1986a); abalone (Shepherd 1991). In others, larvae may be transported thousands of kilometres - for example, gastropod larvae (Scheltema 1986b). Likewise the period over which dispersal occurs varies among taxa (Scheltema 1986a, 1986b; Fairweather 1991a). Many larvae of marine organisms are not passive and are able to orient themselves to take advantage of currents to some extent, thereby increasing their potential to disperse (Scheltema 1986a). How far larvae disperse is dependent on physical properties - currents, wind patterns and shoreline topography - as discussed above (Kingsford 1990). For instance, substantial recruitment of an urchin was found on rocky reefs between headlands compared with adjacent headlands and capes (Ebert & Russell 1988). Absence of recruitment to headlands was thought to be correlated with the presence of upwelling and cold water plumes associated with these geographical features.

There are two important implications (Underwood & Fairweather 1989) of these physical and biological properties:

1. larvae may not necessarily colonize the same area as the adults that spawned them. For example, original colonization of volcanic islands must have resulted from larvae dispersing long distances (Scheltema 1986b); and

2. because of the hazards during the dispersive stage, the numbers of larvae arriving in an area may bear little relationship to the population size from which they came. This lack of relationship causes great uncertainty in the management of marine fisheries. For instance, recruitment of juveniles to the Western Australian rock lobster fisheries is thought to be dependent on the timing and the strength of the Leeuwin Current.

Knowledge of the scale of dispersal will help define the units of marine areas to be managed (Sammarco & Andrews 1988; Fairweather 1991a). Changes to, or disruption of, features facilitating larval recruitment may endanger return of propagules to sites. Marine protected areas may thus provide protection for some areas that are known to produce propagules (e.g. Cairns & Elliot 1987) and therefore act as a source of recruits. Similarly (Fairweather 1991a), protected areas may provide protection for areas that reliably receive propagules (e.g. Ebert & Russell 1988) and thus act as population sinks (Pulliam 1988).

Water quality is probably of more direct importance to the biota living in it than air quality is to terrestrial life forms. We are also in the uneasy situation of being unable to see 'into' the sea and thus the potential exists, we think, for damage to occur without being detected.

A related concern is our tendency to use the sea as a final receptacle for our liquid waste, including domestic and industrial effluents. Only the most toxic fraction of this is handled as hazardous waste. The capacity of the seas to accept this insult affects only the rate at which an eventual turning point (i.e. saturation, in the sense of too much chemical to *not* alter ecosystem function) is reached. The end point is the same environmental mess.

A problem for conservation management arises because the ebb and flow of seawater means that reserved areas cannot be completely isolated from unprotected areas that are open to impacts. Because of this interconnectedness, it is difficult to define biologically meaningful boundaries for marine protected areas to the extent that they are a feature of terrestrial protected areas. Nominal boundaries will not ensure protection because they cannot provide barriers to water masses or migratory species. Awareness of events both within and outside protected area boundaries are also necessary because current patterns may supply the protected area with nutrients, larvae and pollutants from distant sources. This actually argues for a more extensive marine reserve system with even more units than in the terrestrial sphere and the incorporation of buffer zones to protect core areas. Likewise, consequences of terrestrial activities (such as agricultural practices and run-off from land) may affect marine protected areas even if the activity is not in the vicinity. Salm (1984, 1989) and Batisse (1990) emphasised the importance of considering activities in the entire watershed when managing a protected area.

Another fundamental difference between sea and land is a property of relative accessibility - the way the public perceives marine and terrestrial protected areas. Terrestrial protected areas are available to most people and have become increasingly valued by society (Shafer 1990). Because water is a hostile environment to people, subtidal protected areas are experienced by few. Consequently marine protected areas do not enjoy the same degree of support as terrestrial parks. In general, oceans are seen as a common property resource to be enjoyed and exploited by all (Hardin 1968; Rice 1985). As such, marine protected areas often attempt to accommodate multiple uses, including some exploitation of resources.

DESIGN ISSUES WITH MARINE CONSERVATION AREAS

Boundaries

As mentioned above, a major problem in the design of marine reserves is setting the boundaries. Generally there has been little ecological reasoning behind the delineation of marine park boundaries (Rice 1985; Salm 1989). *Failure to recognize and incorporate* ecologically relevant boundaries within marine protected areas has led to degradation of these areas and they have, therefore, failed in their nominal role. Many marine fish may spawn in the open ocean and delineating such areas is impossible. In some cases there may be natural boundaries to benthic communities, such as discrete rocky reefs or coral atolls. Otherwise, knowledge is required of the connectedness between habitats before appropriate boundaries can be designated.

For many marine species it is impossible to protect their entire life cycle because of their migratory habits and/or the dispersive capabilities of planktonic larvae. However, marine reserves may afford protection to marine species at certain and often critical stages of their life cycle. For example it has been widely accepted that seagrass beds within estuaries act as 'nursery grounds' for some juvenile fish and invertebrates. Protection of these habitats thus ensures conservation of species at this stage of their life cycle. Spawning and breeding areas, migratory pathways, feeding grounds and highly productive areas may need to be protected. Conservation of such areas has demonstrated continued recruitment of species (Sammarco & Andrews 1988), successful export of propagules to other locations acting as population sinks (Ebert & Russell 1988) and supply of recruits to seed other areas (Cairns & Elliot 1987). Reserves may thus protect areas that function as population sources or sinks (Pulliam 1988). Although conservation at the scale of marine ecosystems may be impossible, identifying and protecting areas that are critical to certain stages in the life history of species is feasible.

Size

There has been no consensus over the optimal size of reserves, either terrestrial or marine (see Shafer 1990 for review). Goeden (1979) calculated the critical minimum areas for coral reef fish in the Great Barrier Reef to be 35 km^2. This contrasts with Salm's (1984) conclusion that the core area of coral reef protected areas should be at least 3 km^2. The difference in magnitude between the two studies may be due to ecological differences between the areas and species considered, or methodological differences in sampling used to estimate the core areas. If the aim is to conserve organisms, the optimal size will be species specific and possibly very large. However, economic and political constraints may override any conservation issues or intentions (Hatcher, Johannes & Robertson 1989; Salm 1989).

The realized effectiveness of marine reserves may be due less to their size than the distance between protected areas. For example, Sammarco and Andrews (1988) found that units to be managed in coral reef environments could be estimated by the realized dispersal and larval connectedness among reefs. Similarly, the distance between areas may influence restocking of exploited areas by a protected source (Cairns & Elliot 1987). Knowledge of the degree of connectedness for adults of species between habitats and locations will further aid determination of the appropriate unit of a park.

Network or large area?

There has also been much debate over the disaggregation or aggregation of reserves (Salm 1989). Those that argue for preservation of networks of small reserves believe that such an approach will:

1. preserve more species;
2. ensure survival of species competing for limited resources; and
3. ensure survival through catastrophic events.

Yet these arguments have been neither confirmed nor denied in the marine context. Talbot and Anderson (1978) recommended marine reserves to be very large, after calculating the dispersal range of an average planktonic larva. In contrast, Kenchington (1988) believed a network of smaller reserves throughout a very large area is more effective in maintaining a representative source of recruits from undisturbed habitats. In support of this idea, Goeden (1979) suggested the importance of 'stepping stone' coral reefs as a means of increasing dispersal and immigration. However, protection by networks of reserves requires the description, understanding and protection of critical pathways between modules (Fairweather 1991a).

Zoning

A recurring concept in the literature is the zoning of an area using core and buffer zones (e.g. Ray 1976; Salm 1989). Core areas encompass entire ecological units (habitats and communities) as far as possible and buffer zones, next to and surrounding core areas, provide protection of core areas from external impacts. The function and size of these areas have not, however, been adequately defined. For instance, the size and position of a buffer zone and the activities allowed within it will determine the level of protection provided to the core area. Zoning will control human activities within boundaries but will not ensure protection of the enclosed area from other sources of threat.

Because of the three-dimensional nature of marine systems (i.e. sea surface, water column and benthos) it is possible for a variety of complementary activities to be accommodated in a marine protected area. Vertical zoning allows for the seabed and water column to have different restrictions placed on them. For example, the seabed itself may be absolutely protected but fishing and recreational activities may still be permitted on the surface and in the water column. Multiple use zoning of marine parks is a common and often effective strategy. Temporal zoning regulates activities when species may be particularly vulnerable. Such zoning could allow, for instance, commercial trawling in seagrass beds during any months when recruitment of fish and macroinvertebrates is expected to be slight.

EFFECTIVENESS OF PROTECTION: MONITORING AND EDUCATION

The aims of marine protection can be achieved through management strategies that restrict or prohibit certain human activities within some areas. Despite debate about the worth of marine preservation, few studies (table 1) have assessed the effect of protection on the abundance and diversity of species both within the reserve and in adjacent areas. As a result, little is known about the actual effects of protection and therefore their success. Nor has this information been conveyed to the public.

Monitoring

Marine protected areas provide sites for scientific research, monitoring and baseline studies. Research in these areas will not only provide increased information on marine ecosystems but also allow study of species in a protected, perhaps pristine, environment. Monitoring studies incorporating protected areas allow us to assess the effect on protected and adjacent areas of people, zoning strategies and management plans. In this sense they act as 'control' areas for the effects of human intrusion on other areas. These are essential to untangle induced impacts from temporal changes.

There are two types of published studies comparing protected and non-protected marine areas: monitoring studies of the distribution and abundance of whole assemblages; and studies assessing the effect of humans as predators on particular species in terms of abundance and size. Most published studies reviewed (table 1) claimed increased abundance and diversity of species within protected areas (Duran & Castilla 1989). From this it could be assumed that they have been successful in preserving species and communities. However, most of these monitoring programmes were carried out after the park was established (Castilla & Duran 1985; Duran & Castilla 1989) and were short-term studies (Duran & Castilla 1989; Shepherd 1991). Therefore, we cannot claim with confidence that the park itself was responsible for these results, nor that the patterns found will continue through time.

To test directly the effect of protection on marine resources, monitoring programmes must be established, within reserves and in adjacent waters, before the management restrictions are enforced. Such BACI (Before-After-Control-Impact) monitoring programmes are the minimum used to assess environmental impacts (Stewart-Oaten, Murdoch & Parker 1986). However, the type of impact for marine reserves would differ from the impact found in most environmental impact studies, presumably being beneficial rather than detrimental. This design requires that impact and control sites are monitored through time before and after the impact. Control sites should be chosen in habitats similar to the impact site to allow comparison of park and non-park areas. Only two of the studies reviewed used this BACI design (Castilla & Duran 1985; Duran & Castilla 1989).

Distribution and abundance surveys

Most studies of this type concentrated on the effect of fishing pressure on finfishes. All studies showed an increase in the number of species of fish within the protected area (see studies 1-3, 6, 19, 22 & 28-31 in table 1; Ballantine 1987; Craik 1989). In general, the increase in abundance and diversity of species in the reserve was caused by the presence of exploited species of fish, which were rare or absent outside. Fish were often larger in mean size and showed a greater size range within the reserve. Three studies documented changes in abundance and diversity of fish through time. Increased abundance of fish in the reserve was not always found (e.g. Samoilys 1988). Cole, Ayling and Creese (1990) found some species of fish increased in abundance through time at the Leigh reserve (New Zealand) whilst other species showed no consistent trends. Census of the Sumilon reserve (Philippines), after it was reopened to fishing pressure, showed dramatic decreases in abundance of fish and a drastic decline in community diversity (Russ & Alcala 1989; Alcala & Russ 1990). In contrast, census of coral trout on two reefs on the Great Barrier Reef, before and after closure, showed no increase in the size or abundance of fish in these areas (Gray & Craik 1986). However, comparison with non-reserve areas was not done and therefore the changes in the reserve can only be discussed relative to previous changes in the reserve.

Few of the studies reviewed used knowledge of the previous 'natural' variation in abundance of fish through time or details of the biology of the species. Because of variability, changes in abundance among fished and protected areas may not be separable. The longest and most intensive survey of the effects of reserve status on a fish species was a coral trout monitoring programme (GBRMPA 1979; Gray & Craik 1986). This survey has been running since 1981 in the Capricornia Section of the Great Barrier Reef to assess the effectiveness of zoning on trout populations and the health of the reef in these areas. However, as noted by the authors, this survey had several flaws realized in retrospect (Gray & Craik 1986; Craik 1989). There was no assessment of spatial variability of trout within the reef, survey sites within a reef were chosen for convenience rather than similarity, and there was no assessment made of habitat variability (Gray & Craik 1986; Craik 1989). In most studies, sites were chosen because of the similarity of habitat (i.e. macrobenthic communities and physical environment). Few studies assessed the complexity of the habitat in the areas censused, yet Buxton and Smale (1989) found that depth and substratum relief were important determinants of abundance of fish in protected and unprotected areas of reef, and Samoilys (1988) found the type of habitat to influence biomass of some fish species.

Most studies used a visual census technique to assess fish populations. Biases inherent in this technique have been discussed in detail (e.g. Gray & Craik 1986; Lincoln Smith 1988) and may lead to sampling errors, such as found in results of length estimation. A difference in the population structure between surveys carried out in two successive years was found to be due to a diver bias in length estimation of coral trout (GBRMPA 1979; Gray & Craik 1986).

Surprisingly few studies have examined the effect of protected areas on marine invertebrate populations. Those studies indicated that densities of many species are larger in reserves than in areas outside. Cole,

Table 1. Summary of monitoring studies comparing biota of protected areas with either non-protected areas or the same area before protection.

Study No.[a]	Study Year	Location	Habitat	Organism	Method	Div[b]	Abd	Size
1	1979	Australia	Coral	Fish	Census		+	+
2	1984	"	Seagrass	Fish	Tagging		+	
3	1986	"	Coral	Fish	Census		+	+
4	1990	"	Rock	Abalone	Census		+	+
5	1991	"	Rock	Abalone	Census/Expt	-	-	
6	1990	New Zealand	Rock	Fish, macro	Census	0	+	+
7	1984	Chile	Rock	Limpets, algae	Census		+/-	+
8	1985	"	Rock	Mussels, gastro	Census	+		
9	1986	"	Rock	Bivalves, gastro	Census	-	-/+	+
10	1986	"	Rock	Limpets	Census		+	+
11	1987	"	Rock	Gastro	Census		+	+
12	1989	"	Rock	Bivalves, gastro	Census	+	+	
13	1989	"	Rock	Limpets	Expt	+	+	-
14	1989	"	Rock	Kelp	Census		0	+
15	1990	"	Rock	Kelp	Census		-	+
16	1985	South Africa	Rock	Mussels	Census		+	+
17	1987	"	Rock	Bivalves, gastro	Census	-	+	+
18	1989	"	Rock	Mussels	Census		+	+
19	1989	"	Rock	Fish	Census		+	+
20	1991	"	Rock	Limpets	Census		+/-	+
21	1987	Kenya	Coral	Urchins	Census		-	+/-
22	1988	"	Coral	Fish	Census	+	0	+
23	1988	"	Coral	Fish/urchin/coral	Census		+	+
24	1989	"	Coral	Gastro	Census		0	
25	1989	"	Coral	Urchins	Census		-	
26	1990	"	Coral	Gastro	Census	+	0	
27	1990	"	Coral	Urchins, fish	Expt	+	+/-	
28	1983	France	Rock	Fish	Census	+	+	+
29	1985	Philippines	Coral	Fish	Census	+	+	
30	1989	"	Coral	Fish	Census	+	+	+
31	1990	"	Coral	Fish	Census		+	
32	1987	Costa Rica	Rock	Limpets	Census		+	+
33	1989	Florida	Seagrass	Lobsters	Tagging		+	

[a]*Studies:* 1 - GBRMPA 1979; 2 - Jones 1984; 3 - Gray & Craik 1986; 4 - Wells & Keesing 1990; 5 - Shepherd 1991; 6 - Cole, Ayling & Creese 1990; 7 - Moreno, Sutherland & Jara 1984; 8 - Castilla & Duran 1985; 9 - Moreno, Lunecke & Lepez 1986; 10 - Oliva & Castilla 1986; 11 - Duran, Castill & Oliva 1987; 12 - Duran & Castilla 1989; 13 - Godoy & Moreno 1989; 14 - Castilla & Bustamante 1989; 15 - Bustamante & Castilla 1990; 16 - Siegfried, Hockey & Crowe 1985; 17 - Hockey & Bosman 1986; 18 - Lasiak & Dye 1989; 19 - Buxton & Smale 1989; 20 - Lasiak 1991; 21 - Muthiga & McClanahan 1987; 22 - Samoilys 1988; 23 - McClanahan & Muthiga 1988; 24 - McClanahan 1989; 25 - McClanahan & Muthiga 1989; 26 - McClanahan 1990; 27 - McClanahan & Shafir 1990; 28 - Bell 1983; 29 - Russ 1985; 30 - Russ & Alcala 1989; 31 - Alcala & Russ 1990; 32 - Ortega 1987; 33 - Davis & Dodrill 1989.

[b]*Codes:* Div - species diversity; Abd - abundances; Size - size frequency distributions of selected species: effect of protection is either + (increase), - (decrease) or 0 (no change); gastro - gastropods; macro - macroinvertebrates; expt - experiments.

Ayling and Creese (1990) found higher densities of sea-urchins inside the reserve than outside. In contrast, rock lobsters were absent from sites surveyed outside the marine reserve, but large and plentiful inside. Klima, Roberts and Jones (1986) found increased densities of pink shrimp in the Tortugas Sanctuary (USA) compared with adjacent areas outside the reserve.

Studies of human subsistence predation

The other type of study assessed the direct effect of human collecting in intertidal communities. Harvesting by humans often results in removal of the top predator, which is important because significant changes have been caused in a community by altering predator-prey relationships (e.g. Paine 1980). These studies found large differences in community diversity (usually increased species richness at exploited sites) and abundance of individuals between harvested and non-harvested areas. Only three studies attempted to quantify harvesting pressure among the areas compared (Siegfried, Hockey & Crowe 1985; Duran, Castilla & Oliva 1987; Lasiak 1991).

Seven studies of human predation on invertebrates were made on rocky shores in Chile. Three of these studies (Castilla & Duran 1985; Moreno, Lunecke & Lepez 1986; Duran & Castilla 1989) monitored densities of *Concholepas concholepas*, a prized gastropod, in harvested and non-harvested areas. Castilla and Duran (1985) and Duran and Castilla (1989) monitored an area before and after declaration of a marine reserve and compared this with two areas that were continually being harvested. Over the five year period of the study, large differences in community structure developed between protected and exploited areas (Duran & Castilla 1989). In the reserve, densities of *C. concholepas* increased whilst the width of mussel beds decreased due to predation by *C. concholepas*. By the end of the study the intertidal area was dominated by barnacles inside the reserve, and by mussels in the harvested areas, outside the reserve. Sizes and densities of other shellfish were also different inside and outside the reserve (Duran & Castilla 1989).

Moreno, Lunecke & Lepez (1986) found similar results when monitoring the same species over several years in a different rocky intertidal marine reserve in Chile. Unlike Duran and Castilla (1989), the difference in abundance of the gastropod *C. concholepas* between the reserve and non-reserve only occurred in one year of the three year study and was due to decreases within the non-reserve area rather than increases inside the reserve. However, in the reserve area, there was a significant increase in sizes of gastropods and a reduction in abundances of three species preyed upon by *C. concholepas* compared with non-reserve areas. Thus, community structure and complexity within the reserve were altered. In South Africa, Siegfried, Hockey and Crowe (1985) found similar results of increased diversity and smaller sizes of exploited species of mussels in harvested reefs compared with protected reefs.

Another commonly harvested shellfish in Chile is a key-hole limpet. Where the limpets are harvested the growth rate and fecundity of another limpet, a competitor, was shown to increase due to increased macroalgal cover, their primary food source (Godoy & Moreno 1989).

Six studies assessed the effect of human predation in protected and unprotected reefs in Kenya. Diversity of gastropods was higher in protected reefs but densities were similar among protected and unprotected reefs (McClanahan 1989). However, abundances of gastropods were greater in lagoons of unprotected reefs, attributed to removal of their predators, fish. This result was also found for a sea-urchin (McClanahan & Muthiga 1989). Increased distribution range and abundance of the sea-urchin in unprotected reefs was attributed to removal of predators by fishing. McClanahan and Shafir (1990) showed experimentally that decreased predation of sea-urchins by fish was occurring in unprotected areas. They also compared differences in percentage cover of hard coral, substratum diversity and topographic complexity between protected and unprotected areas. Changes in abundance of herbivores (from fish in protected areas to urchins in non-protected areas) was suggested to explain these differences. However, this hypothesis was not tested experimentally.

Only two studies concerned harvested algae; both monitored the abundance, standing crop and size of the harvested kelp *Durvillea antarctica* in Chile (Castilla & Bustamante 1989; Bustamante & Castilla 1990). Castilla and Bustamante (1989) monitored kelp populations in three sites, one of which was protected as a reserve during the study. After seventeen months of protection, a greater abundance of *D. antarctica* was found in the reserve. Bustamante and Castilla (1990) extended the spatial scale of this survey to compare other non-harvested areas (offshore islands) with adjacent non-protected areas on the mainland. The standing crop and plant density of *D. antarctica* were twice as great on the offshore islands than on the mainland, although individual plants were smaller. Bustamante and Castilla (1990) suggested that these non-harvested islands may be acting as refuges for plants normally exploited by humans and as such provide a source of propagules for the adjacent exploited mainland.

Problems with monitoring studies

Several studies found that partial closure of a reef, as a reserve, to fishing pressure enhanced yield per recruit in the unprotected area (Gitschlag 1986; Ballantine 1987; Davis 1989; Davis & Dodrill 1989; Alcala & Russ 1990). It has been hypothesized that populations increase in the protected area and lead to localized shortages of some resources resulting in adult animals migrating to non-reserve areas or the protected area may act as a sanctuary for a species for only part of their life stage - for example, shrimp (Gitschlag 1986); lobster (Davis & Dodrill 1989). Therefore it may be important to monitor populations adjacent to reserve areas. Tagging studies of species in a reserve may aid such an investigation. Jones (1984) tagged fish in an estuarine reserve and revealed the importance of this small area to fisheries.

A similar result was shown by a tagging study on juvenile spiny lobsters in Florida (Davis & Dodrill 1989). The area acted as a sanctuary; lobsters used it as a nursery ground for less than three years, then moved onto coral reefs outside the reserve, thus enhancing fisheries there. No comparison with other unprotected areas was made, so the relative importance of these sanctuaries remains equivocal. However, the success of this area may be shown in the growth rates of juvenile lobsters, the fastest on record.

Tagging of organisms is fraught with problems and effects of a tag on the mobility, growth and survival of an organism are uncertain. Retrieval of tags also poses a problem because tags may go undetected or be lost, and fishers must be relied upon to report tags found. Gitschlag (1986) questioned the validity of data he collected in a tagging study on pink shrimp. Identification of movement patterns of shrimps out of the reserve relied on information supplied by fishers, which was often inaccurate and incomplete. In many tagging studies there is insufficient recapture of tagged organisms, for example Gray & Craik (1986) reported only a 2% recovery of tagged coral trout.

Pseudoreplicated comparisons (Hurlbert 1984) constituted a problem inherent in nearly all studies reviewed. Comparing more reserve and non-reserve areas would increase the validity of tests and therefore improve the basis upon which conclusions and recommendations are made. Some of these studies have highlighted a possible 'dilution effect' caused by the reserve, that is the reserve had less impact on marine resources at greater distances. Sampling control areas next to reserves and further away are necessary to test this hypothesis. Such monitoring programmes may be modelled on point source pollution studies (e.g. Fairweather 1990b) as the dilution effect evident in these studies is analogous. As was shown by Moreno, Lunecke and Lepez (1986), change in biological systems is often slow and therefore difficult to detect. Long-term monitoring strategies are therefore necessary.

Most studies suggest that reserves enhance the abundance and diversity of species, even in adjacent unprotected areas. However one study that monitored abalone populations over a 20 year period in a reserve established for that purpose, found that populations in the reserve declined drastically (Shepherd 1991). Intense removal of adult abalone in adjacent unprotected parts of the reef reduced the overall breeding stock resulting in poor recruitment of juveniles back into adult populations on the reef. This came about due to ignorance of the limited larval dispersal patterns of abalone (Shepherd 1991).

Comparisons of the density and richness of species found in reserve and non-reserve areas are essential not only to evaluate the effect of the reserve on these species but also to assess the success of management. Such information may then be used to guide designation of other reserves with similar restrictions. Despite their rarity so far, adequate monitoring programmes which are rigorous in biological and methodological aspects (e.g. Phillips, Keough & King 1993) are essential for the continued management and viability of marine reserves.

Role of education

Effective enforcement of regulations is necessary to achieve more than mere lip-service to the goal of marine conservation. Education about ongoing management is the only means of getting the public on side and being accountable for the considerable effort which must be expended in maintaining protected areas.

Therefore, any attempt at protection should coincide with an education campaign designed to inform the public of the management scheme and monitoring programme. It may also be possible to enlist the active aid of the public in policing an area.

CONCLUSIONS

Compared with terrestrial systems, proportionally less of the marine environment is protected, despite even less being known about the ecological processes that maintain it. Objectives and criteria used to select reserves are the same regardless of the system and the effectiveness of reserve design depends upon our knowledge of relevant ecological and physical processes. More precise criteria and objectives used to establish marine reserves are required. In particular, the degree of difference between marine and terrestrial environments means that different design strategies (i.e. sizes, minimum core areas, boundaries) are required for marine reserves.

The philosophy of physically discrete boundaries protecting entire ecosystems, species ranges or life histories, as is used in terrestrial parks, will not be appropriate to marine ecosystems. They may provide protection to species at a critical stage in their life history. Debate continues over the strategies for marine reserve design: size; shape; area; and zoning. These problems can be resolved through experiments and monitoring of reserve areas already established. Monitoring of species within reserves will also be essential to assess the effects of zoning and management strategies. It may be necessary to control activities outside the reserve area but which ultimately affect the core.

Finally, no matter how well marine parks are designed or how much legal protection is given, the success of these areas will be determined by the management implemented (Hatcher, Johannes & Robertson 1989; Salm 1989) and the public support ensuing from education campaigns.

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