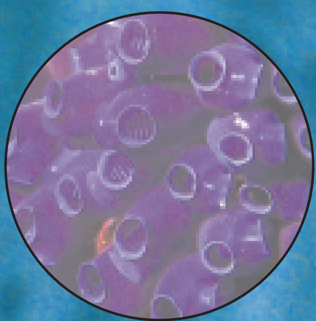
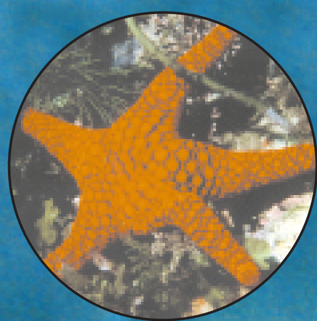
GUIDE
to
Marine Protected Areas

J.L. Baker B.Sc. M.Env.St.

April 2000





Department for Environment and Heritage

Government of South Australia

DEPARTMENT FOR
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Guide to
Marine Protected Areas

Prepared in April 2000 by

J. L. Baker B.Sc. M.Env.St.

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South Australia

Government
of South Australia

Guide to Marine Protected Areas

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ABBREVIATIONS

AGSO	Australian Geological Survey Organisation
ANZECC	Australian and New Zealand Environment and Conservation Council
GRMP	Great Barrier Reef Marine Park
GBRMPA	Great Barrier Reef Marine Park Authority
IMCRA	Interim Marine and Coastal Regionalisation for Australia
IUCN	International Union for the Conservation of Nature
MOMPA	Multiple Objective Marine Protected Area
NRSMPA	National Representative System of Marine Protected Areas
SSB	spawning stock biomass
TFMPA	Task Force on Marine Protected Areas
UNEP	United Nations Environment Program
UNESCO	United Nations Educational, Scientific and Cultural Organisation

1 INTRODUCTION

The list of current and future threats to the integrity, functions, and numerous values of coastal marine environments throughout the world, is familiar to those with even a passing interest in the sea. Some of the most cited examples include:

- decline in water quality through domestic, industrial and agricultural discharges and inappropriate coastal marine activities and developments
- overfishing of marine resources, with consequent declines in species abundance and distribution, and changes to ecosystem structure and function
- physical damage to benthic marine ecosystems through poorly managed dredging, mining, trawling, and coastal marine developments
- exacerbation of coastal and subtidal erosion due to poorly managed coastal and marine activities and developments
- introduction of exotic species, with consequent declines in water quality, native species replacement, proliferation of diseases and toxic organisms, and damage to benthic habitat structure and function.

Many of these impacts are not visible or well publicised, and may therefore continue without public pressure for their cessation until critical levels of ecosystem change and environmental damage occur. These problems are exacerbated by:

- the growing number of coastal marine users and uses
- the increasing conflicts between existing and proposed economic developments, marine environmental protection measures, and resource management objectives
- recent political and economic trends in some countries that have reduced funding and human resources for conservation, with consequent inconsistencies in political commitment to marine conservation, monitoring and management
- the lack of co-ordination between national, State and local government agencies and their jurisdictions
- incomplete data and inadequate understanding of the impacts of uncontrolled multiple use on coastal marine environments.

Well-managed marine protected areas (MPAs) are now widely considered to be one of the most effective methods for protecting marine environments and their component biodiversity, sustaining productivity of marine resources, and managing multiple uses in coastal marine environments. Although it is recognised that the principles of conservation and ecologically sustainable use must extend beyond the bounds of protected areas (IUCN 1991, cited by Kenchington 1993), MPAs are recognised as one of the most important and practical approaches to ecosystem management and sustainable resource use.

MPAs can also help to overcome the problems associated with a multitude of single-sector, single-use or single-resource management decisions that have been responsible for many of the problems outlined above.

MPAs can be designated for one or many overlapping and related reasons, including:

- *ecological purposes*—conserving representative examples of biogeographic regions and ecosystems, protecting critical habitat types, conserving biodiversity 'hotspots', maintaining genetic diversity and protecting rare or threatened species or habitats

- *social and economic purposes*—managing and enhancing fisheries, conserving and managing areas that are important for recreation / tourism, education or research
- *cultural purposes*—protecting sites for traditional Aboriginal use, conserving historic shipwreck sites and other cultural features
- *other reasons*—protecting aesthetic values and pristine ‘wilderness’ areas, maintaining undamaged marine ecosystems, and conserving marine environments for the future (ie inter-generational equity).

This report is provided as a general guide for those who are involved with the process of MPA planning, establishment and management, including coastal marine environmental managers and fisheries managers, scientists, non-government organisations, industry representatives, conservationists, educators, and others. Chapter 2 defines MPAs, and discusses the numerous marine protected area classifications that have arisen during the past 30 years. Examples of increasingly common and successful types of MPAs are provided in Chapter 3, such as multiple-use marine parks and fisheries reserves. The newer trend towards establishing MPAs for biodiversity protection is also critically reviewed.

All State governments in Australia have endorsed the development of a national representative system of marine protected areas (NRSMPAs) as part of the National Strategy for Ecologically Sustainable Development (Muldoon and Gilles 1995). Chapter 4 discusses the history of MPA declaration in Australia, the background programs, such as the Commonwealth’s Ocean Rescue 2000, that contributed to the current strategies of the National Representative System, and an overview of each State's progress towards meeting the NRSMPA objectives. Current national policies that can influence the implementation of the NRSMPA are also critically examined in Chapter 4.

In Chapter 5, the history of MPAs in South Australia is reviewed, gaps in the existing system of MPAs are discussed, and an overview of the long-term framework for establishing a representative system is provided. Threats to the quality and biodiversity of South Australia’s near-shore ecosystems are also reviewed in Chapter 5, because these concerns are related to the planning and management of the NRSMPA in South Australia.

Chapters 3, 4 and 5 also provide international, national and South Australian examples of existing MPAs that have successfully served a number of purposes. These purposes include fisheries sustainability and enhancement; protection of ecosystems; conservation of rare or endangered species; biodiversity conservation *per se*; and provision of opportunities for sustainable tourism and recreation, education and research.

In Chapter 6, the many different approaches to MPA design are considered. This includes discussion of:

- the methods that have been devised, both internationally and nationally, for classifying and mapping coastal marine biogeographic regions and ecosystems, with the aim of representing examples of those classified areas in MPA systems
- the increasing acceptance of an ecosystems-based network approach to MPA design
- the data needs and design criteria for establishing MPAs that assist fisheries sustainability and replenishment
- newer scientific methods of designing MPAs to satisfy numerous competing objectives
- miscellaneous approaches to MPA design.

Also mentioned in Chapter 6 are the importance of socio-economic criteria in MPA selection, and the role of community participation in the selection process. Potential problems with the various approaches to MPA design and selection are also discussed, such as proportional representation of areas. International and national examples of user group conflicts in MPA establishment, and potential solutions, are detailed in Chapter 7. The various aspects of MPA planning, legislation, management, and ongoing monitoring of user group impacts are examined in Chapter 8. Much has changed during the past decade in all of these aspects of MPA establishment, particularly the increased public participation in MPA management and monitoring programs.

Clearly, the use of MPAs in coastal marine conservation and management will be less effective if coastal marine ecosystems and resources are continually degraded and depleted. Therefore, the important roles of ecosystem management, adaptive management practices, and the adoption of the precautionary principle must be considered, and these requirements are discussed in Chapter 9.

Finally, the report provides examples of the ways in which some previously established MPAs have failed to meet their objectives. Such reasons include design flaws; inadequate protection of MPAs from pollution; unawareness of the effect of oceanographic variables on MPA function; breaches of management regulations, and socio-political compromises to MPA planning and establishment. Knowledge of such pitfalls to MPA success can assist in the planning, siting and management of new MPAs.

2 DEFINITION, CLASSIFICATION AND TYPES OF MPAS

2.1 DEFINITION OF MPAs

Numerous definitions of marine protected areas (MPAs) exist, and most refer to:

> '*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.*' (IUCN 1988, cited by Zann 1995b).

The definition that has been adopted by the Australian Commonwealth in the *Guidelines for Establishing a National Representative System of Marine Protected Areas* (ANZECC TFMPA 1998) is that of the International Union for the Conservation of Nature (IUCN):

> '*An area of land and/or sea specially dedicated to the protection and maintenance of biological diversity, and of natural and associated cultural resources, and managed through legal or other effective means.*' (IUCN 1994).

MPAs range from small, highly protected areas in which no extractive uses are permitted, to large, multiple-use areas that are often zoned to provide different levels of protection, and permit various activities and resource usage. Examples of large multiple-use MPAs include the Great Barrier Reef Marine Park in Queensland and Ningaloo Marine Park in Western Australia. MPAs in Australia range in size from 0.1 square kilometres (Shiprock Aquatic Reserve in New South Wales) to 344 000 square kilometres (the Great Barrier Reef Marine Park) (GBRMPA 1995).

2.2 CLASSIFICATION OF MPAs

'Marine Protected Area' is an internationally-recognised, all-encompassing term, that includes more than 90 different names for types of reserved areas (Silva *et al* 1986, cited by Kenchington 1993 and McNeill 1994). The broad term 'MPA' is confusing, because not all 'marine protected areas' protect the substrate, waters or biota within the designated area, due to physical damage, chemical pollution and/or unmanaged extraction of resources. Some types of MPA terms currently in use include those in the following table. Note that categories of MPAs are not standardised between states or countries, and thus a single category, such as 'park', 'sanctuary' or 'reserve' can refer to MPAs of vastly different sizes, levels of protection, and permitted activities.

Some of the different terms in Table 1 (below) describe the same type of MPA. For example, a 'marine fishery reserve' and a 'fish sanctuary' may be described, operated and managed in the same way, but have different names according to the state or country in which they were designated.

Although there is currently no international standard for MPA classification according to size, 'marine parks' generally refer to large, multiple use areas (from hundreds to thousands of square kilometres) with specified zoning that may include 'no-take' areas. 'Marine reserves' refer to smaller areas, ranging in size from hundreds of square metres to tens of square kilometres. Confusingly, not all countries adopt these meanings for parks and reserves, with highly-protected marine parks in part of Africa, for example, not permitting multiple uses, whilst marine reserves permit fishing activities (McClanahan and Kaunda-Arana 1996).

Table 1: Examples of various terms used to describe MPAs around the world, from 1975–1999.

Parks	Sanctuaries	Reserves	Other names/types of MPA
marine park marine national park	national marine sanctuary marine sanctuary fish sanctuary wildlife sanctuary	marine biosphere reserve nature reserve nature preserve marine nature reserve national nature reserve aquatic reserve marine reserve marine fishery reserve marine ecological reserve replenishment reserve ‘no-take’ reserve fish habitat reserve integral reserve ecological reserve bio-reserve	protected seascape wilderness area wildlife conservation area national wildlife refuge conservation area natural monument marine national monument habitat/species management area marine managed area managed resource protected area fish habitat protection area fishery preserved zone no fishing zone harvest refuge/refugium hunting refuge

Note: see Ivanovici (1984) for a list of MPA types used in Australia

The differences between countries and states in the definition of a single MPA term is illustrated by ‘sanctuaries’, which, according to various definitions, can range in size as follows:

- large, multiple-use areas covering thousands of square kilometres (such as the Florida Keys and Monterey Bay National Marine Sanctuaries in North America)
- highly-protected MPAs that have the objective of protecting ‘all species’, according to a South African definition of marine sanctuaries (Hockey and Branch 1997)
- small, ‘no-take’ MPAs for habitat protection and fisheries management (such as the fish sanctuaries in the estuaries of North Queensland, Australia)
- intertidal MPAs for wading bird conservation (Southport Lagoon Sanctuary Conservation Area in Tasmania, Australia).

Sanctuaries in Australia are not zoned, but sanctuaries in North America can incorporate ‘no-take’ ‘ecological reserves’ within the bounds of the larger sanctuary area (eg Western Sambos Ecological Reserve within the Florida Keys National Marine Sanctuary).

There is a clear need for a standard terminology for MPAs, and this has been recommended several times during the past decade in Australia (Ivanovici *et al* 1993; McNeill 1994).

3 MPAS IN AN INTERNATIONAL CONTEXT

3.1 BACKGROUND

MPAs have been recognised as important for marine conservation and management since the early 1960s (Kriwoken and Haward 1991, cited by McNeill 1994). MPAs are now considered by major international conservation organisations—World Wild Fund for Nature (WWF), International Union for Conservation of Nature (IUCN), United Nations Environment Program (UNEP)—to be a very effective means of protecting marine biodiversity, resources, and the ecosystems of which they are part (Norse 1993).

Ray (1975, 1976) was the first to provide a set of objectives for MPAs, which included habitat and species preservation; conservation of genetic resources; research; recreation/tourism; education; aesthetics; cultural purposes; special uses; and multiple uses. Most countries ignored the recommendations to establish MPAs at the time. During the same decade, increasing pressures on marine environments at all scales prompted the IUCN to conduct a series of international workshops, which culminated in the production of international guidelines for the declaration of marine and coastal protected areas (Salm and Clark 1984). By 1987, increasing pressures from physical and chemical pollution, over-harvesting, conflicting uses and habitat destruction resulted in the development of a policy framework for marine conservation, devised by the IUCN General Assembly and the World Wilderness Congress (Kelleher 1991). The IUCN called for cooperation between the public and all levels of government to develop national systems of MPAs. The framework established by this process included:

- dividing the world's marine regions into biogeographic zones
- identifying gaps in MPA representation within those zones
- identifying potential sites for MPAs, to fill the gaps (Kelleher 1991).

The resultant sets of guidelines (Kelleher and Kenchington 1991; IUCN 1994; Kelleher *et al* 1995) form the basis for much MPA planning at international and national levels today.

The IUCN international standard of categories for MPA classification (IUCN 1994) is based upon levels of protection and various uses of MPAs. Internationally recognised selection criteria devised by the IUCN for MPA establishment include naturalness, biogeographic importance, ecological importance, economic importance, social importance, scientific importance, international or national significance, and practicality/feasibility. These original categories have been slightly modified since that time, and a list of currently used selection criteria is outlined in Section 4.3.1

Well over 1000 MPAs have been established around the world during the past 20 years (Sobel 1993). Apart from declaration of MPAs by individual countries on a case-by-case basis, several different international initiatives have been used to establish MPAs. These include UNEP's Regional Seas Program; UNESCO's Man and the Biosphere Program; UNESCO's Marine Science Program; the South Pacific Regional Environment Program, and initiatives of the Food and Agriculture Organisation of the United Nations, the International Maritime Organisation, and the International Whaling Commission (Kelleher and Kenchington 1991).

The development of MPAs has now become an important international issue. In North America, for example, the Vice President urged Congress in September 1999 to approve the Administration's Land Legacy Initiative, to allocate US$29 million for existing marine sanctuaries, another US$25 million to acquire and protect critical fish habitats, and US$10.3

million to protect and restore reefs. Expanding the current ‘no-take’ (ie non-extractive) proportion of MPA area will be a major part of this program. This initiative may have come about in response to the continued decline in habitat quality and resources in North America’s large multiple-use marine sanctuaries, which have not previously incorporated ‘no-take’ areas or pollution control as key parts of their MPA policy. The use of MPAs as a marine management tool is also gaining impetus in some European countries. Spain, for example, which has used MPAs for marine conservation and management since 1982, has legislation that specifically provides for their creation, and 11 new MPAs have been proposed during the past decade, to provide a more even distribution of habitat protection (Ramos-Espla and McNeill 1994).

In some countries, the cynical term ‘paper parks’ does apply to MPAs, because they are seemingly not protected or managed in a sustainable way. MPAs in some parts of Canada, for example, purportedly do not prohibit or limit the adverse effects of ballast water dumping, pollution outfalls, herbicide spraying near watersheds, fish farming, introduction of marine pests, unmanaged shellfish harvesting, spearfishing, bottom trawling, or industrial or residential development near or in the MPAs. Almost all of the MPAs in British Columbia have no ‘no-take’ zones, and it is reported that less than 1% of coastal waters in that region is fully protected (Georgia Strait Alliance 1998, unpublished).

The following sections (3.2.1 and 3.2.2) discuss two major types of MPAs:

- Marine Parks and Biosphere Reserves, which are types of multiple-use MPAs that have been recommended by marine planning and management experts as a way of integrating conservation with commercial and recreational interests
- MPAs for fisheries management, which have been used in many countries to protect and manage marine resources and activities during the past 20 years. MPAs for fisheries can be incorporated into a multiple-use marine park system.

Section 3.3 discusses MPAs in the context of biodiversity protection, which has been one of the most widely debated concepts in marine environmental conservation and management during the past five years. Section 3.4 provides international examples of MPAs that meet specific objectives, such as fisheries management, conservation of endangered species, and ecosystem protection, among other objectives.

3.2 MAJOR EXAMPLES OF MPA TYPES

3.2.1 Marine biosphere reserves and multiple-use marine parks: two names for the same?

Biosphere Reserves are a type of multiple-use MPA that was developed in terrestrial systems to assist conservation and management of whole ecosystems, whilst also allowing for sustainable use of resources, research and education activities. The IUCN Guidelines for MPA establishment (Kelleher and Kenchington 1991) suggest that the biosphere reserve concept is appropriate for MPA planning and designation. Internationally, there has been recent interest in biosphere reserves and their application to MPAs (eg Agardy 1994).

Biosphere reserves essentially comprise three components (see below). The association of conservation with carefully planned and managed development is a key element of biosphere reserves. There is similarity between biosphere reserves and multiple-use MPAs, if those areas include:

- ‘core’ high-level conservation zones (‘reserves’ or ‘sanctuaries’ in the strict sense)

- managed 'buffer' zones (in which particular, limited activities are permitted, if they don't conflict with the objective to protect the core areas)
- 'general use' or 'transition' zones (in which various managed uses and activities are permitted, if those activities do not adversely affect the core protected areas).

IUCN's (1995) policy for MPAs in Australia also recommended the concept of buffer zones (in which activities are closely managed) to protect 'core areas' from impacts. This is usually possible only in large MPAs such as marine parks, in which areas that are large enough to 'buffer' or 'dilute' impacts can be designated.

The 'core', highly protected areas in a biosphere reserve should be an example of an ecosystem type that is typical of a biogeographic region. This part of a biosphere reserve is designated to protect naturalness, biodiversity (including genetic diversity), viable species populations, and is selected for its perceived effectiveness as a conservation unit. Activities in the core are limited to those that will not adversely affect the functioning of the area in any way.

Ideally, biosphere reserves encourage interdisciplinary research programs on ecosystem functioning, biodiversity, and the impacts of pollution on the structure and function of the biosphere. The provision for development in the transition zone of a biosphere reserve enables monitoring of the effects of uses and activities in the biosphere, and can assist the development of sustainable production for impacted and degraded areas. Biosphere reserves are not intended to replace other methods of biodiversity conservation, but are seen as complementary efforts (Kelleher and Kenchington 1991). Kenchington and Agardy (1990) and Agardy (1994) discuss in detail the applicability of biosphere reserves to marine conservation.

Internationally, marine biosphere reserves currently exist, but unless these multiple-use MPAs are managed according to principles of conservation and sustainability, they are biosphere reserves in name only. The Central California Biosphere Reserve (McArdle 1997) is an example of a biosphere reserve that is not managed according to the principles espoused by Kenchington and Agardy, and could thus be considered as a 'paper park'.

Although the use of biosphere reserves has not been officially embraced by MPA planners in Australia, zoning systems in some of Australia's large, multiple-use MPAs are similar to those of biosphere reserves.

These include the Great Barrier Reef Marine Park (GBRMP), which has 120 'core' (highly protected) areas linked by continuous 'buffer' (limited use) and 'transition' (general use) zones covering the entire 250,000 square kilometres. However, the 'core' areas currently total only 2% of the entire GBRMP (Rigney 1990), although GBRMP authorities are planning to increase this percentage in the 2000s.

Similar to the GBRMP, Marine Parks in Western Australia also have:

- core protected areas (Sanctuary Zones) to protect (for example) significant ecosystems, sensitive habitats, biodiversity or threatened species
- buffer zones (such as the 'Recreation Zones'), in which limited activities are permitted
- 'General Use Zones', in which a variety of activities are permitted.

Such zoning is consistent with the national directive to establish a system of multiple-use Marine Protected Areas. Regardless of the terminology (Biosphere Reserves or Multiple-Use Marine Parks), such areas can be incorporated into a National Representative System of

MPAs, using the IUCN's Protected Area categories to organise zones and permitted activities. This is discussed in Section 4.

The establishment of large, multiple-use MPAs, in which a variety of zoned activities can co-exist, may not necessarily conflict with conservation objectives. For example, a Marine Park has recently been proposed for the ecologically, socially and economically valuable Port Philip Heads region in Victoria, Australia (Environment Conservation Council 1998). Similar to the zoning of a Biosphere Reserve or a multiple-use Marine Park of the type established in Queensland and Western Australia, the proposed park would include 'no take' sanctuary zones and 'special nature sites' within a multiple-use matrix that includes fishing and shipping zones.

Similarly in South Australia, the recently declared Great Australian Bight Marine Park contains a sanctuary area at the head of the Bight for Southern Right Whale breeding and calving. Activities in this area that would compromise the value of the site for whale protection are not permitted (Government of South Australia 1998).

3.2.2 MPAs for fisheries management: a global success story

> *'Everything else has failed. Marine Protected Areas are the last resort for fisheries management.'*
>
> (Eduade Houde, Chair of North America's National Research Council, cited by Molyneaux 1999.)

Although MPAs designated specifically for fisheries management meet a limited number of objectives regarding the establishment of a *biogeographically representative* system of MPAs (see Section 4), their value for fisheries is supported by overwhelming evidence, and they are perhaps the best known examples of successful MPAs. Fisheries MPAs (often called *fisheries reserves*, or *'no-take' reserves*), can be incorporated as 'no-take' zones into large, multiple-use MPA networks, or can be used as 'stand-alone' fisheries management and site-management measures in areas without a multiple-use network of managed zones.

MPAs for fisheries management aim to:

- protect critical life stages of fished species and their associated habitats
- sustain and enhance fisheries for those species through 'export' of adult fish, juveniles, larvae and / or eggs into surrounding fished areas.

The numerous benefits of fisheries MPAs have been extolled in dozens of papers and reports during the past two decades—Alcala 1981 and 1988; Davis 1989; Plan Development Team 1990; Bohnsack 1993 and 1994; Dugan and Davis 1993; Buxton 1993; Ballantine 1987 and 1989; Roberts and Polunin 1993, Rowley 1994, Baker *et al* 1996; Ballantine 1996, Ruckelshaus 1997, Allison *et al* 1998, CALM 1998. Leading international fisheries scientists (eg Caddy 1999), who have worked through the successes and failures of fisheries management of the past two decades, now advocate MPAs as a major strategy for fisheries management in the 2000s.

If fishing regulations were successful in all regions, for all fished species, and if there were no uncertainty about the reaction of stocks to both fishing pressure and environmental variables, then 'no-take' MPAs would not be necessary. Experience has shown that traditional fisheries management regulations are often unsuccessful in sustaining stock levels, especially if the resilience of stocks is further reduced by oceanographically driven events such as a succession of poor spawning events. Even limitations on the catch of declining species can fail to protect the stocks if, for example, either adults or juveniles are caught as by-catch, or critical habitats for those species are degraded. MPAs are becoming necessary as a supplementary measure to assist fisheries sustainability.

In Australia, it has been suggested (Kearney *et al* 1996) that fisheries MPAs will become more commonly used as a management tool, as fisheries are increasingly placed in broader marine ecosystem management frameworks. MPAs are a 'pro-active' approach to sustainable fisheries management, because they can prevent overexploitation of fished species, provide an insurance mechanism for fisheries (especially when stock recruitment and persistence levels are unpredictable), and also protect habitats prior to critical levels of degradation. This is important because species replenishment (including brood stock transplants) following decline, and restoration of degraded habitats, are more costly and difficult to achieve than sustainable resource use in the first place.

Some researchers have recently concluded that 'no-take' MPAs for fisheries management can result in the same fish harvest levels for industry as can be produced from traditional management measures (Hastings and Botsford 1999), and that MPAs are the preferred management approach for reef fishes and invertebrates.

Fisheries MPAs are especially recommended for:

- large reef fish species that are vulnerable to over-exploitation due to their life history characteristics
- juveniles of coastal fish species that use coastal nursery areas
- species populations composing benthic adults with either locally or widely dispersed larvae (such as lobsters and crabs)
- molluscs, urchins and other 'site-attached' invertebrates.

Specific examples in which fisheries MPAs have improved abundance, size, reproduction, recruitment and/or fishery yields of various fisheries species from around the world are provided by Roberts and Polunin (1991), Fairweather and McNeill (1992), Dugan and Davis (1993), Baker *et al* (1996), Kripke and Fujita (1999), and in Tables 6, 7, 8 and 9 in this report.

A summary of the benefits of fisheries MPAs follows.

Protection of critical habitats

These include seagrass beds and reefs, which are important for many fish and invertebrate population activities such as spawning, aggregating, feeding, and growing in sheltered sites away from predators. Critical habitat may be defined as 'spawning grounds and nursery, rearing, food supply and migration areas upon which fish depend directly or indirectly in order to carry out their life processes.' (Canadian Fisheries Act 4, cited by Department of Fisheries and Oceans, 1986). Two of the most significant ecosystem types that provide critical habitat for fish in southern Australia are discussed in Appendix 2.

Maintenance of critical habitat sizes

The amount of suitable habitat for fish species close to the coast is gradually diminishing as coastal development and pollution increase. MPAs can be used to maintain (or even relatively increase) the portion of suitable habitat that can be occupied by fish, and thus guard against what metapopulation dynamics theorists often refer to as the 'minimum habitat threshold' (see Lande 1988), below which populations become increasingly prone to decline, or even local-scale extinctions.

Protection of adult spawning stock and spawning grounds

These supply eggs, larvae and 'new recruits' to fished areas. Fisheries biologists have long asserted that a threshold size for many fish species populations exists, below which recruitment success is critically affected. This is difficult to demonstrate in marine systems because population sizes and densities are difficult to determine, and recruitment success is also strongly influenced by density-independent oceanographic factors. The number of examples of critically low population sizes and spawning levels affecting fisheries sustainability and recruitment success is increasing, however, and a few of many examples include prawns (Caputi 1993), small pelagic fish (Beverton 1990), and various abalone populations around the world (Shepherd *et al* 1991; Shepherd *et al* 1995; Shepherd and Partington 1995; Shepherd and Baker 1996). Based upon the general decline of spawning stocks and recruitment in some European, American and Canadian fisheries that have been over-harvested for a long period, there is evidence to suggest that long-term recruitment success is partly density-dependent. Over space and time, there are critical adult spawning stock sizes below which recruitment potential will be jeopardised (and these effects will be exacerbated during 'natural' periods of environmentally-regulated poor recruitment). Maintaining a critical stock size, known as 'spawning stock biomass' (SSB), and potential level of population egg production protects the stock against decline resulting from what fisheries biologists refer to as 'recruitment over-fishing', a state in which the spawning stock of fish is harvested faster than they can produce young 'recruits' to the fishery. Maintaining SSB is now one of the cornerstones of fisheries management. There is much fish population modelling work to support the aforementioned spawning stock-recruitment relation (Sissenwine and Shepherd 1987; Gabriel *et al* 1989; Mace 1991 and 1994; Mace and Sissenwine, 1993)[1]. MPAs can help to protect so-called 'mega-spawners', the older age classes of some fish populations, which make significant contributions to population egg production, and thus help maintain the reproductive potential and population viability of heavily fished stocks. Even though knowledge of critical levels of SSB may be incomplete, MPAs can at least be used as an 'insurance measure' against stock declines, by maintaining and relatively increasing the numbers of (i) older, larger individuals which maintain spawning potential, and (ii) younger fish, who will become the 'mega-spawners' of the future. Russ (1987), Plan Development Team (1990) and Roberts and Polunin (1991) all contend that MPAs may reduce the risk of recruitment over-fishing by maintaining SSB. Polacheck's (1990) population modelling results have shown that MPAs can substantially increase the spawning stock biomass realised from a cohort of fish, especially heavily harvested stocks. Tables 9 lists results of studies conducted in MPAs that apparently protected SSB, and/or provided eggs, larvae or recruits to surrounding areas, thus 'replenishing' parts of the population outside the MPA.

Protection of vulnerable life stages (eg juveniles)

Coastal MPAs are an important management strategy for the juvenile life stage of many species of fish and invertebrates. Juveniles of many coastal species settle out of the plankton in response to a combination of oceanographic fluxes, physical and biological cues and stimuli (Boehlert and Mundy 1988), that are associated with near-shore coastal ecosystems, such as seagrass estuaries, mangrove–saltmarsh complexes, and coastal reefs. Appendix 2

[1] The need to protect as much spawning stock as possible becomes even more important when we consider that (i) the average batch of eggs can suffer 99.999% mortality before recruiting, and a mere 0.1% change would mean the difference between 0 and 1000 survivors (Collie 1988); and (ii) in some cases only *part* of the entire spawning stock may be producing eggs at any one time that eventually result in recruitment to the fishery, depending upon age structure and location of the spawning populations, among other factors (Caputi 1993).

discusses the dependence of juvenile fish and crustaceans upon some of these critical ecosystem types. Because coastal environments are often subject to physical damage and pollutants from land-based sources, juvenile coastal fish can be vulnerable due to their proximity. Protection of juveniles and their associated habitats is important to fisheries sustainability, and a direct link between juvenile abundance and subsequent catches of adults has sometimes been demonstrated (eg Caputi 1993).

Restocking fished areas with young and/or adult fish

One of the contentions that is hard to prove concerns the movement of adult fish out of MPAs into the adjacent fishery to enhance yields with the increased size and abundance of the fish moving into the fishery. This is variously referred to as the ‘spillover effect’, ‘biomass export’, ‘emigration’, or ‘fisheries replenishment’. Evidence from MPAs in several countries (Table 10, and other references in Chapter 3) is showing that this fisheries replenishment effect does occur, and helps to sustain local fisheries, particularly those areas in which stocks are severely depleted. Local fisheries can be strongly enhanced by MPAs. For example, fishers who ‘fish the boundary’ are often well rewarded with higher catches than prior to the MPA.

Increasing size/age range and biomass of fish

Table 6 in Appendix 1 lists studies in which size range and biomass of species were shown to be greater inside MPAs compared with similar but fished habitats outside the reserves. The general increase in fish size reported for many MPAs over periods ranging from several years to two decades (see Table 6) often indicates that fish in the MPA are living longer than their fished counterparts, and thus age structure is being maintained in the population. This increase in size range and biomass can benefit the fishery, through protection of spawning potential (see above), ‘emigration’ of adults out of the MPA, and/or through movement of eggs, larvae or juveniles to the surrounding fished areas.

Maintaining the natural population age structure

MPAs can prevent selective over-harvesting of largest fish, which can damage the reproductive potential of stocks and also reduce the genetic diversity of the population. Maintaining age structure is related to the need to maintain spawning stock biomass, as discussed above. For example, according to Jones *et al* (1990), one 20 year-old adult snapper (*Pagrus auratus*), which is a major commercial and recreational fish species in southern Australia, can produce around five million eggs, more than 10 times higher than the egg production of a young (four year-old) adult snapper[2].

Increased abundance and/or density of fish

Table 8 in Appendix 1 shows that abundance and/or density of fished species commonly increase (i) inside MPAs, compared with adjacent fished areas, and often (ii) in the fished area adjacent to the MPA (or even considerably distant from the MPA according to the mobility of the fish, as shown in Table 10, and in the sections of this report on *Examples of MPAs for Fisheries Management and Enhancement*).

Protection of populations with poor reproductive potential

MPAs can provide protection for populations that:

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[2] Similar results from New Zealand showed that a 25 cm snapper produces 80 000–200 000 eggs, and a 50 cm older snapper produces between 4.5 million and six million eggs (Crossland 1977).

- are spatially isolated (geographically separated), may be at the end of the optimum range for that species, and are subject to poor larval replenishment levels and/or infrequent migration of adult fish into the area
- have restricted ranges
- have naturally low fecundity or fertility.

Small, reproductively ‘isolated’ populations are particularly vulnerable to decline: they may not be replenished by larvae from other parts of the populations, and thus use their own ‘internal dynamics’ to maintain themselves. These types of populations cannot persist easily when faced with factors that reduce population size or reproduction potential, and MPAs can be used to protect them. MPAs can also protect populations in which, depending upon their size and shape, survivorship, fecundity, or fertilisation success decrease as critical densities of spawning adults are reduced by natural environmental processes or by fishing—the ‘Allee effect’ (Allee *et al* 1949, cited by Shepherd and Brown 1993; Quinn *et al* 1993).

Protection of ‘site-attached’ populations

MPAs can protect populations of species that utilise natural and artificial reefs. ‘Site-attachment’ to reefs makes some fish populations particularly vulnerable to over-fishing. Many reef areas, even those in remote areas, are no longer refuges for fish, due to improved fishing knowledge, increased numbers of anglers, and advanced location-finding and fishing devices (Bohnsack 1993; Jones and Luscombe 1993a and 1993b).

Protection of large predatory reef fish with ‘vulnerable’ population dynamics

MPAs have been effective in protecting the spawning stock biomass and size/age structure of reef fish whose population dynamics make them highly vulnerable to over-fishing, such as snappers, grunts and groupers. These have low natural death rates, grow to large sizes, are long lived, have highly fecund older adults, aggregate on reefs to spawn, and are ‘site-attached’ to reefs for much of their adult life. The removal of the highly fecund, large (old) adults by commercial, sports and recreational fishing has endangered many populations of these types of reef fish, and MPAs are considered one of the only effective ways to manage such populations, and sustain the fisheries through larval ‘export’ from the MPAs (Huntsman *et al* 1997, and references in Section 3.4.1).

Maintenance (or recovery) of genetic diversity

MPAs protect genetic diversity in populations that are vulnerable to changes in population dynamics due to fishing. This is important for heavily fished stocks; for example, those in which all the largest individuals are targeted. Heavily fished stocks have selection pressure upon them to mature earlier, and such fish may also display a smaller adult size, shorter life span, altered behaviour, and be of unnaturally low densities (Ricker 1981; Roberts and Polunin 1991; Bohnsack 1993). Although the effect of fishing upon genetic variation is poorly studied, heavy fishing removes more of a population every year than would die of natural causes. This change to the natural balance of life and death means that fish population dynamics must over time ‘adapt’, both ecologically and genetically, to the rapid changes in population numbers. MPAs help to maintain a portion of the stock and also to increase the numbers of juvenile fish, and thus reduce the likelihood of the entire stock evolving into a less desirable form due to a fishing-induced reduction in genetic diversity.

Maintenance of ecosystem dynamics

By providing refuge for at least a portion of a harvested stock, MPAs can guard against the probable consequence of severely altering the structure and dynamics of marine communities and ecosystems, caused by persistent harvesting of 'key' marine species (Bohnsack 1993). Because fishing has traditionally been driven by economic factors, government regulations, or 'common property' notions, fishers often have limited opportunity or no incentive to 'switch species' during times when the abundance of target species is low. Therefore, fishing is often not ecologically balanced, and continued harvesting of declining species can cause, for example, 'snowball effects' up or down food chains, altering the balance in predator-prey cycles of abundance, and completely changing the species composition of fish species in an area (Estes *et al* 1989 and Dayton *et al* 1995, cited by Allison *et al* 1998). The result of serial depletion of major stocks is often a shift in ecosystem dynamics. For example, disturbing the balance between numbers of predators and prey can lead to significant changes in habitat structure, due to feeding patterns of large numbers of the small 'prey species', which are often herbivorous. A country in which fishing declines are evident typically moves from fishing long-lived species to short-lived species, mature individuals to immature individuals, and from higher trophic members (eg large predatory fish) to lower trophic members (eg small planktivorous schooling fish). Done and Riechelt (1996) stated that such ecosystem concepts are rarely considered by managers, scientists and fishers. Fisheries MPAs can lessen the extent to which these ecosystem shifts occur.

Protection of biodiversity and habitat quality within the MPA

Habitat protection using MPAs benefits not only the life processes of fished species, but also contributes to marine biodiversity conservation and natural restoration of damaged habitats. For example, trawling has demonstrated adverse impacts on benthic species composition and the condition of benthic habitat (Collie *et al* 1997; Prena *et al* 1999). Prohibition of trawling in fisheries MPAs would therefore assist the recovery of biodiversity and habitat structure in such areas.

'Stock insurance' against natural or fishing-induced changes in population dynamics

MPAs can protect a portion of the stock from detrimental influences in surrounding areas, and can thus provide some insurance against 'collapse' of an entire stock (Bohnsack 1993; Roberts 1997; Lauck *et al* 1999). This is most important if (i) habitats outside the MPA which provide vital resources for fish are physically degraded and / or chemically polluted; (ii) fish outside the MPA are over-harvested, and population numbers outside the MPA decline to critically low levels; and / or (iii) traditional fisheries management measures are not successful, despite the best available stock assessments and scientific advice to managers. If a harvested stock *does* suffer 'unnaturally' rapid changes in abundance, biomass or spawning, MPAs can be used, according to Bohnsack (1993) and other MPA advocates, to help rebuild a stock at a faster rate than would be possible without the MPA. MPAs also help to maintain populations faced with natural changes over time in species distribution, abundance or recruitment. For example, maintaining age structure and size structure is believed to help populations persist and adapt when faced with environmental changes (Carr and Reed 1993). Maintaining age structure is particularly important to sustain populations during periods of poor recruitment, and also maintains the proportion of older (and usually highly fecund) animals that help replenish the population (see above).

Providing insurance against management failures

This is particularly desirable when traditional management controls such as quotas, legal minimum sizes and bag / trip / boat limits fail in the long term to adequately protect stock levels from decline. Ludwig *et al* (1993); FAO (1994); and Guenette *et al* (1998) provide examples and reviews of fisheries collapses that have occurred around the world, despite the best scientific information and ongoing management strategies.

Simplify fisheries management controls, and reduce needs for management data

MPAs have been shown to be a successful tool for managing fisheries, even in the absence of data traditionally used for management, such as estimates of stock size, growth and mortality rates and reproductive success. Some MPAs, particularly those that are well managed and have a high level of community support, can also reduce the need for complicated enforcement controls (bag / trip / boat limits, fish measuring, monitoring of fishing equipment, quota regulation, etc).

Sites for fisheries science/population dynamics research

MPAs can provide abundant, large and approachable organisms, and undisturbed experimental areas (Bohnsack 1993; and see references by Ballantine). Unbiased scientific studies of fish behaviour, natural mortality, population growth rate, trophic and social interactions, and other dynamics can sometimes be useful for fisheries management, and cannot be conducted easily or objectively in fished areas (Shepherd 1991; Bohnsack 1993). MPAs can provide a 'standard' for monitoring population parameters and dynamics, against which the effects of fishing and habitat degradation can be gauged. Fisheries–independent data from undisturbed areas (ie MPAs) also improve fisheries population management models (Bohnsack 1996). Fisheries MPAs provide areas for ecosystems research, such as predator-prey relationships, in which predators (large carnivorous fish, for example) have not been removed.

Social benefits

Because fisheries MPAs protect habitats and their component species, many marine interest groups in society benefit. Direct benefits include increased value of fisheries yields; increased marine recreation / tourism and education opportunities; and increased conservation value of marine sites. Indirect benefits include 'existence value' (ie psychological value of knowing that undamaged coastal marine ecosystems and un-exploited portions of marine populations still exist) and 'bequest value' (the value of conserving marine habitats and resources for future generations, which is one of the principles of Ecologically Sustainable Development). Social value has been demonstrated by many MPAs, including the Leigh Reserve in New Zealand (Ballantine 1987 and 1989; Davis 1989), central American MPAs, and island MPAs in the Philippines (McManus 1988; White 1989), where tourism, recreation and involvement of local communities in fisheries management all increased after the MPAs were established. In some countries, local communities assist in promoting, managing and monitoring fisheries MPAs. Such 'sustainable stewardship' can significantly benefit those communities. For example, apart from the economic benefits, many communities pride themselves in their locally abundant resources.

Specific examples of MPAs that were successful in achieving one or more of these aims are provided later in this report, in the sections on international and national examples of *MPAs for Fisheries Management and Enhancement* (Section 3.4.1).

3.3 MPAs FOR BIODIVERSITY PROTECTION: A NEW PARADIGM, OR A REPACKAGING OF EXISTING MPAs?

Maintaining coastal marine biotic diversity, in the face of increasing development, habitat alteration, and over-use of resources, has recently attracted the attention of major international conservation bodies (World Conservation Union, WWF, UNEP, Centre for

Marine Conservation). MPAs have been recommended at both international (Norse 1993; IUCN 1994) and national levels (Ray and McCormick-Ray 1992; HORSERA 1993; ANZECC TFMPA 1998 and 1999) as one of the most effective means of conserving marine biodiversity.

Although many different definitions of biodiversity exist (see Gaston 1996), marine biodiversity can be defined for conservation purposes as:

> *'….variation in the different kinds of biotic entities at a number of levels (such as species or functional groups), their relative frequency (eg abundance) over defined spatial scales (eg ecosystems), and their relative 'difference' between each other (measured in terms of dissimilarity in taxal composition, structure or function.'* (Baker 2001, in prep.).

Despite the worldwide establishment of hundreds of MPAs during the past two decades, Norse (1993) considered that design of MPAs is still in an 'experimental' stage for biodiversity conservation. The world's MPA system has developed without sufficient knowledge of biodiversity in many areas. Norton and Ulanowicz (1992, cited by Heywood 1994) suggested that this ignorance could be overcome by adopting a hierarchical approach to biodiversity assessment and conservation.

Norse (1993) stated that all types of MPAs, ranging from large, multiple-use reserves to small, highly protected areas, could be used for biodiversity conservation, and his sentiments are echoed by administrators across the world who are keen to present existing and proposed MPAs as beacons of biodiversity conservation. If such MPAs were sized and spaced to include representative examples of each ecosystem and biotic assemblage in each biogeographical region, then the objective of MPA-based biodiversity conservation might be closer to achievement. However, it is unlikely that the diversity of ecosystems and biota within each region is being protected by the current MPA systems in most countries, including Australia. There is a concern that existing MPAs which, in most countries, do not adequately represent biodiversity might be 'repackaged' under national or international guidelines to establish a more 'representative' MPA network, and presented as sufficient for biodiversity protection.

A major problem with 'representing' and conserving biodiversity in a MPA system stems from inadequate knowledge of what biodiversity actually exists in marine environments. The extent to which both existing and proposed MPAs are adequate representatives for biodiversity protection is scale-dependent, and depends on the definition of 'biodiversity' that is used, and the type of biodiversity that is measured. There are obviously many biodiversity indicators to choose from, ranging from the impossible to quantify, to inadequate surrogates, and including:

- 'all' species
- rare or vulnerable or endangered species
- 'keystone' species that perform major ecological functions
- socially important ('flagship') species
- environmentally sensitive indicator species
- major representatives of higher taxa
- functional groups
- major structural groups
- communities / assemblages
- ecosystem types.

To date, the 'biodiversity problem' has not been fully assessed in any country. Questions such as the following should be answered in relation to existing MPAs and new proposals:

- *What elements of marine biodiversity are being protected?*
- *Is biodiversity being protected at local, regional or national scales?*
- *What evidence is required to determine whether the goal of biodiversity protection is being met?*

Obviously, MPAs that protect habitat and do not permit activities that physically or chemically damage the MPA, and do not permit extraction of resources, contribute to the protection of biodiversity in some way, at a specified scale.

There has been a somewhat subjective call for priority protection to be given to 'hot-spot' areas of high species diversity, high degree of endemism, and high productivity (Norse 1993). However, protecting representative examples of biodiversity in all ecosystem types would include areas that do not fit those three criteria. Apart from the fact that collation of complete species-level inventories for 'hot-spot' identification is infeasible, such a limited approach to biodiversity conservation that focuses on species richness 'hot-spots' may prove inadequate in the long term for biodiversity protection at local, regional and national scales. In coastal marine ecosystems, 'hot-spots' of species diversity are often relatively small, attractive areas that are well-known by divers/tourists, and their representation in MPAs is quite high compared to other ecologically important sites. Regarding areas of high endemism, this implies good knowledge of the systematics and distributions of marine groups. With the exception of marine mammals, fish, specific groups of macro-crustacea and macro-molluscs, and marine plants, it is likely that few other marine groups are sufficiently known taxonomically to assess their degree of endemism.

Another approach to MPA-based biodiversity protection uses 'biogeographic zones' as surrogates for biodiversity conservation. A 'bioregional' approach was recommended for Australia's MPA system (Ray and McCormick-Ray 1992), and has been adopted by the Commonwealth as a framework for MPA establishment in the 2000s (ANZECC TFMPA 1998). The 'bioregional' approach to MPA identification and designation is discussed in Section 4.

As with the 'hot-spot' approach to biodiversity conservation, the use of biogeographic regions alone may not be adequate. Designating 'representative examples' of biogeographic regions in a MPA network may be considered too large a scale for adequate biodiversity protection. For example, representing one MPA in each bioregion, or a proportion of area within one bioregion, would not adequately represent all of the ecosystems/habitat types within that bioregion, nor the biodiversity contained in those ecosystems. Further, the adherence to a simplistic 'broad-scale' approach may fail to recognise ecologically and functionally important variations in biodiversity, both within and between the bioregions that are defined. Other concerns with the bioregional approach to biodiversity conservation are discussed in Chapters 4 and 6.

It could be argued that quantifying coastal marine biodiversity at several hierarchical scales is an essential first step, prior to designing a MPA network of 'priority areas' for effective biodiversity conservation. The opposite approach could also be argued, namely that a MPA network for biodiversity protection must be established as soon as possible, despite inadequate knowledge, because biodiversity quantification is a long, expensive process, and coastal marine environments will continue to be degraded in the forthcoming years whilst biodiversity research work is undertaken.

As a way of integrating the small scale ('hot-spot') and large scale ('bioregion') approaches to biodiversity protection, Norton and Ulanowicz (1992) recommended a hierarchical, multi-scaled approach, emphasising biodiversity conservation at the level of ecosystem and landscape processes, which support the dynamics of smaller scale biodiversity. Each country has a different approach to biodiversity protection using MPAs, and a detailed analysis of this topic is beyond the scope of this report.

Section 4.3.5 discusses the use of MPAs in Australia for biodiversity protection, and Sections 3.4.3 and 4.4.3 provide examples of international and national MPAs that have protected biodiversity within limited contexts, such as protection of critical habitats and ecosystems, local species diversity, and rare / vulnerable / endangered species.

3.4 INTERNATIONAL EXAMPLES OF SUCCESSFUL MPA ESTABLISHMENT

3.4.1 MPAs for fisheries management and enhancement

With the exception of New Zealand, most examples of successful MPAs for fisheries management and enhancement during the past decade have come from tropical countries, as shown in the tables below. However, increasing evidence of MPA benefits is emerging for temperate areas, such as North America, South Africa, Australia and Western Europe.

Although some of the earliest, most informed and impassioned arguments for the creation of fisheries MPAs came from North America (eg Davis 1981 and 1989; Plan Development Team 1990; Bohnsack 1993 and 1994), that country has not traditionally advocated 'no-take' MPAs in practice. Most of the MPAs in the USA permit various types of fishing activity. Notable exceptions include a small number of MPAs established during the 1980s and 1990s for protection of snapper, rockfish and lingcod stocks (Kripke and Fujita 1999), and temporary MPAs for the replenishment of abalone, blue crab, rock lobster and scallop stocks (Bohnsack 1982; Davis 1989; Debenham 1999; Molyneaux 1999). Only recently have 'no-take' areas been widely considered as a major tool in fisheries management.

Renewed interest in the use of MPAs (Roberts 1997; Allison *et al* 1998; Lauck *et al* 1998) has coincided with a national directive in North America for increased protection of declining stocks, in light of the apparent failure of traditional fisheries management techniques. The reauthorisation of the *Magnuson-Stevens Fisheries Conservation and Management Act* in 1996 has provided for the establishment of 'no-take' MPAs to assist fisheries management in North America. On both the east and west coasts of North America, MPAs are now being seen by policy makers, managers and scientists as the 'last option' (and possibly the only effective option) to manage reef fish species that have seriously declined in size and abundance from over-fishing. Fisheries Management Councils are particularly interested in using 'no-take' MPAs to protect spawning grounds, and for reefs where 'site-attached' fish reside, such as snappers, grouper, porgy, and drum fish (Molyneaux 1999; Debenham 1999).

In California, the recent Assembly Bill (*Marine Life Protection Act*) that was passed in 1999 requires the Department of Fish and Game to prepare action plans for the siting of new MPAs for fisheries management. Similar bills and national working groups on MPAs are developing all over North America and Canada. These legislative changes during the late 1990s have resulted in another spurt of scientific writing on the benefits of MPAs. For example, 20 fisheries scientists and ecologists recently produced a joint article, summarising the benefits of fisheries MPAs and discussing potential design criteria (Murray *et al* 1999).

On the other side of the world, the European Commission has formally endorsed a new Communication on Fisheries Management and Conservation. This provides for the establishment of the Natura 2000 Network (for conservation of critical marine habitat), as well as conservation areas known as 'boxes' (which might include fisheries MPAs for depleted species) (http://europa.eu.int/comm/dg14/info/info62_en.htm—Europa On-Line Information Page(1999)).

At a global level, MPAs have been especially valuable for population replenishment of:

- site-attached reef fish such as snappers, groupers, grunts, wrasses and others, which can be eliminated from some reefs by over-harvesting of the older, highly fecund adult fish (Alcala 1988, Roberts and Polunin 1993; Roberts 1995; Wantiez *et al* 1995; Beet and Frielander 1999)
- invertebrate species which have limited movement, but whose larvae are wide-ranging, such as lobsters (Ballantine 1989 and 1991; Roberts and Polunin 1993), or locally dispersed, such as some mollusc species.

The benefits of MPAs for fisheries management and stock protection and enhancement are discussed in Section 3.2.2. Tables 7 and 8, in Appendix 1, outline the benefits of fisheries MPAs for increasing size range, biomass, abundance and/or densities of harvested species. It is becoming increasingly accepted among fisheries scientists and managers that maintaining spawning stock size, biomass and age structure can improve fisheries, by sustaining long-term recruitment potential in heavily fished stocks (see Section 3.2.2). Table 9, in Appendix 1, provides examples of MPAs that are assumed to have improved fisheries by protecting spawning stock biomass (SSB) and spawning events. Table 10, in Appendix 1, provides examples of MPAs that are either known or assumed to have 'exported' larvae, juveniles or adult fish to adjacent fished areas, or fisheries in the region.

Apart from the empirical evidence presented above, an increasing number of mathematical models of fish population dynamics (both traditional yield-per-recruit and more developed simulation models), have shown that MPAs are a useful management strategy for exploited fish stocks. Such models are demonstrating that MPAs, especially those for heavily fished reef fish and 'sedentary' invertebrate populations, can:

- sustain spawning stock biomass
- maintain the critical densities of adults required for reproductive success
- maintain size and age structure
- provide a source of recruits to over-fished areas; and/or increase fisheries yields outside of the MPA (eg Quinn *et al* 1993, Man *et al* 1995, Holland and Brazee 1996, Guenette and Pitcher 1999, and comprehensive review by Guennette *et al* 1998).

MPAs are considered to be a less successful management option for highly mobile species with limited 'site-attachment' (de Martini 1993), such as long-ranging migratory species.

The numerous examples of MPA success listed in Tables 7, 8, 9 and 10 provide some indication that fisheries MPAs cannot be discounted as a useful fisheries management option. Although fishers usually object to the designation of a 'no-take MPA', this method of fisheries management has long been used (and accepted) in another guise, namely the 'closed area' to replenish overexploited stocks. One example is the menhaden fishery on the east coast of North America, in which 40% of the fishing area was closed following collapse. Landings have reportedly remained stable since the closure, and the fishery is now considered to be 'relatively healthy' (Bohnsack and Ault 1996, cited by Bohnsack, 1999, unpublished comment to Californian Marine Protected Area Network).

De facto MPAs for fisheries include areas that are not designated as MPAs, but are inaccessible to fishers. Such areas are becoming rarer due to improved fishing technology—GPS, coloured echo sounders, faster boats, rock-hopper trawl gear, larger fishing nets, improved dive gear, etc. One example of a *de facto* MPA is the 'depth refuge' for abalone in California, where abalone are found at a site that is too deep and rough for divers to visit. This depth refuge is considered to have sustained what is considered to be the only currently viable abalone fishery in California, since the other fisheries have been depleted by a combination of over-fishing and coastal pollution, according to Tegner (1993) (Bohnsack 1999, unpublished comment to California Marine Protected Areas Network).

Despite the apparent global success of many MPAs for fisheries management, there is still debate amongst some scientists and managers, and most fishers, about the usefulness of MPAs, including those MPAs for which benefits have been documented. One example of the debate over the success of a temporary 'no-take MPA' (ie an area without official status as a MPA, but serving the same purpose) is the closure of almost one third of Georges Bank in Maine USA in 1994. Four years after the closure, the decimated haddock levels reportedly increased to such an extent that the permissible catch per trip for fishers was raised by three times the initial weight (Bohnsack, pers. comm., cited by Molyneaux 1999). The closure of part of Georges Bank to protect groundfish stocks appeared also to have benefited the scallop fishery: an incremental decline had led to the collapse of the fishery in the early 1990s, but after the closure, sea scallop biomass increased in a short time to the 'original levels'. Twenty months after the MPA was declared, (i) average densities of adult scallops within the closed areas were about three times higher than in open areas, and (ii) abundance and biomass of both adult and juvenile scallops were almost as high as occurred in the 'halcyon' fishing days during the 1970s. Some researchers concluded that that area closure (ie MPA) is thus a viable option for increasing spawning stock biomass (Lai and Rogo 1998). However, an alternative view is that scallop biomass would obviously increase with any reduction in fishing effort (ie without the need for MPAs), and also that the increased densities in and around the MPA could result from a year of naturally strong recruitment, irrespective of the MPA (Kenchington 1999, pers. comm. to Californian Marine Protected Areas Network).

Such debate about MPA effectiveness is common, and will continue for any MPA from which evidence is equivocal, particularly for new MPAs from which potential benefits are unlikely to accrue for several years, and those MPAs which are not regularly monitored. Rarely are the fisheries' benefits of MPAs examined over a long period, both before and after MPA establishment. Perhaps the strongest empirical evidence for the role of MPAs in improving fisheries comes form the long-running MPAs in the Philippines. Several studies over 10 years showed increased biomass and numbers of fish, both inside and outside the MPAs (see table above), which sustained local fisheries. Russ and Alcala (1996a) reported that fishers of large predatory coral reef fish were unanimous that their yields of fish had increased since the MPA at Apo Island was established, and that fish were indeed 'exported' from the MPA to the surrounding fishery. The researchers' conclusions were based upon detailed, long-term monitoring of the MPAs and surrounding fished areas. When MPA protection broke down, densities of snapper and emperor fish declined by 94% in the fished area outside the MPA, catches declined by 50% (McManus 1988; Russ and Alcala 1996b), and the improvements in biomass and density that had accrued during five to nine years of MPA operation were eliminated in one to two years by unregulated fishing.

Similar to the Philippines MPAs, studies in MPAs in Kenya and Belize have shown that fish populations generally increase in the closed areas, and that fishers in adjacent areas start to get larger catches after MPA establishment (Wells 1998). Table 9 in Appendix 1 lists examples of MPAs that have been demonstrated to improve the catches and/or sustainability of fisheries outside the MPAs.

3.4.2 MPAs for ecosystem/habitat protection

Protected areas can be used to ensure that ecosystems of major significance, such as estuaries, seagrass beds, algal forests and coral reefs, are not physically damaged. The ecological significance of coral reefs has been well documented during the past two decades, and will not be reiterated here. The ecological significance of two major temperate ecosystem types, seagrass beds and macro-algal dominated reefs, is presented in Appendix 2.

MPA management that prohibits bottom trawling, boat anchorages in areas without moorings, speedboats and jetskis, and trampling by divers and other tourists (all of which can either scour and/or gradually erode shallow benthic environments) contribute to ecosystem protection in some way. Examples of such MPAs are numerous, and many of those are listed in other sections of this report, because those MPAs served other valuable purposes in addition to ecosystem protection.

The proportion of marine ecosystem types per region that are protected by MPAs has never been quantified on a global scale. However, most examples of high protection MPAs in individual countries (that were often originally designated because of their attractiveness for diving, or their importance to fisheries) serve an ecosystem protection purpose.

It has been suggested that fish species diversity is maintained by MPAs, and that without this form of protection, ecological 'shifts' in species composition can occur; when large carnivorous fishes are removed from the ecosystem by over-fishing, other smaller species increase in abundance. Letourneur (1996) provided some evidence for this ecosystem shift in a study of species composition within and outside of a MPA in the Western Indian Ocean. The 17 marine reserves in the Mediterranean (Goni *et al* 1998) are also considered to serve a similar ecosystem protection purpose, because changes in community structure at all trophic levels have been noted, through species interactions, following protection of the exploited fish species by MPAs.

The no-take zones in the Florida Keys National Marine Sanctuary are helping to restore ecosystem function in the food chain of the Florida Keys (Haskell 2000). Increases in large predatory fish species, and concomitant declines in small herbivorous fish species have recently been observed in the no-take areas, indicating that the dominant predatory fish in the system are now starting to 'rebound' after fishing-induced declines.

3.4.3 MPAs for biodiversity conservation

There is probably no country in the world that has an adequate system of MPAs to represent all biogeographic regions and ecosystem types, and their component biodiversity. The problems associated with assessing the effectiveness of the biodiversity conservation goal of MPAs are outlined in Section 3.3.

Clearly, MPAs that do not permit extractive uses, and also protect ecosystems from physical damage and chemical pollution, offer some protection to marine biodiversity. For example, a field study by Collie *et al* (1997) showed that sites which are undisturbed by trawling have higher biomass, species abundance, and species diversity than disturbed sites. Dayton *et al* (1995) provided a global review of the significant adverse impacts of trawling on biodiversity, regarding both benthic impacts and fisheries by-catch issues.

Recently, Prena *et al* (1999) showed that otter trawling can cause significant decreases in the abundance and diversity of benthic fauna such as crabs, basket stars, sea urchins, sand dollars, brittle stars and soft corals. Presumably, areas that are protected from trawling (as MPAs) might eventually start to recover their biodiversity.

There appear to be no published examples in which the biodiversity of all major taxonomic groups has been assessed before and after MPA establishment. Biodiversity protection in MPAs is usually an implicit assumption, and one that has not been tested. Almost all of the world's MPAs were not specifically designated for biodiversity protection, but many achieve this aim to some extent, in a *de facto* way.

One of the few examples of MPAs that purportedly have biodiversity protection as the major goal is the Far East State Marine Reserve in Russia, which allegedly has 'the most species diversity of marine animals in Russia' (Zhirmunsky 1994). It is recognised, however, that a single MPA obviously cannot represent examples of the biodiversity of the entire Eastern Seas region, and a system of new marine 'aquatories' (MPAs) has been devised.

Some MPAs that were designated for fisheries enhancement have increased species diversity. Bell (1983) demonstrated increased fish species diversity in a north-western Mediterranean MPA compared with fished areas outside the MPA. The additional species that were found only in the MPA included those that were vulnerable to spearfishing, recreational angling or commercial gill netting. Increased diversity of tropical reef fish species was also recorded in the Philippines reserves, at Sumilon and Apo, during the period in which they were protected (White 1989). Francour (1992) also reported that the number of fish species in a Corsican MPA was almost twice as high as in the fished areas, and were more evenly distributed in abundance. More recently, Jennings *et al* (1995) reported increased diversity in protected and lightly fished areas for families of fish containing target species (Lutjanidae, Lethrinidae). Another study by Jennings (Jennings *et al* 1996) showed that the diversity of nine families (and 115 species) of reef fish was significantly higher in protected Seychelles MPAs compared with fished areas. Wantiez *et al* (1995) studied the diversity of numerous species from nine families of reef fish, before and after establishment of MPAs in New Caledonia, and recorded a 67% increase in fish species diversity in the MPAs compared with the currently fished areas outside the MPA. Letourneur (1996) also reported differences in diversity (including species composition) between MPAs and adjacent fished areas.

At an international level, major centres of marine biodiversity ('The Global 200 Ecoregions') are being identified by the WWF, in an effort to ensure that the MPAs that are designated by each country in future will be more representative of the biodiversity in those countries (Well 1998). The goal of the World Wildlife Fund and the IUCN to establish a comprehensive global network of ecologically representative MPAs (Wells 1998), is considered to be one way in which each nation can meet its requirements to protect biodiversity under the International Convention on Biological Diversity (1994).

3.4.4 MPAs for rare/vulnerable/endangered species conservation

There are several examples of MPAs being designated to protect marine mammals, particularly whales, dolphins, sea lions and other 'charismatic' species. The Stellwagen Bank Marine Sanctuary in Massachusetts Bay, for example, was designated in 1993 to protect an area that is considered critical to the feeding and/or migration paths of humpback, minke, right and fin whales. The value of the sanctuary area for whales was previously threatened by applications for sand and gravel mining (Eldredge 1993). At the time of declaration, there were still concerns about future physical and chemical pollution from sediment dumping and sewage discharge.

More than a dozen MPAs in South Africa have been established, and all of them prohibit the killing or harming in any way of whales, dolphins and great white sharks. Great white sharks are a protected species in South Africa.

MPAs also provide the opportunity for rare fish species to build up their populations, including those that have become ‘site-rare’ due to severe depletion by fishing. Bohnsack (1993) has shown that a reef fish MPA in Florida increased the sightings of rare species that had purportedly been virtually eliminated by fishing. Roberts and Polunin (1993) also report for a Caribbean MFR, increases in large predatory species that had been seriously depleted by fishing, and were thus ‘rare’ at local scales prior to MPA establishment.

Numerous MPAs in the world may protect particular rare or threatened biota, but since this is usually not the primary reason for MPA establishment, many of those examples are not documented.

3.4.5 MPAs for tourism/recreation, education and/or research

Most of North America’s large marine sanctuaries, especially those in the southern part of the country, have demonstrated major benefits for tourism/recreation, education and research. The Channel Islands Marine Sanctuary, for example, is particularly active in both research and education activities (NOAA 1997). These include such activities as:

- compilation of marine flora and fauna inventories for the Biosphere Reserve program
- algal bloom and pollution plume research and monitoring
- mapping of historic shipwrecks
- marine benthic habitat mapping
- ecosystem surveys (investigating the linkages between krill, seabirds and marine mammals)
- large-scale national fish surveys (‘The Great American Fish Count’) and seabird counts
- marine litter surveys
- volunteer-based and school-based research and education programs, and public expeditions and ‘underwater workshops’.

Recreationally, the Channel Islands are very popular; over 100,000 SCUBA divers visit the Channel Islands every year (Crosby 1994). Sailing is also a popular activity, with 100 designated boat anchorages. The large Florida Keys National Marine Sanctuary also has a major tourist industry, and a marine education program. The Gulf of Farallones National Marine Sanctuary has an ‘experiential’ education program that fosters stewardship of resources, and aims to educate students about the need for MPAs, and their role in biodiversity conservation and ecosystem ‘health’ (Brown and Brown 1997).

Most tropical MPAs have high recreational usage, such as the coral reef MPAs in Egypt, Central America, and South East Asia, among many others. The Bonaire Marine Park in the Caribbean is one of the first self-funding MPAs, supported entirely from tourist revenues (which also contribute to half of that country's total gross domestic product) (WRI 1997). Tourism, recreation and involvement of local communities in fisheries management all increased following the designation of MPAs for reef species replenishment in the Philippines (McManus 1988; White 1989).

Some MPAs have also assisted marine research, particularly fisheries research. Examples include the Philippines MPAs at Sumilon and Apo (see references by Alcala and Russ), which have been the subjects of fish population studies and fisheries dynamics for more than two decades, and a reserve for spiny lobsters in Florida (Davis and Dodrill 1989, cited by Fairweather and McNeill 1992).

4 MPAS IN AUSTRALIA

4.1 BACKGROUND

Up until the 1960s, MPA selection in Australia is considered to have been *ad hoc*, based upon assumptions about the importance of scenic areas (eg for diving and tourism), or individual perceptions of which biota and habitats were the most threatened or endangered (Bridgewater and Ivanovici 1993). During the 1960s and 1970s, there was a rise of environmental awareness due to increasing marine pollution, coastal development and habitat destruction. MPA development emphasised the need to protect resources (especially fish stocks) and their associated habitats; to reserve more areas for endangered, popular or 'unique' species; and to conserve unusual / uncommon habitat types that were threatened. At that time, the Australian National Parks and Wildlife Service commissioned a report on policy development for marine reserves in Australia (Rooney *et al* 1978). The report described in detail the role of MPAs in (i) the preservation of processes, habitats, and species; (ii) the management of commercial stocks; (iii) the provision of dedicated areas for recreation, education, scientific research; and (iv) the protection of sites of archaeological and historical interest. The report also detailed the potential threats and user group conflicts involved with MPA establishment.

In 1985, the Commonwealth Council of Nature Conservation Ministers (CONCOM) endorsed the IUCN's criteria (see Section 3.1) for MPA establishment (CONCOM 1985). The CONCOM report emphasised the need for:

- a nationally coordinated approach to MPA declarations, involving inter-agency co-operation
- a national education and information program, to increase public awareness of the importance of MPAs
- development of cost-effective methods of surveillance and monitoring in MPAs.

Interestingly, CONCOM recommended a bioregional approach (termed 'geographic zoning' in the 1985 report) to MPA classification, a strategy that was later adopted and modified by CONCOM's successor (ANZECC) and its technical working group (IMCRA), using a more comprehensive biogeographic classification scheme (IMCRA Technical Group 1998). The IMCRA biogeographic classification is used at a Commonwealth level as the basis for national MPA planning in its present state, and is discussed in Section 4.3.2. Table 2 outlines many of the criteria that have been used to select MPAs in Australia during the past 20 years. Such criteria include the representation of 'unique' or otherwise significant marine ecosystem types, the conservation of rare / vulnerable / endangered species, and the creation of reserves for the protection and enhancement of fisheries (often single, economically significant species).

Table 2: Examples of criteria used to establish MPAs in Australia prior to 1998.

Reason for Declaration	Examples	Comments
To protect ***representative examples*** of marine and estuarine habitat and ecosystem types.	Fitzgerald River (WA) Coorong (SA) Hinchinbrook Channel (Qld).	Areas may be representative of *major ecosystem types* found within a *biogeographic region* (eg mangrove-saltmarsh ecosystem; seagrass ecosystem, calcareous reef ecosystem), or be representative of the biota characteristic of an entire biogeographic region.
To protect ***unique example****s* of marine and estuarine habitats and ecosystem types, or areas containing unique species.	Nypa Palms (Qld) Hamelin Pool Marine Nature Reserve (WA) Port Davey/Bathurst Harbour (Tas) Ashmore Reef National Nature Reserve (NT).	Such MPAs include areas declared to protect patches of *biogeographically important* habitat (eg southernmost stands of tropical mangrove species), or ecosystem types which are unique within a bioregion (and may contain a unique assemblage of flora and/or fauna).
To protect ***natural*** or ***wilderness*** areas.	Macquarie Island (Australian Antarctic).	Includes areas that have not been subject to any human-induced change. MPAs are declared to protect such areas from future interference.
To conserve ***rare/endangered/ vulnerable*** species and their habitats.	Seal Beach/Seal Bay (SA) Great Australian Bight Marine Park (SA) Ningaloo Marine Park (WA) Sea Elephant River Wildlife Sanctuary (Tas).	Such MPAs are often declared due to public concern for the perpetuity of a socially valuable species (eg whales, sea lions, dugongs, whale sharks, wading birds).
To protect and enhance fish and other exploited populations to assist ***fisheries management.***	Blanche Harbour-Douglas Bank Aquatic Reserves (SA) Swan Bay Marine Reserve (Vic) German Bar and Maaroom Fish Habitat Reserves (Qld).	MPAs designated to protect spawning, feeding, and/or juvenile fish nursery areas, providing for increased size, numbers and densities of fish; and/or increased spawning potential and/or larval production. Note that most MPAs for fisheries management in Australia have been designated to protect nursery areas containing juvenile fish. Other life stages and their associated habitats are have not usually been considered for protection using MPAs.
To protect areas and features of ***social significance.***	Ship Rock and North Harbour Aquatic Reserves (NSW) Port Campbell (Vic).	Such reserves include attractive diving and other sites of significance for *recreation and tourism* (often with high aesthetic value).
To protect areas and features of ***cultural significance.***	Casuarina Coastal Reserve (NT) Japanese Submarine (NT) Zanoni Shipwreck (SA) Oyster Cove (Tas).	Includes areas of significance to Aborigines for traditional usage, indigenous cultural values, or areas having native title considerations. Include shipwrecks, historical and archaeological sites.

Reason for Declaration	Examples	Comments
To protect areas or features of ***scientific importance.***	West Island (SA) North Creek (NSW) Crayfish Point (Tas).	Includes sites of long-term population monitoring studies; significant areas for studies which support resource management; and sites of scientific interest due to rarity of habitat or community types. Some temporary reserves of scientific importance may eventually become permanently designated to assist resource management (eg rock lobster sanctuaries in SA are now permanently designated to assist fisheries management, but were originally designated for scientific study of rock lobster populations).
To protect areas of ***educational significance.***	Goose Island Aquatic Reserve (SA) Long Reef Aquatic Reserve (NSW).	Used to educate students and the general public about marine conservation, marine biology, ecology and biodiversity.
To provide physically undisturbed ***buffer zones*** (sinks) for catchment runoff or other pollutants.	Barr Creek and Half Moon Creek Wetland Reserves (Qld).	Such reserves are usually subject to contamination and degradation from runoff, sewage discharge and/or proximal development.
For reasons of ***practicality and/or accessibility.***	Wilsons Promontory Marine Park (Victoria).	Includes areas which are largely insulated from external destructive influences (eg areas adjacent to terrestrial National Parks); areas whose designation as MPAs would be politically acceptable and/or receive a high level of community support; areas of value for monitoring marine activities; highly accessible areas for tourism/recreation or education; areas whose designation would be compatible with existing uses, or which would be relatively easy to manage under existing management arrangements for the area (Kelleher and Kenchington 1991).

Note: adapted from Ivanovici 1984; Kelleher & Kenchington 1991; GBRMPA 1995.

4.2 MPA PLANNING AND CLASSIFICATION IN AUSTRALIA PRIOR TO THE NATIONAL REPRESENTATIVE SYSTEM (NRSMPA)

The Commonwealth's Ocean Rescue 2000 Program (OR2000), and its successor, the National Representative System of Marine Protected Areas (NRSMPA), were developed to coordinate and standardise the approach to MPA identification and designation in Australia, and to develop a biogeographically representative system of MPAs at national and regional levels. These initiatives are discussed in detail in the following section (4.3).

Prior to the development of OR2000 and the NRSMPA and its associated recommendations, the *ad hoc* nature of marine reserve identification and selection during the past two decades in Australia resulted in at least 23 major categories of MPAs (listed in Ivanovici 1984). Many of these various types of MPAs were declared for socially beneficial reasons, such as fisheries management or preservation of attractive dive sites (see Ivanovici 1984 and Zann 1996).

Apart from public lobbying, the variety of MPA categories resulted from a lack of national, State and local government coordination in MPA designation, coupled with the fact that MPA declaration could occur under more than one Act of legislation in each State (see below). In Victoria, MPAs have been declared under the *Fisheries Act 1968*, the *Crown Lands (Reserves) Act 1978*, and the *National Parks Act 1975* (Malcolm 1993). The situation is similar in other States, with MPAs usually being declared under State Acts for Fisheries, Forestry, National Parks and Wildlife, Conservation and Land Management, and Historic Shipwrecks.

To date, identification and selection criteria and management protocols differ in each State and territory. McNeill (1994) provided a listing of the various categories and sizes of MPAs in each State and territory. The *State of the Marine Environment Report* (Zann 1995) has also summarised the number and nature of the current suite of marine reserves in each State and Territory, and restated the oft-quoted fact that 74% of Australia's MPA area is contained in the Great Barrier Reef, leaving many biogeographic regions, ecosystems and community types underrepresented. McNeill's (1994) national analysis of MPAs is often reiterated in national and State Government reports, particularly the striking statistics such as the fact that, during the mid-1990s (i) less than 1% of Australia's marine habitats were protected by MPAs, if the area of Great Barrier Reef Marine Park is excluded; and (ii) the percentage of State waters protected by MPAs in South Australia was 1.4%, the lowest of any State (McNeill 1994).

From the 1970s to the present, there has been no standardisation of MPA classification in Australia, despite calls for consistent nomenclature from MPA planners and researchers (Ivanovici *et al* 1993, McNeill 1994). To add to the confusion, some MPA classifications change over time. For example, the Solitary Islands Marine Reserve in New South Wales has recently become a Marine Park, to reflect the multiple-use zoning. Furthermore, some MPAs classifications such as 'aquatic reserve' refer to zoned, multiple use areas in one State (eg the Towra Point Aquatic Reserve in New South Wales), and to highly protected, 'no-take' MPAs in another State (eg West Island Aquatic Reserve in South Australia). Even within one State, the terminology for MPAs can change over time (eg new MPAs in New South Wales are now called 'marine reserves' rather than 'aquatic reserves', the former term for MPAs established during the 1980s).

The nomenclature problem is exacerbated by the fact that more than one State agency is often responsible for managing MPAs. In New South Wales, for example, there are two State agencies and one national agency responsible for MPA management, each with their own preferred terminology for MPAs. More than one agency is responsible for MPA management in all other States, usually the State fisheries management department and the environment department.

Ostensibly, marine reserves in Australia have been declared according to criteria specified by the IUCN in 1975, endorsed by the Council of Nature Conservation Ministers in 1985, and later summarised into sets of guidelines for marine reserve establishment in Australia (Kelleher and Kenchington 1991; Kelleher *et al* 1995). It is by coincidence rather than design that many of the MPAs declared in Australia during the past two decades have satisfied these criteria. Many MPAs were identified and declared through public lobbying of recreation groups, conservation groups, interested private citizens, or scientists working at sites that they considered should be reserved. Until recently, standardised environmental criteria were rarely used for MPA selection, despite the inclusion of both biogeographic and ecological criteria in sets of national guidelines for MPA selection (Kelleher and Kenchington 1991; Kelleher *et al* 1995). For many MPAs, there is overlap in the reasons for designation. For example, an area designated for its social significance may also be categorised according to ‘accessibility’ or ‘representativeness’.

4.3 MPA PLANNING AND CLASSIFICATION UNDER OCEAN RESCUE 2000 AND THE NRSMPA

It was not until the early 1990s that the concept of a coordinated, comprehensive, *national representative system* of MPAs was advanced. In 1991, the Commonwealth (Labor) government recognised the inadequacy of Australia’s marine protected area system, and through the initiation of a 10-year research and development program called Ocean Rescue 2000, endeavoured to establish a biogeographically and ecologically-representative system of MPAs in Australia. This program aimed to assist each State to develop a more representative MPA system, particularly through the use of marine surveys, and collation of existing survey data and analysis, to identify biogeographical and ecological gaps in the existing MPA network.

In 1991, the Australian National Parks and Wildlife Service commissioned a report on MPAs, which drew on the expertise of an internationally renowned marine researcher (Ray and McCormick-Ray 1992). The report aimed to assist the process of developing a nationally representative system of MPAs to protect Australia’s diverse marine ecosystems and resources. Two years later, another national report titled *Marine Protected Areas and Biosphere Reserves: Towards a New Paradigm* (Brunckhorst 1994) was completed. The ‘Brunckhorst Report’ emphasised:

- a hierarchical systems approach to MPA site identification, bioregional representation, and protection of habitats that are linked by physical processes
- the Biosphere Reserve concept as a model for multiple-use MPAs that contain core protected sites (see Section 3.2.1)
- an integrated management approach to MPA development, involving national, State and local governments.

Since 1991, the Ocean Rescue 2000 Program has also coordinated a series of national meetings and workshops to develop ‘national, representative’ criteria for marine reserve identification and selection in Australia (eg Ivanovici 1993; Muldoon 1995; Thackway 1996).

The Ocean Rescue 2000 Program also developed a coastal marine bioregional classification scheme. The Commonwealth currently aims to use this classification scheme as the basis for (i) determining the representativeness of existing MPAs in each State and territory, and (ii) utilising a biogeographical framework for new MPA establishment, based upon the best available physical and biological information (Thackway and Cresswell 1996; IMCRA Technical Group 1998).

In 1997, Senator Robert Hill's (Minister for the Environment) press secretary publicly stated the Commonwealth Government's commitment to establishing a National Representative System (NRS) of MPAs, emphasising the need for a stronger, more strategic approach, compared with the *ad hoc* efforts of previous decades (Commonwealth Government Media Release, 5 September 1997). Senator Hill considered that whilst the Commonwealth would have a strong leadership role in this process, State governments, community and non-government organisations would be closely involved. The Marine Protected Areas component of the Commonwealth's Coasts and Clean Seas Program would be used to stimulate the States to contribute to the NRS.

Much of the groundwork laid for a national representative system by the OR2000 Program has appeared in the latest national productions, discussed below. However, there is contention as to whether the latest recommendations under the NRSMPA provide for adequate representation of all ecosystems / habitat types, and whether the system will include enough 'core' protected areas within its 'multiple-use' framework. This is discussed in Section 4.3.4.

It is clear from the examples cited above that significant work in MPA policy, planning and research has been undertaken at Commonwealth and State levels during the past decade, as part of the Ocean Rescue 2000 Program, towards the establishment of biogeographically and ecologically representative system of MPAs. It is important that this work be put to use in establishing the NRSMPA. Current MPA policy guidelines should incorporate the recommendations contained in documents pertaining to MPA planning and establishment, such as Kelleher and Kenchington's 1991 guidelines for MPA establishment, Keller *et al*'s 1995 update of the MPA guidelines; Ray and McCormick-Ray's (1992) consultancy report for the Commonwealth; the 1993 proceedings of the national Fenner conference on establishing MPAs (Ivanovici *et al* 1993); the 'Brunckhorst Report' (1994) on key principles for establishing a national representative system of MPAs; the Ocean Rescue 2000 Workshop Report series, and Thackway's (1996) proceedings of the 1996 workshop on developing a National Strategy for Identifying MPAs in Australia.

4.3.1 National guidelines for MPA establishment in Australia

The development of Australia's National Representative System of MPAs (NRSMPA) is one of the components of the Commonwealth Oceans Policy (Commonwealth of Australia 1998). The NRSMPA is ostensibly being implemented through a number of other Commonwealth Government strategies, such as the National Strategy for Ecologically Sustainable Development (Commonwealth of Australia 1992b), and the National Strategy for the Conservation of Australia's Biological Diversity (Commonwealth of Australia 1996). The States and territories have endorsed the NRSMPA, through the Intergovernmental Agreement on the Environment (Commonwealth of Australia 1992a).

Funding is being provided at the State level, from the Marine Protected Areas Program of the Commonwealth's Coasts and Clean Seas Program, for projects relating to the establishment of MPAs under the NRSMPA Guidelines.

ANZECC's National Representative System for MPAs has the primary goal of:

> '*…establishing and managing a comprehensive, adequate and representative system of MPAs, to contribute to the long-term ecological viability of marine and estuarine ecosystems; to maintain ecological processes and systems; and to protect Australia's biological diversity at all levels*' (ANZECC TFMPA 1998).

Achieving this goal would enable Australia to fulfil its international obligations as a signatory to UNEP's Convention of Biological Diversity (UNEP 1994), and as a contributor to the IUCN's Global Representative System of Marine Protected Areas (Pretty 1999).

Under the NRSMPA, new MPAs should be developed according to the following principles:

- *comprehensiveness* refers to the (possibly unachievable) goal of representing the full range of Australia's ecosystems, communities and their component biodiversity in the NRSMPA
- *adequacy* refers to the objective that the NRSMPA will have the size and the level of protection to ensure that ecological processes and viable populations, species and communities are not compromised
- *representativeness* means that the selected areas should reflect both 'typical' examples of the taxa, communities and ecosystems that occur at defined scales, as well as the variability within these units ('atypicalness').

The ANZECC Guidelines list a number of secondary goals, providing for a broad range of commercial, social and cultural activities in the MPA system. Apart from protecting 'representative' environments (see Section 4.3.3), the NRSMPA will also ensure the protection of (i) areas of high conservation value, such as species 'hotspots', centres of endemism and 'refugia'; (ii) rare, threatened or depleted populations; (iii) 'rare' communities and ecosystem types (eg that have unique physical and/or biological attributes compared with any other communities or ecosystem type, at a defined scale; and (iv) 'special groups of organisms', such as migratory species or those with complex life cycles.

The NRS approach to potential MPA identification and selection has adopted a *bioregional* framework for classification (ie the *IMCRA* framework, which was developed under the Ocean Rescue 2000 Program, and is discussed in the Section 4.3.2 below).

According to Environment Australia (*http://www.environment .gov.au/bg/nrs/nrsindex.htm (2000)*—Marine Protected Areas Program Information Page), the National Representative System proposes that new MPAs must be:

- implemented at the scale of ecosystems within bioregions (using the national IMCRA framework)
- established under nationally agreed guidelines (ANZECC TFMPA 1998)
- effectively managed to ensure biodiversity conservation and ecologically sustainable use of resources
- developed and implemented with full stakeholder consultation, including participation of all relevant community and resource user groups.

The NRS approach to potential MPA identification and selection has also adopted the IUCN's (1994) criteria for identification of potential MPAs. The IUCN identification criteria include:

- representativeness
- comprehensiveness
- biogeographic importance
- naturalness
- ecological importance
- international or national importance

- uniqueness
- productivity
- vulnerability.

Each of these criteria is described in the ANZECC (1998) Guidelines, and explained in detail in other documents (such as Kelleher and Kenchington, 1992, and the handbook *Application of IUCN Protected Area Management Categories* (IUCN 1995), and thus will not be reiterated here. The Commonwealth aims to work with States to ensure consistent application and interpretation of these categories (ANZECC TFMPA 1999).

Under the NSRMPA, new MPAs being added to the system should be defined to *protect biodiversity*, and should be classified into one or more of the IUCN categories outlined below. The problems with determining whether potential MPAs will protect biodiversity are discussed in Section 3.3 and 4.3.5.

Classification of MPAs in Australia under the NRSMPA must now follow the IUCN's international standard of categories for protected areas (IUCN 1994). These are shown in the table below.

Table 3: Protected area management categories.

Category	**Description**	**Comments**
1a	Strict nature reserve: MPA managed mainly for science.	Area of land and/or sea possessing some outstanding or representative ecosystems, geological or physiological features and/or species, available primarily for scientific research and/or environmental monitoring.
1b	Wilderness area: MPA managed mainly for wilderness protection.	Large area of unmodified or slightly modified land and/or sea, retaining its natural character and influence, without permanent or significant habitation, which is protected and managed to preserve its natural condition.
2	National Park: MPA managed mainly for ecosystem conservation and recreation.	Natural area of land and/or sea designated to (a) protect the ecological integrity of one or more ecosystems for this and future generations; (b) exclude exploitation or occupation inimical to the purposes of designation of the area;and (c) provide a foundation for spiritual, scientific, educational, recreational and visitor opportunities, all of which must be environmentally and culturally compatible.
3	Natural Monument: MPA managed for conservation of specific natural features.	Area containing one (or more) specific natural or natural/cultural feature which is of outstanding value because of its inherent rarity, representative or aesthetic qualities or cultural significance.
4	Habitat/Species Management Area: MPA managed mainly for conservation through management intervention.	Area of land and/or sea subject to active intervention for management purposes, to ensure the maintenance of habitats and/or meet the requirements of specific species.
5	Protected Landscape/Seascape: `MPA managed mainly for landscape/seascape conservation and recreation.	Area of land, with coast and seas as appropriate, where the interaction of people and nature over time has produced an area of distinct character with significant aesthetic, cultural and/or ecological value, and often with high biological diversity. Safeguarding the integrity of this traditional interaction is vital to the protection, maintenance and evolution of such as area.
6	*Managed Resource Protected Area*: MPA managed mainly for the sustainable use of natural ecosystems.	Area containing predominantly unmodified natural ecosystems, managed to ensure long-term protection and maintenance of biological diversity, while providing at the same time a sustainable flow of natural products and services to meet community needs.

Note: from ANZECC 1998.

Following 'identification' of potential MPAs, selection criteria must be prioritised according to:

- economic interests
- indigenous interests
- social interests
- scientific interests
- practicality / feasibility
- vulnerability assessment
- replication.

Again, these criteria are explained in the ANZECC Guidelines (ANZECC TFMPA 1998), and will not be detailed here.

There is a need to ensure that IUCN Protected Area Management criteria in the NRSMPA are not used to 'prop up' existing *ad hoc* reserves of inadequate size, space or level of protection to suit their intended purposes. It is possible that complete reliance upon IUCN criteria could result in more 'patchy' reserves which satisfy a number of important criteria on paper (eg 'of ecological importance' (as defined by Kelleher and Kenchington 1992 and Kelleher *et al* 1995)[3], but fail to adequately represent all bioregions and ecosystem types (and their component biodiversity). Similarly, a current MPA may satisfy a number of IUCN criteria, yet be subject to numerous abuses that devalue its conservation significance.

4.3.2 IMCRA: mapping bioregions for the NRSMPA

IMCRA, the Interim Marine and Coastal Regionalisation of Australia, is a regional planning framework for marine conservation and management, based upon GIS mapping of the physical and biogeographical attributes of coastal and marine regions around Australia. The IMCRA classification is based upon (i) a qualitative approach in which experts for each jurisdiction provided marine physical and biological data for inclusion in the IMCRA classification, and (ii) quantitative, analytical methods of biophysical classification. The mapping is hierarchical, from continental provinces (1000s of km), to bioregions (100s to 1000s of km), to biounits and ecosystems (10s to 100s of km) within bioregions. Because detailed data at the scale of ecosystems and biounits were not available, 'surrogate' data have been used to define coastal marine environmental biounits and their boundaries, based upon the best available physical, oceanographic and biological data. Technical details about the use of IMCRA and its component data at State level for MPA planning are provided in Chapter 6, on MPA Design**.**

Under the Commonwealth Coasts and Clean Seas Program, the IMCRA bioregional planning framework has been accepted to help guide the identification of areas for inclusion in the NRSMPA (IMCRA Technical Group 1998). However, the creators of IMCRA admit that, in order for IMCRA to be used as a bioregional planning framework for the NRSMPA, additional information regarding conservation attributes in each region will be required (IMCRA Technical Group 1998). The IMCRA framework is a practical one for identifying the 'broad-scale' components of a representative system of MPAs, at the scale of bioregions, but

[3] 'Ecological importance' includes areas which: contribute to the maintenance of essential ecological processes, such as a source of larvae for downstream areas; contain important feeding or breeding areas; contain rare or unique habitat types; contain rare or endangered species; contain 'genetic diversity', as defined by Kelleher *et al* as 'diverse or abundant' in species terms.

inadequate for determining MPA representation of ecosystem types within each bioregion, or for assessing levels of biodiversity within those defined scales. IMCRA regions and boundaries were developed using mainly physical and oceanographic information. In order for the IMCRA framework to be used to represent areas to achieve NRS goals such as ‘ecosystem maintenance’ or ‘biodiversity protection’, there is a need to determine concordance between major biodiversity patterns and the IMCRA bioregions. For example, distribution patterns of major indicator / structural / functional taxa in each bioregion and the ecosystem within them have not been mapped, so the extent that such classifications reflect marine taxal biogeography and biodiversity is limited. The broad-scale distributions of common (or well-known) species that have been used for IMCRA mapping appear not to be a very useful framework for biodiversity conservation, because the maps are based mainly upon existing, large spatial scale information for very few groups, and major marine groups have been omitted. It is possible that a classification system based only upon physical attributes such as substrate type and oceanographic variables will not be sufficient for identifying the variety of ecosystem types and their component biota, required for representation in a MPA network. This issue is further explored in Chapter 6.

Furthermore, MPA establishment mostly occurs at much smaller-area scales than the limits of resolution of regions which have been identified by the IMCRA, and the detailed site information required for potential MPA identification is not present in the IMCRA maps or databases. For example, conservation decisions are often made according to where (or how much) of a species / assemblage / ecosystem occurs relative to other areas / other systems.

4.3.3 Progress at State level in defining representative areas

Under Ocean Rescue 2000 and the NRSMPA, the Commonwealth and States are identifying biogeographically and ecologically representative areas for inclusion in the NRSMPA. A ‘representative’ area is one that comprises a particular set of oceanographic, physical and biological attributes, at a specified scale, that is ‘typical’ of areas within that specified scale, and distinguishes it from other surrounding representative areas with different sets of those attributes. Some areas are already well represented in the protected area network. Others are currently not included at all. The goal of the NRSMPA is to include examples of all ‘representative’ ecosystem types as MPAs, as well as ‘rare’ ecosystems / habitat types, and other sites of biological and ecological importance, within each bioregion (See Section 4.3.1). Classifying and identifying the range of representative areas is the first step. After scientific review of the classification process, selecting representative areas for inclusion in the NRSMPA is the next step, and requires both detailed consultation with (and input from) stakeholder groups, and analysis of the social, cultural, economic, ecological implications and ‘costs’ associated with different configurations of areas for NRS inclusion. This is the approach currently being taken in Queensland, for example (GBRMPA 1999). Some States have been involved with the classification and identification phases during the 1990s (see below), and will be engaged in the selection process in the early 2000s.

At a State level, government agencies and their collaborators in some parts of Australia have been classifying and identifying biogeographically and ecologically representative areas during the past seven years (eg Ortiz and Burchmore 1992 and Ortiz and Pollard 1995 for New South Wales; Edgar *et al* 1995 and 1996 for Tasmania; Edyvane and Baker 1995, 1996a, 1996b, 1996c, 1998, 1999a; Edyvane 1999, for South Australia). This has entailed various methods of classifying and mapping oceanographic, climatological, biophysical and biological data, from marine survey, and collation of existing records (eg from government departments, museums etc). Techniques for regionalisation have thus far employed a hierarchically-scaled approach, for mapping and classifying bioregions, smaller-scale ‘biounits’ and ecosystems / habitat types, using satellite imagery, aerial photography, multivariate analysis, GIS-based 'gap analysis' and other techniques. Qualitative methods (using the so-called 'Delphic' approach of collating data and knowledge from panels of marine experts) have also been used to complement the quantitative work. Details of these, and other, approaches to define representative areas for inclusion in the NRSMPA are discussed further in Chapter 6.

Recently, the Great Barrier Reef Marine Park Authority and other agencies connected with GBRMP management, have embraced the concepts of comprehensiveness, adequacy and representativeness, and embarked upon a five-year program to identify and select a range of representative ecosystems/habitats in the Great Barrier Reef, for inclusion in the NRSMPA. This will ensure that a more even representation of ecosystems/habitats are protected in the GBRMP, rather than limiting the 'protected' parts of multiple-use MPAs to dominant ecosystems (coral reefs, in this case) or single, high-profile species (GBRMPA 1999).

4.3.4 NRSMPA and 'MOMPAs': multiple objective or multiple abuse marine protected areas?

In parallel with the development of the NRSMPA, elements of Ocean Rescue 2000s Protected Area Policy have been incorporated into the new Commonwealth Oceans Policy (Commonwealth of Australia 1998). The Oceans Policy seeks to create a comprehensive, compatible and equitable management strategy for Australia's seas and oceans, based upon all major uses and user groups. As part of this strategy, which has marshalled strategic coordinators and technical working groups in each State for the creation of a 'Marine and Estuarine Strategy', an emerging trend in marine protected area planning has been the *MOMPA* (multiple-objective marine protected area). This is yet another term for large, multiple-use MPAs. Examples of existing MOMPAs include the Great Barrier Reef Marine Park in Queensland, The Great Australian Bight Marine Park in South Australia, and the Coburg Peninsula Marine Park in the Northern Territory.

There appears to be increasingly national and State-level interest in the creation of MOMPAs (ie large, multiple-use marine parks). There is concern that MOMPAs may not include an adequate number of 'core' protected areas (IUCN categories 1 and 2), nor protect a sufficient number of areas from detrimental impacts. National (1998 Oceans Policy) and State-level (eg South Australian Marine and Estuarine Strategy 1998) documents state that conservation of biodiversity and ecological processes is the major objective of MOMPAs. Whether MOMPAs will actually assist in biodiversity conservation will probably be determined more by the number of higher protection category MPAs that are permitted in each proposed new MOMPA.

Traditionally, the greater the number of small, single objective reserves that are declared at State level, the smaller are the number of future options available for representing more ecologically and biogeographically significant areas. It would be unfortunate if the current *ad hoc* patchwork of MPAs that will 'sit' inside new large MOMPAs, are accepted as sufficient 'core protected' sites within the 'sea of multiple use'.

The Federal Environment Minister Senator Robert Hill's press secretary stated in 1997 that Australia's various ocean industries will be involved with the process of MPA (including MOMPA) establishment, and that:

> *'…marine protected areas should not be seen as a threat to the viability of those industries and, if consistent with the conservation values we are seeking to protect, we believe that industry can have continuing access to natural resources within these areas' (Commonwealth Government Media Release, 5 September 1997).*

It remains to be seen how the '*conservation values we are seeking to protect*' will be integrated within the multiple-use framework of MOMPAs. Thackway (1996) listed key questions that have not been answered about MOMPAs; these questions include the following:

- Are MOMPAs and biosphere reserves (see Section 3.2.1) equivalent to IUCN categories V and VI (see Table 3)?
- Is there a risk that when MOMPAs have broad-based conservation goals, they can become ineffective for biodiversity conservation?
- Will the goals of establishing MOMPAs produce a different set of MPAs than a set that is predicated upon ecological criteria and biodiversity conservation?
- Is there an imbalance in the MOMPA model regarding economic goals and uses having a greater 'right' to the resources than ecological goal? That is, the areas set aside within MOMPAs for conservation purposes only seem very small within a 'sea' of regulated resource access and development rights given to industry (Thackway 1996).
- Where competing resource uses have the potential to degrade selected areas of the coast and/or marine environment, should MOMPAs be established to minimise or regulate these conflicts and conserve the natural environment? If so, what are the targets, and what selection criteria, data and information should be used?

There is some concern regarding the potential use of a new set of multiple-objective, multi-user, and industrially-driven guidelines for MPA establishment, along the lines of the Commonwealth's proposed system of MOMPAs (Commonwealth of Australia 1998). Although stakeholder involvement is important in MPA planning and establishment (see Chapters 7 and 9), it is possible that Commonwealth provision for 'primary stakeholders' (fishers and other resource extractors) to have *priority* involvement in MPA identification and selection, could result in a biased process of identifying suitable areas for MPA establishment. This would conflict with the NRSMPA's own stated objectives of achieving a comprehensive, adequate and representative system of MPAs which 'maintain ecological processes and systems', and 'protect biodiversity at all levels'. The stakeholder conflicts associated with MPA identification and designation are discussed in Chapter 7.

4.3.5 Will the NRSMPA protect biodiversity?

In Australia, the importance of protecting this country's marine biodiversity has been highlighted recently in a number of Commonwealth initiatives, including the National Strategy for the Conservation of Australia's Biological Diversity (Commonwealth of Australia 1996), the State of the Marine Environment Report (Zann 1995) and the National Strategy for Ecologically Sustainable Development (Commonwealth of Australia 1992). A consultancy report to the Commonwealth on a strategy for a National Representative System of MPAs (Ray and McCormick-Ray 1992) considered biodiversity representation and conservation as a major guideline in MPA establishment for Australia.

These documents recognise Australia's globally significant levels of marine biodiversity and endemism, and the importance of biodiversity maintenance at all levels for many reasons, such as:

- the continuation of 'ecological services' (eg pollutant absorption; climate regulation; prevention of coastal and subtidal erosion; water quality maintenance; nutrient storage and recycling; substrate production; and protection of ecosystems against establishment of invasive species)
- highly valued social and economic benefits (eg wild fisheries, recreation and tourism, derived marine industrial and medical products)

- the maintenance of genetic variability and 'feature' diversity, which provides the material for maintaining future states of biodiversity at many levels
- subjects for scientific research and natural area management, among other values.

Biodiversity conservation has other, socially useful and practical benefits, including the ability to offset the rising costs of damage mitigation and remediation in degraded ecosystems (Commonwealth of Australia 1996), and to provide sensitive indicators to environmental change. Apart from these utilitarian aspects of marine biodiversity conservation, the intrinsic values of maintaining intact ecosystems and habitats, and their component biotic assemblages, should not be underestimated.

The House of Representatives Standing Committee on the Environment (1993) considered that an essential feature of Australia's biodiversity strategy should be a system of protected areas, '*designed and managed to represent and protect the diversity of ecological communities, species and gene pools*'. (HORSERA 1993; p. 7). Wholesale biodiversity protection is not possible using MPAs, because of the great number of them needed to fully represent marine biodiversity at many taxonomic levels and ecological scales. Also, it is socially and economically infeasible to restrict use of the large number of areas that would be required in a comprehensive network for biodiversity protection. Although MPAs have been often recommended as a tool for biodiversity conservation (Ray and McCormick Ray 1992; Brunckhorst 1994; Vanderklift *et al* 1998; Allison *et al* 1998; ANZECC TFMPA 1998 and 1999), the majority of current MPAs in Australia were not designated for that purpose. Prior to the publication of the NRSMPA Guidelines, existing MPA guidelines (Kelleher *et al* 1995) did not specifically provide for the maintenance and protection of biodiversity as a criterion for MPA declaration, despite the inclusion of both biogeographical and ecological criteria in those guidelines. Marine biodiversity conservation in Australia has adopted a piecemeal approach to date, emphasising the conservation of (i) specific species, without consideration of the biodiversity and ecological significance of co-occurring taxa, or (ii) specific locations within ecosystems, without knowledge of the biodiversity which they, and other non-protected ecosystem types, contain.

Ray and McCormick-Ray (1992), suggested that 'seascape' (bioregional) level measurement of biodiversity was a practical means of ensuring representation, compared with thorough, species level biodiversity inventories, which are infeasible given constraints in time, knowledge and resources. The 'bioregional' approach has been adopted by the Commonwealth, using the IMCRA framework (discussed above). The potential for IMCRA to be used for biodiversity conservation, and its limitations, are discussed above, in Section 4.3.2.

It is not clear from the current NRSMPA policy and guideline documents to what extent 'biodiversity' will be protected by the representative system, especially if all ecosystem and community types in each State's waters are not represented in the systems. There is a need for current Commonwealth policy to not conflict with international (IUCN, UNEP, WWF) or national objectives for biodiversity conservation. This would include the National Strategy for the Conservation of Australia's Biological Diversity (Commonwealth of Australia, 1996), and HORSERA's (1993) report on the importance of protecting Australia's biodiversity at all scales. The Commonwealth proposes that biodiversity protection and maintenance of ecological processes are the *major* goals of the NRSMPA, although there appears to be little formal provision for investigating or protecting marine biodiversity, particularly at State levels. It should be recognised that maintaining biodiversity in a comprehensive, adequate and representative way is a unit-specific and scale-dependent process (see Section 3.3). Ideally (but perhaps not practically), the process requires adequate types and amounts of biodiversity data, high levels of site protection at the community and ecosystem levels, and active, adaptive management of extractive activities and systemic impacts. These requirements are also likely to conflict with 'primary stakeholders' interests in resource extraction and coastal marine developments (see Section 4.3.4).

A potential strategy for ensuring that the NRSMPA is used for biodiversity conservation might include the following:

Standardising the State-level approaches to biodiversity assessment for MPA planning. This would involve standardising, at a national level, the current disparate analytical techniques that exist for classifying temperate Australian marine bioregions and ecosystems and their component biodiversity (Ortiz and Pollard 1995; Edyvane and Baker 1995 and 1996; Edgar *et al* 1995; Thackway and Cresswell 1996; GBRMPA 1999). Data collation and analysis for biodiversity conservation should consider how adequately biodiversity can be represented at various space and time scales, and which units of conservation should be chosen as biodiversity representatives (see Section 3.3). Chapter 6, on MPA design, discusses the various approaches that have been used by each State. For areas without adequate biodiversity data, classifying and representing the range of physical attributes of ecosystems and communities (supplemented by available biodiversity data) is considered to be an interim substitute (IMCRA Technical Group1998; GBRMPA 1999)[4].

Ranking the 'biodiversity value' of sites, according to nationally standardised criteria. If coastal marine areas can be evaluated according to their 'biodiversity value', then a framework for protecting 'priority areas' can be produced, using MPAs as one part of 'whole system' management. Ideally, before such biodiversity evaluations can be made, biodiversity should be quantified, particularly objective measures of biodiversity between samples and sites. This is a long, expensive and difficult task. Recent developments in computer-based biodiversity survey data analysis and mapping can assist in some areas for which biodiversity data are available, and only if such products are properly used. Examples include the WORLDMAP program (British Museum of Natural History), and the BIORAP products developed in Australia by CSIRO, the Australian National University and DEST.

Enhancing taxonomic capacity. Basic taxonomic inventory of many marine biotic groups, at all scales, is lacking in Australia. A commitment to biodiversity conservation would address this shortcoming. Some groups, such as pelagic fish and algae, are better known and quantified than others (crustaceans, sponges, cnidaria) (Poore 1995)[5]. The basic kinds of data sets that are used in terrestrial habitat inventory and conservation assessment have not been compiled for most marine regions and ecosystems. This has far-reaching effects, because objective site selection techniques for biodiversity conservation (using the principles of *comprehensiveness, adequacy, and representativeness*) should utilise good taxonomic information. The taxonomic impediment has made it difficult to even define marine biodiversity in a workable way for MPA-based biodiversity conservation. Ideally, funds would to be made available to museums, universities and research institutes to (i) identify existing collections;

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[4] It is interesting to note that the classification system being used in Queensland to identify representative ecosystems and habitats for inclusion in the NRSMPA is based mainly upon physical data, and that GBRMPA (1999) considers that there is a lack of biological/biodiversity data for broad areas of the GBRMPA. This is surprising because, compared with other States, the knowledge of marine biodiversity in the GBR appears to be quite advanced, due to the large amount of marine surveying, mapping and data collection that has been undertaken in that region for several decades.

[5] HORSERA (1992) reported that most of the invertebrate marine fauna of Australia has never been documented, and catalogues and descriptions are only available for those few groups in which active taxonomic work is being undertaken. For example, even as recently as the past decade, several new genera and hundreds of new species of nudibranchs and other opisthobranch molluscs have been described from Australian waters by Brunckhorst and associates, indicating the previous poor state of knowledge of this mollusc group.

(ii) sample those areas in which the biotic composition is still unknown; and (iii) permit the information to be used for more informed biodiversity conservation decision-making. If applied correctly, computer-based rapid biodiversity assessment techniques (see above) can assist this purpose.

Ensuring that 'core' protected areas are part of multiple-use MPAs. Biodiversity conservation cannot realistically be achieved by zoning entire coastlines into multiple-use areas without 'core' protected areas (eg IUCN categories I and II) that prohibit any form of interference or extraction. There is provision for these 'core' protected areas to be established in the NRSMPA, and it should be ensured that resource management issues do not override this major objective of the NRSMPA.

4.3.6 How 'representative' will new MPAs be?

The NRSNMPA stresses that a biogeographic/ecosystems approach will be used to identify gaps in Australia's current MPA network. When planning new MPAs, there is need to ensure that the area actually represents the entity that it is supposed to be representing at a given scale, such as a 'bioregion', 'biounit', or 'ecosystem type'. It is hoped that historical approaches to MPA identification (often chosen to meet specific stakeholder objectives or single species protection) are not perpetuated in the NRSMPA as the major driving force for new MPA establishment. Existing MPAs should be assessed in terms of the biogeographic, ecological, and social criteria identified in the NRSMPA, and existing MPAs in particular locations should not, by their existence, limit opportunities for a more bioregionally-representative MPA system. New MPAs that are designated at State level for single species protection should also not preclude possibilities for bioregionally-representative examples in that State to be included in the MPA network. An example of this problem would be designating as a MPA a breeding ground for a large marine mammal, and that habitat might comprise bare sand as the major mappable 'ecosystem type'. If other ecosystem types, such as rocky reefs and seagrass beds are found in the bioregion, they might not be considered for inclusion in the MPA network if a portion of the bare sand ecosystem has already been declared as a MPA, and there are limits to the percentage representation of the bioregion (eg 10%-20%) that are politically feasible and publicly acceptable.

4.4 NATIONAL EXAMPLES OF SUCCESSFUL MPA ESTABLISHMENT

It is difficult to separate the benefits of MPAs into specific categories, because many of Australia's MPAs, particularly the larger ones, serve a variety of purposes. Such purposes include protection of biogeographically or ecologically representative areas; conservation of endangered or rare species; provision of recreation/tourism areas; and protection of important nursery areas for fisheries management. The Shark Bay Marine Park (containing the Hamelin Pool Marine Nature Reserve) is an example that serves all of the aforementioned purposes.

4.4.1 MPAs for fisheries management and enhancement

There are a number of MPAs in Australia that protect the estuarine mangrove and seagrass nursery habitats for juvenile fish and crustaceans, and fish spawning habitat. Although most of these are not monitored, their role in sustaining coastal fisheries is implicit. Examples include Queensland's 'no-take' fish sanctuaries such as Swan Bay (Stradbroke Island), designated for the protection of spawning grounds of commercial fish species, and other sanctuaries in North and Central Queensland, which have been designated for the conservation and management of barramundi, mud crabs and other harvested coastal marine species (Ivanovici 1984).

Relatively few quantified examples of the benefits of MPAs for fisheries management exist in Australia, compared with the rapidly increasing evidence from overseas studies (see Section 3.4.1). Until recently, most Australian work of this kind was conducted in closed areas in the Great Barrier Reef Marine Park (GBRMP). Despite early criticisms of 'no-take' zones in the Great Barrier Reef, it has become accepted that they have been successful, enabling conservation and multiple-uses to co-exist. Comparatively few restrictions are placed on most user groups. For example, prawn trawlers lost only 5% of their total trawlable area when the large GBR Marine Park was zoned during the 1970s (Shorthouse 1991).

Examples from the Barrier Reef include results from reefs closed to fishing for experimental purposes (temporary MPAs). Ayling and Ayling's (1986, cited by Roberts and Polunin 1991) comparative study of the leopard coral-grouper (*Plectropomus leopardus*) on fished and unfished reefs showed that grouper were, on average, 10 cm larger on the six protected reefs, and 78% of fish were over 35 cm long (compared with 46% on the six fished reefs). The reefs had been closed to fishing from 2.5 to 12 years. Beinssen (1989, cited by Roberts and Polunin 1991) compared populations of the same grouper species in fished and unfished reefs in a more northerly section of the GBR, and found that after 3.5 years, grouper were around 13 cm bigger in the unfished areas. Craik (1991) reported that other Barrier Reef MPAs closed to fishing (including trawling) provided successful 'refuges' for various harvested coral reef fish species.

Rigney (1990) reported significant increases in the abundance of large coral trout in a 'no-take' area on the Barrier Reef, and approximately twice as many juvenile trout on fished reefs compared with the situation prior to the closure. This provided some indirect evidence for replenishment of the surrounding fished area with recruits from the MPA. However, more recent work with coral trout (Zeller and Russ 1998) has shown that although Barrier Reef 'no-take' MPAs act as refuges for 'sedentary', site-attached fish such as coral trout, such species do not readily move out of MPA boundaries into the adjacent fish area. This contrasts with the behaviour of more mobile species, which have been demonstrated to move out of MPAs (eg see examples in Sections 3.4.1 and 5.2.1 on replenishment of fished areas with 'exported' fish from MPAs).

The successful MPA reported by Rigney (1990) was also a prawn habitat, and prawn fishers operating outside the MPA reported increased catches of prawns. Other evidence of MPA success in the GBRMP comes from trawl fishers who operated in the area around the 'no-take' Shelburne Bay Cross Shelf Transect during the late 1980s, and reported increased catches from 'fishing the line' (ie trawling close to the closed area) (Shorthouse 1991).

As the above examples show, 'temporary MPAs' (fishing closures for specified time periods) are becoming more frequently used in Queensland, as a means of permitting heavily harvested (and sometimes overfished) populations to recover. A more recent example is the closure in 1998 of some GBR reefs to assist the recovery of heavily harvested black teatfish (a sea cucumber species). Densities of teatfish in the largest closed area have recovered after only one year of protection (Australian Institute of Marine Science, unpublished media release, 1999).

Regarding the link between protection of fish spawning stock and recovery of depleted abundance, there is evidence from two Western Australian gulfs in which tiger prawns are fished; the populations in both gulfs reportedly declined due to over-fishing, but only one population was subsequently protected, and it was that population which recovered abundance (Caputi 1993).

Benefits for fisheries are also emerging from work in Tasmanian MPAs. Recent studies on the east coast of Tasmania (Edgar and Barrett 1997 and 1999) reported that *one* year after it was declared, the MPA at Maria Island produced (i) increased densities of rock lobster, sea urchins, wrasse, leatherjackets and abalone; and (ii) a statistically significant increase in average size of black-lip abalone (*Haliotis rubra*). *Six* years after MPA establishment, significant gains in species and habitat protection and fisheries enhancement were noted compared with fished reference sites. Examples included (i) significant increases in the number of fish, invertebrate and macroalgae species; (ii) significant increases in the densities of larger fish (> 32.5 cm), bastard trumpeter (*Latridopsis forsteri*), and rock lobster (*Jasus edwardsii*); and (iii) significant increases in the mean size of blue-throated wrasse (*Notolabrus tetricus*) and black-lip abalone. An order of magnitude increase in lobster biomass was noted in the largest MPA, as was an increase of two orders of magnitude in the abundance of bastard trumpeter.

The success of New Zealand's famous Leigh (Cape Rodney-Okakari Point) Reserve MPA is included as an example in this section, because New Zealand's marine ecosystems and fish fauna are similar to those in southern Australia. Ten years after the MPA was established to protect the rocky reef and kelp beds near Leigh, surveys of the area over 12 years showed (i) significantly increased sizes and densities (ie from 2.5 to 20 times higher) of rock lobster (*Jasus edwardsii*); (ii) significantly increased abundance and/or densities of moki (*Cheilodactylis spectabilis*), snapper (*Pagrus auratus*) and blue cod (*Parapercis colias*); (iii) increased numbers and 'more natural' densities and distribution of other fish species; and (iv) 'generally higher' sea urchin density (Ballantine 1987, 1988, and 1989; Cole *et al* 1990).

The rock lobster and scale-fish fisheries in mid-northern New Zealand benefited from the Leigh reserve. Some of the lobster and fish (which had increased in abundance and density due to the protection of the MPA), 'migrated' into the adjacent fisheries, as evidenced by tagging studies and capture of tagged individuals by fishers (Ballantine 1987; Rigney 1990; Ballantine (pers. comm.) cited by Rowley 1994). During the 1980s, fishers were fishing the boundary of the reserve because lobsters were larger and 10 times more plentiful than in other fished areas (Rigney 1990); ie, lobster pots were often placed at the boundary of the MPA to 'catch the spillover' (McDiarmid and Breen 1993).

4.4.2 MPAs for ecosystem/habitat protection

Many MPAs in Australia have been declared to protect estuarine fish habitat, particularly in Queensland. Although fishing is permitted in most of these MPAs, any activity that might damage the samphire, mangrove and/or seagrass estuarine habitat is prohibited. Major seagrass habitats that are of ecological significance, and also important to commercial and recreational fisheries, have also been protected in Swan Bay, Victoria. Shark Bay Marine Park in Western Australia protects the largest seagrass ecosystem in the world, and has World Heritage status. Ivanovici (1984, and 1993 update) provides a complete list of MPAs that have been declared to protect estuarine and seagrass habitats.

Some MPAs in Australia have been declared specifically to protect 'unique' ecosystem types. The Ninepin Point Marine Reserve in Tasmania, for example, was designated to protect an unusual ecosystem in which deeper water species (from 20 metres or more) occur in relatively shallow water (8 metres) due to poor light penetration associated with the tannin-rich waters from the Huon River. Another example of an unusual habitat type protected by a MPA is the Nypa Palms National Park at the delta of the Herbert River in Queensland, which protects the southernmost stand of Nypa Palm. Western Australia has one of the most unusual coastal marine ecosystem types in the world, the highly saline near-shore pools of the Hamelin Pool Nature Reserve, containing stromatolites, unique structures up to 1.5 metres high, which are built by single-celled algae (CALM 1999). Several other examples of Australian MPAs declared to protect unique ecosystem types are listed in Table 2, and in Ivanovici (1984 and 1993).

A significant example of ecosystem protection using MPAs comes from Tasmania, where increases in species richness of macro-algae, as well as a change in the dominant species of macro-algal reef cover, accompanied the increase in biomass, density and / or abundance of fish and invertebrate species in the largest of four MPAs (at Maria Island) (Edgar and Barrett 1999). Using as evidence the difference in macro-algal communities and species compositions inside the MPA compared with the surrounding fished area, the authors conclude that Tasmanian shallow reef ecosystems are overfished, to the extent that ecosystem composition is now vastly different compared with what it would be in an unfished situation.

4.4.3 MPAs for biodiversity conservation

In simplistic terms, it could be argued that any MPA that protects a single fisheries species, a representative ecosystem / habitat, or a regionally 'unique' ecosystem type, is contributing in some way to biodiversity conservation, by providing *de facto* protection to all species associated with that MPA, regardless of its primary designation. If that is the case, then many of Australia's current MPAs fall into that category, and have been successful at local scales in protecting biodiversity, at particular scales, from destructive influences. (Specific examples are provided below). However, as discussed in Sections 3.3 and 4.3.5, Australia's current system of describing biodiversity and protecting it in MPAs is inadequate, with much of Australia's marine biodiversity still unknown and unprotected, and many types of ecosystems / habitats not represented at all in MPAs, particularly in southern Australia.

Protected 'core' areas (such as 'preservation zones', 'sanctuary zones' and sites of special scientific interest) in large, multiple-use marine parks, such as Great Barrier Reef and MPAs in Western Australia, provide biodiversity protection by prohibiting, as far as possible, any destructive influences on the fauna, flora, water column or seabed associated with the site.

One of many examples of existing Australian MPAs that protect biodiversity is Rowley Shoals Marine Park, off the coast of Broome in Western Australia. Rowley Shoals provides protection for three steep coral atolls (each approximately 80–90 square kilometres) that are considered to be exceptionally diverse in their fish and invertebrate fauna, with 233 species of coral and 688 species of fish. The atolls are also considered to be unique in their composition and species abundance, compared with other coral reefs in Australia (CALM 1999). Another Marine Park in Western Australia (Shark Bay), has an exceptionally high diversity of seagrass species (12 species have been recorded) (CALM 1999).

Even 'tiny' MPAs, such as the Shiprock Reserve near Port Hacking New South Wales, can assist biodiversity conservation, depending upon their location and contents relative to surrounding areas. Shiprock, for example, contains an 'extremely rich and varied fauna'; 'a rich growth of marine invertebrate from many different phyla'; 'a large population of fishes' and is 'surrounded by comparatively sterile[6] sand flats' (Lawler pers. comm. 1972, cited by Pollard 1977).

4.4.4 MPAs for rare/vulnerable/endangered species conservation

The National Representative System of MPAs' Policy specifically provides for the protection of rare and threatened (eg vulnerable or endangered) biota. However, concern has been expressed about the potential inadequacy of the NRSMPA to provide protection for such

[6] Note that in terms of marine biodiversity, sand flats are rarely 'sterile', but rather contain high diversity of small infauna and interstitial fauna that is often not regarded by the general public as an important component of biodiversity, compared with larger, colourful, attractive biota.

biota, because a system of MPAs based upon biogeographic regions and their component ecosystems may not take into account the distribution of threatened species (Preen 1998).

Apart from marine mammals, marine reptiles and some seabird species, the conservation status of most marine fauna and flora in Australia is poorly known. It is likely that numerous endangered, rare and vulnerable species exist in Australian waters, but have not been given an IUCN listing (nor protected status in MPAs) due to this paucity of knowledge regarding their existence, their population sizes and their ranges.

An example of a 'unique' ecosystem type that contains many rare and/or endemic species is the Port Davey/Bathurst Harbour MPA in Tasmania. The MPA includes complex, shallow water habits for soft corals, sea whips, sponges, bryozoans, angler fish and lingcod, all of which are usually found in deeper water, and most of which are not found in elsewhere in Tasmania (Kriwoken 1993).

Hamelin Pool Marine Nature Reserve in Western Australia protects one of the world's truly unique marine biotic formations, the stromatolites. CALM (1999) reported that a metre-high stromatolite could be hundreds of millions of years old, due to the exceptionally slow growth rates of these living structures. The Hamelin Pool stromatolites are one of the major living examples of their kind in the world, most other stromatolites being fossilised in rocks (up to 3500 million years old).

A recent example of MPA-based protection for 'rare' and endemic species is the nomination of the Lord Howe Island sea mount and its associated ecosystems. The island group is situated at the boundary of tropical and temperate water masses, and therefore supports a highly endemic mix of biota, including tropical coral species at the southern limits of their distriubtion, and sub-tropical species that are rare or absent from the Great Barrier Reef system (Wilks 2000).

A number of MPAs have a *de facto* role in endangered species conservation, even though they were established for other purposes, such as fisheries management. For example, some of the fish habitat reserves in north and central Queensland contain seagrass habitat for dugongs. Although fishing is permitted in these reserves, habitat must be protected under the *Queensland Fisheries Act, 1976-89* (Ivanovici 1984 and 1993). Two of Western Australia's MPAs also provide protection for dugongs, which are considered to be threatened in the Indian Ocean due to mesh netting and hunting (Preen 1998). Dugongs are also protected in the Ningaloo Marine Park in Western Australia (CALM 1999).

Similarly, the Yongala historic shipwreck MPA near Townsville in Queensland was not declared for the preservation of rare or endangered species, but also serves that purpose, because it contains populations of two rare and endangered black coral species, and a regionally endangered fish species (grouper). Julian Rocks (Byron Bay), a MPA that was declared to conserve a representative sample of a tropical/temperate transition ecosystem, also contains endangered black coral species (Ivanovici 1984 and 1993).

Several tropical MPAs serve to protect nesting sites for turtles. For example, the coral reefs and cays of the Capricorn-Bunker Marine Park in southern Queensland provide protection for important nesting areas for seabirds and turtles, including green, loggerhead, leatherback and hawksbill turtles (Ivanovici 1984 and 1993). HORSERA (1992) considered the loggerhead turtle to be rare on a global scale, apart from its numbers in Australia, and the 'leathery' turtle to be the most endangered of the turtle species found in Australian waters. The sanctuary at Coburg Peninsula in Northern Territory also protects turtles (green, leatherback, hawksbill and Olive Ridley species), as well as dugong, although traditional hunting of species for Aboriginal food sources is permitted. One of Australia's few offshore MPAs (Ashmore Reef National Nature Reserve in the Timor Sea) has been successful in protecting turtles and seabird species from over-exploitation. Breeding success of both groups has increased since the MPA was established, particularly the number of breeding pairs of least frigate birds, which were previously unsuccessful at breeding on Ashmore Reef due to over-harvesting before the MPA was established (GBRMPA 1995).

A number of MPAs occur in areas visited by endangered whale species. Coringa-Herald and Lihou Reef National Nature Reserves in North Queensland, for example, protect areas for the Humpback whale (*Megaptera novaeangliae*), and Humpbacks and Southern Right Whales are also seen in Marmion Marine Park, north of Perth. Some of Australia's MPAs have seasonal restrictions that require shipping routes to be changed during times of whale migration, to reduce impacts.

Compared with marine mammals, the conservation status of many non-commercial fish species poorly known in Australia. One exception is the group of handfish species, which are some of the very few marine fish species in the world to be listed as either rare or endangered. These species occur only in Tasmania. Although the Maria Island Marine Reserve in eastern Tasmania was not designated to conserve handfish, it protects the cave habitats of the red handfish, and mating pairs of this endemic species of very limited distribution, live in the MPA.

4.4.5 MPAs for tourism/recreation, education and/or research

The most widely known example of an Australian MPA that serves valuable purposes for tourism, recreation and marine education is the Great Barrier Reef Marine Park, but other large, multiple-use marine parks in Western Australia have similar roles. State, national and international tourists regularly visit Ningaloo Marine Park in WA. Ningaloo is the only accessible coral reef system in the world where the world's largest shark (Whale Shark *Rhiniodon typus*) seasonally comes close to shore, and this feature has given the park an international reputation for nature-based tourism, which is highly beneficial for the local economy. Similarly, WA's Shark Bay Marine Park is famed for its nearshore dolphins, at Monkey Mia, which have been the subjects of much tourist attention for many years.

Marmion Marine Park, north of Perth, is one of the most popular dive spots in Western Australia due to the diversity of marine life associated with its underwater features such as caves, ledges and swim-throughs, and the prevalence of sea-lions and dolphins in the park. Most of Western Australia's other multiple-use MPAs (Ningaloo, and Rowley Shoals Marine Parks, for example) have highly diverse coral reef ecosystems within the parks, which are an important feature for tourism / recreation.

Australia's large marine parks permit recreational fishing outside of sanctuary zones. Much of the area of the GBR is available to recreational fishers, in the form of 'general use zones', 'habitat protection zones', and 'estuarine conservation zones'. Similarly, Western Australia's marine parks have large 'recreation zones' for a variety of activities, including recreational fishing.

Many other smaller MPAs in Australia serve a valuable purpose for tourism / recreation and education, including those that were originally established for other purposes, such as protection of reef fish populations and their associated habitat from spearfishing, or the protection of historic shipwreck sites. For example, six shipwrecks in Queensland have been declared as MPAs, and are popular dive spots.

There are many examples of small MPAs that have important recreational / tourism functions. The Solitary Island Marine Reserve (now Marine Park) in New South Wales, contains a few small sanctuary zones in which fishing is excluded, and SCUBA diving in that MPA is considered to be a 'big income earner' for the local economy of Coffs Harbour (GBRMPA

1995). Similarly in Tasmania, the local economy at Bicheno has improved due to the increase in tourism (particularly dive tourism) since the establishment of the Governor Island Marine Nature Reserve (GBRMPA 1995). The Annulus (Popes Eye) Marine Reserve in Victoria, an artificial bluestone reef MPA covered with sponges, ascidians, soft corals, bryozoans, sea fans and many fish species, has become one of the most popular snorkelling and SCUBA diving sites near Melbourne (GBRMPA 1995).

A marine park has been proposed for Port Phillip Heads area in Victoria (Environment Conservation Council 1998). Apart from the ecological values of the region, it is estimated that 80% of Victorian charter boat diving occurs in the southern Port Phillip Bay region, with a combined recreational and commercial dive value in the multi millions per annum.

New Zealand's Leigh Reserve is a popular example of the social benefits of MPA establishment[7]. The area is regularly visited by large numbers of divers, and a visitor survey during the 1980s (New Zealand Department of Lands and Survey 1984, cited by Cole *et al* 1990), showed that the existence of the MPA was likely to be a contributing factor to human use of the relatively remote Leigh area. Survey respondents stated that the prime reason for visiting the Leigh MPA was the opportunity to see large numbers of fish and lobsters, which they would not otherwise have an opportunity to see because the surrounding areas were continuously fished. Both the Leigh and Poor Knights Islands Marine Reserves in New Zealand have been the subject of numerous scientific studies in fisheries ecology, benthic ecology, fish population dynamics, during the past two decades (see references by Ballantine), and have provided some of the best evidence of the effectiveness of MPAs for protecting habitats, conserving local marine populations, and enhancing fisheries.

Australian examples of MPAs that are used for marine education include (i) the Tinderbox Marine Nature Reserve in Tasmania, in which an underwater interpretive trail has been established for snorkellers and divers; and (ii) Long Reef Aquatic Reserve in New South Wales, which is used to educate students in nearshore marine ecology (GBRMPA 1995).

[7] New Zealand is included in this section due to the similarity of its marine ecosystems to those on the east and southern coast of Australia.

5 MPAS IN SOUTH AUSTRALIA

5.1 BACKGROUND

Much has been written about the classification, management arrangements, uses, and benefits of existing MPAs in South Australia (see Ivanovici 1984; Johnson 1988a and 1988b; Neverauskas and Edyvane 1991; Edyvane 1996b; Lewis *et al* 1998), therefore such information will not be reiterated in detail for this report. To summarise, there is one *marine park* in South Australia (Government of South Australia 1998) and 14 *aquatic reserves*. Other types of MPAs in South Australia:

- offer specific protection to a single, economically important species, and permit all activities in the area except the collection of the protected species (four *sanctuaries* for rock lobster)
- are terrestrial, with an intertidal component, but offer *no subtidal protection* (one *national park*, five *conservation parks*, one *game reserve*)
- prohibit specific types of fishing activities, such as netting or spearfishing (*restricted use areas* such as jetties, piers, wharves, netting closures). Around South Australia, more than two dozen netting closures have been designated for the purpose of fish stock sharing and protection (eg PISA 1994);
- protect the structures, associated biota and surrounding waters of a sunken ship of historic / cultural significance (one *historic shipwreck protection zone*).

Several of the 14 currently established Aquatic Reserves in South Australia were designated to protect major fish nursery, feeding and / or spawning habitats, especially mangrove-saltmarsh ecosystems and seagrass-dominated shallows, for coastal fisheries species (Johnson 1988a; Neverauskas and Edyvane 1991; Edyvane 1996b).

Primary reasons for declaration of South Australia's system of MPAs are shown in Table 4. These reasons, to date, have included:

- conservation of species and their associated habitats for commercial and / or recreational fisheries management
- conservation of sites for endangered / vulnerable / rare species (particularly marine mammals)
- protection of sites which are significant for recreation / diving, scientific research and / or education.

Most of South Australia's MPAs serve several purposes as specified in international and national guidelines for MPA establishment (such as protection of critical habitats, and protection of biodiversity at local scales), even though they were usually designated for a single purpose. The small MPA at Goose Island, for example, which was conserved for educational reasons, serves *de facto* functions in the protection of an endangered species, and as a site for recreation / diving.

Table 4: Summary of MPA declarations in South Australia 1971–1998, according to primary purpose

	Conservation of Species and Associated Habitats for Commercial and/or Recreational Fisheries	**Site of Endangered, Vulnerable and/or Rare Species**	**Diving / Recreational Significance**	**Scientific Research or Educational Importance**
Aquatic Reserve	American River Onkaparinga Estuary Barker Inlet-St Kilda St Kilda-Chapman Creek Whyalla-Cowleds Landing Blanche Harbour-Douglas Bank Yatala Harbour	Seal Bay Point Labatt	Aldinga Reef Port Noarlunga Reef Troubridge Hill	West Island Goose Island
Marine Park		Great Australian Bight		

5.2 EXAMPLES OF SUCCESSFUL MPA ESTABLISHMENT

5.2.1 MPAs for fisheries management and enhancement

South Australian MPAs that protect nurseries and feeding grounds for coastal fish species have been considered by the former Department of Fisheries to be an investment in maintaining fisheries resources and, if properly managed, an effective way to ensure the continued exploitation of coastal resources.

The Barker Inlet-St Kilda Aquatic Reserve, for example, which protects shallow marine seagrass and mangrove ecosystems from physical damage, was shown by Jones (1984) to be an important nursery area for a number of major commercially and recreationally-important fish species. These species include King George whiting (*Sillaginodes punctata*), garfish (*Hyporhamphus melanochir),* yellow-eyed mullet (*Aldrichetta forsteri*), Australian salmon (*Arripis truttaceaus*), yellow-fin whiting (*Sillago schomburgkii*) and western king prawns (*Penaeus latisulcatus*). The MPA supplies 'new recruits' to commercial and recreational fisheries along the metropolitan coast, and as far south and west as Cape Jervis, Stansbury and Port Vincent. Barker Inlet is also a spawning area for jumper mullet (*Liza artengeus*) (Jones 1984), and a destination for juvenile tommy ruff (*Arripis georgianus*) which reach South Australian estuaries following the adult spawning events in Western Australia (Kailola 1993).

The samphire, intertidal mangrove and shallow marine seagrass ecosystems in upper Spencer Gulf that have been designated as MPAs (Whyalla-Cowleds Landing, Yatala Harbour, Blanche Harbour-Douglas Bank) are also important fish and crustacean nursery areas, supplying southward-moving populations of fish, prawns and blue crabs throughout Spencer Gulf and beyond (Ivanovici 1984). The American River Aquatic Reserve serves a similar purpose, and juveniles from the estuary serve the fisheries on Kangaroo Island.

There is a 'temporary MPA' at Waterloo Bay, which was closed to fishing at the end of 1995, because selective fishing of larger abalone is believed to have produced a 'stunted stock' in the bay, through genetic selection (Shepherd 1995). This temporary MPA is used as an experimental site for research into abalone population dynamics, and as a site for 'genetic recovery' of the stock, whilst still allowing for irregular harvests of abalone by commercial abalone divers.

The waters around Cape Jaffa, Gleason's Landing, Margaret Brock Reef and Penguin Island in the South East are designated as rock lobster (*Jasus edwardsii*) sanctuaries, to assist management of this commercially and recreationally-important species.

5.2.2 MPAs for ecosystem/habitat protection

Examples of the protection of significant ecosystems / habitat types by South Australian MPAs include:

- The samphire, intertidal mangrove and shallow subtidal seagrass ecosystems of upper Spencer Gulf and Gulf St Vincent (although these are not in pristine condition, and the Barker Inlet Aquatic Reserve, in particular, is subject to chemical pollution and ecosystem changes from thermal effluent, sewage discharge, and heavy metal contamination). Four of the MPAs in SA encompass parts of mangrove ecosystems (see Section 5.2.1).
- The Onkaparinga Estuary Aquatic Reserve, which contains an example of a regionally uncommon ecosystem type; with the exception of the Coorong–Murray mouth region, the Onkaparinga is the largest estuary between south-western Victoria and south-western Western Australia (Ivanovici 1984).
- small patches of reef in some parts of the State, mainly in MPAs that were designated for their interest to divers. The number and size of such MPAs is very small compared with the large extent in South Australian waters of metamorphic basement, sandstone, limestone and granite rocky reef ecosystems, and their great diversity of flora and fauna.

Significant gaps in the system to protect representative examples of marine ecosystem types in South Australia are discussed later in this report.

5.2.3 MPAs for biodiversity conservation

None of the MPAs in South Australia was specifically designated to protect biodiversity, but all play a small role in biodiversity conservation *per se*. As is the case with other States in which MPA development was previously *ad hoc*, 'hotspots' of species richness, as well as both representative and 'unique' examples of ecosystem types for biodiversity conservation (as recommended by the NRSMPA), were not considered as being important reasons for MPA establishment until more recently.

Some of South Australia's small MPAs offer protection for reef biodiversity at small scales. Two small reef MPAs in Gulf St Vincent (Pt Noarlunga and Aldinga reefs) were established in 1971 to protect reef fish species from spear fishing. There is some evidence to show that these MPAs have assisted the return of previously depleted reef species such as magpie perch, leatherjackets and wrasses (Johnson 1988a and 1988b). Troubridge Hill, adjacent to southern Yorke Peninsula, was established in 1983 to protect reef fauna and flora that were attractive to divers, and is now recognised as a site of high benthic biodiversity. Apart from protecting the benthic diversity, Troubridge Hill also provides some protection for numerous species of reef fish that frequent the area (Johnson 1988a), including species that are uncommon in many parts of South Australia.

MPAs in South Australia that prohibit damage to the benthos, as well as the biota in overlying waters of those areas, contribute in some way to biodiversity protection at specific scales. Significant gaps in the system are discussed in Section 5.3.

5.2.4 MPAs for rare/vulnerable/endangered species conservation

The Seal Bay / Seal Beach Aquatic Reserve on Kangaroo island (South Australia) protects a breeding and haul-out site for the rare and endangered Australian Sea Lion (*Neophoca cinerea*), and is one of the largest breeding areas in the world for this species (Robinson and Dennis 1988, cited by Edyvane 1995; GBRMPA 1995).

The Aldinga Reef Aquatic Reserve was originally designated to conserve a limestone reef platform which is popular for diving, and to protect the depleted fish populations from further exploitation by spearfishers. However, it also assists in the protection of locally endangered species such as gorgonian corals (*Melithaea* species) and the blue groper fish (*Achoerodus gouldii*) (Ivanovici 1984). Gorgonian corals have also been recorded at Troubridge Hill Aquatic Reserve (Ivanovici 1984), as well as regionally uncommon fish species such as blue devil fish.

The Blanche Harbour-Douglas Bank Aquatic Reserve in upper Spencer Gulf, which was declared for fish habitat protection and fisheries management, is in a distinct bioregion (IMCRA Technical Working Group 1998) that contains endemic invertebrate species, some with tropical or sub-tropical affinities (Shepherd 1983a and 1983b). These include several coral taxa, some of which are either endemic to upper Spencer Gulf (eg the soft coral *Echinogorgia* species, and the pennatulacean *Scytalium* species), or whose South Australian records were considered to be restricted to upper Spencer Gulf (*Virgularia mirabilis* and the tropical species *Telesto multiflora*). The upper Spencer Gulf bioregion also contains stands of two large brown macroalgal species with tropical affinities (*Sargassum decurrens* and *Hormophysa triqueta*), that are not recorded in other parts of South Australia (Edyvane and Baker 1996b).

Although the American River Aquatic Reserve was designated mainly for protecting the habitat of commercially and recreationally-valued species, part of the reserve helps to protect local populations of rare and possibly endangered endemic pipefish and blenny species.

The Onkaparinga Estuary Aquatic Reserve is a known spawning site for black bream (*Acanthopagrus bucheri*), and although black bream is not rare or endangered, it is a regionally uncommon fish species in South Australia.

Significant gaps in the system of MPAs for protection of threatened species are discussed in Section 5.3.

5.2.5 MPAs for tourism/recreation, education and/or research

Seal Bay and Seal Beach on southern Kangaroo Island is an outstanding example of the success of a MPA in attracting tourism to a local area. The South Australian National Parks and Wildlife has an education program that accompanies organised public visits to Seal Beach. This program has successfully educated thousands of international, national and State visitors per annum, in basic biology and conservation issues of the Australian Sea Lion (*Neophoca cinerea*), which is classified as both rare and endangered.

Examples of MPAs that have important roles in recreation / tourism and marine education include attractive dive sites, such as Port Noarlunga, Aldinga Reef, Troubridge Hill and Goose Island. Port Noarlunga Reef Aquatic Reserve, for example, which was originally designated to protect declining fish species numbers from spearfishing, now has an underwater trail that has been popular with divers since it was established during the mid 1990s.

Recreational fishing is another significant activity in South Australia that is accommodated in some of the current MPAs (eg Barker Inlet-St Kilda; Port Noarlunga-Onkaparinga Estuary).

Few MPAs in South Australia are used for research, although there is one outstanding example. The West Island Aquatic Reserve is the only site in the world in which long-term abalone population dynamics have been monitored, and the site is also used by numerous State, national and international researchers for marine biological and ecological research. The existence of West Island Aquatic Reserve (Shepherd 1991) has spawned many dozens of scientific studies, and has resulted in internationally significant knowledge in abalone population dynamics, fish behaviour, predator-prey interactions, food web studies, benthic ecology, and macroalgal physiology, among other studies.

5.3 GAPS IN SOUTH AUSTRALIA'S COASTAL-MARINE MPA NETWORK

The following sections discuss the current 'gaps' in South Australia's MPA network for near-shore marine bioregions, ecosystems and their component biota. Note that this report does not discuss the use of MPAs to protect and manage offshore ecosystems and the overlying waters, to the limit of State jurisdiction and further into Commonwealth waters. However, it is important to note that a representative system of MPAs would include portions of these offshore environments in each bioregion. Edyvane (1999) discussed the values of, and potential threats to, South Australia's offshore marine environments.

5.3.1 Gaps in bioregional and ecosystems representation

In South Australia, perceived social benefits have shaped MPA identification and selection to date, despite the Commonwealth's endeavour throughout the 1990s, to assist States to identify a more biogeographically and ecologically representative system of MPAs (see Chapter 4).

The table below shows the current proportional representation of MPAs, as a percentage of each IMCRA-defined bioregion in South Australia.

Table 5: Percentage of marine bioregions in South Australia designated as MPAs (excluding rock lobster sanctuaries and terrestrial conservation parks with intertidal components)

IMCRA bioregion	Area of bioregion (km^2)	Percentage of bioregion designated as MPA
Eucla	111 115	10.39
Murat	35 587	0
Eyre	72 165	0.05
Northern Spencer Gulf	4 448	1.76
Spencer Gulf	11 875	< 0.01
Gulf St Vincent	12 838	0.45
Coorong	31 972	<0.01
Otway	37 331	0

(Adapted from Appendix 4, ANZECC TFMPA 1999.)

It is clear that the percentage of each South Australian bioregion protected and managed as MPA is very low for all of SA's bioregions. Some bioregions are not represented at all in the

NRSMPA. Many ecosystem types within all of SA's bioregions are also not represented as MPAs. Edyvane (1996b) discussed the bioregions and ecosystems that are not yet adequately represented under the NSMPA in South Australia. These gaps in the NRSMPA include the Maugean sub-province (Otway Bioregion); the Murat and Eyre Bioregions in the western part of South Australia, and Northern Spencer Gulf (which has three small MPAs designated for the protection of fish habitat). Ecosystem types that are not yet well protected by the MPA system in South Australia include kelp communities, soft-bottom benthos, estuaries, saltmarsh, beach habitats, coastal marine ecosystems that are bordered by wave-exposed cliffs (Edyvane 1996b, Edyvane 1999). The biota of almost all of South Australia's near-shore (to 30 metres) granite, limestone, sandstone, and metamorphic basement reefs is also not represented in the current MPA system. Reefs outside of the gulfs (with the exception of the small MPAs at West Island and Troubridge Hill) have not been protected using MPAs.

Significant ecosystems that are represented in the MPA network, but still subject to numerous pollutants and other impacts that make them vulnerable to continued degradation, include the seagrass ecosystems of Spencer Gulf and Gulf St Vincent (see Section 5.3.5). For example, at least 5000 ha of seagrass has been lost from the metropolitan coast due to sewage effluent and stormwater discharges (DELM 1993). Hart (1997) provided a graphic analysis of the losses over several decades in both area and density of seagrass, in Adelaide's metropolitan waters. The ecological, social and economic significance of protecting southern Australia's seagrass meadows and rocky reefs is discussed in Appendix 2.

Intertidal estuarine and shallow marine ecosystems in sheltered areas of upper Spencer Gulf and upper Gulf St Vincent are quite well represented *within those bioregions*, compared with other bioregions and their ecosystem types in South Australia. Estuaries are an uncommon ecosystem type in South Australia, compared with other States, and only three of the 15 estuaries in this State have been designated as MPAs (Edyvane 1996b). Other estuaries of outstanding conservation significance (Lloyd and Balla 1986) that currently have no protection include Tourville Bay, Little Para River, Tod River, Eight Mile Creek and three estuarine rivers on Kangaroo Island (Stunsail Boom, Harriet and Cygnet) (Edyvane 1996b).

Note that this report does not discuss gaps in the bioregional representation of SA's offshore marine environments (to the limit of State waters and further out into Commonwealth waters), although implementing an NRSMPA will need to include part of these environments.

5.3.2 Gaps in MPAs for fisheries management

To date, South Australia's aquatic reserves have been successful in protecting coastal fish in selected areas from over-exploitation, and providing some protection for their associated habitats. However, it could be argued that most of South Australia's aquatic reserves that were designated for fish stock conservation and fisheries management are either:

- too small and too few in number to protect regional stocks of commercially and recreationally significant species (e.g. West Island and the upper Spencer Gulf reserves)
- subject to numerous pollutants which have reduced habitat quality over many decades (e.g. Barker Inlet; upper Spencer Gulf).
- clustered in limited parts of the upper gulfs, and thus do not provide protection for important habitats and processes that are important for the population dynamics of fish in other parts of the State. Even nursery areas are not adequately protected on a State-wide basis. According to Bucher and Saenger (1989, cited by Edyvane 1995) one third of South Australia's estuaries are considered 'under threat' or 'considerably modified'; 13 of

the 15 estuaries are considered to have 'fisheries value', and most of these estuaries, such as those on the west coast, have no protected status at all.

- provide some protection for juveniles of major fished species, but are not integrated into an effective management system for other life stages of those species. (For example, members of most populations are heavily fished when they move out of their nurseries into other habitats.)
- do not provide adequate protection for fish in habitats other than nursery areas. Species that move southwards out of their nursery areas in the gulfs were formerly partly protected by the 'natural refuges' of less accessible deeper waters, but improvements in fishing technology have meant that areas that were once relatively inaccessible 'refuges' for fish, can now be easily reached and fished.
- do not provide protection for vulnerable 'site-attached' species, such as reef-dwelling snapper, which are now becoming increasingly vulnerable, due to advanced navigation and fish spotting aids (Jones and Luscombe, 1993a and 1993b; Jones, pers. comm.), and increasing harvests of both the old spawning stock (that helps to perpetuate snapper populations in SA), and the new recruits (the 'mega-spawners of the future').
- do not provide adequate protection for sedentary populations such as abalone, which have demonstrably variable population dynamics over their spatial range, and which are declining at local scales due to differential responses to fishing effort.

Furthermore, some of South Australia's coastal fish and crustacean species have reached 'fully exploited' status, and declines in some regions are becoming evident. Fisheries declines are often incremental, starting with the most susceptible parts of populations, whose population dynamics and response to fishing pressure may act as a warning signal for impending larger scale (regional) decline. For example, incremental decline has been shown for a number of abalone metapopulations (Keesing and Baker 1996; Shepherd and Baker 1996). Recent unpublished analyses of long-term catch and effort statistics and survey data indicate that the declines are apparently extending to a significant number of other metapopulations. Long-term regional declines in the snapper fishery were noted for Gulf St Vincent by Jones *et al* (1990) and Rohan *et al* (1991), and recent evidence of declines in the yields from Spencer Gulf, indicates that the current management strategies for snapper may require revision to ensure sustainability in the fishery. Other commercially and recreationally significant species are also showing signs of over-fishing, but annual aggregated catch statistics which cover the entire State (SARDI 1996, 1997, 1998) mask declines within regions.

Current gaps in the use of MPAs to protect and manage offshore fish, crustacean and mollusk species populations will not be discussed in this report.

5.3.3 Gaps in MPAs for conserving vulnerable/rare/endangered species

Compared with terrestrial systems, the extent to which marine flora and fauna are known to be rare, vulnerable or endangered is poorly known, apart from marine mammals and seabirds. Also, rarity is a scale-dependent assessment, since some taxa, for example, may be locally abundant but regionally rare. Much inventory work, through survey and existing data collation, needs to be done to improve the poor state of knowledge in this area. Given the current low level of knowledge, it is likely that many taxa in South Australia that could be considered rare, vulnerable and/or endangered under IUCN classification are not represented at all in MPAs. This includes numerous marine invertebrate taxa, many of which have not yet even been described. In contrast to marine mammals, smaller, less mobile taxa such as the majority of marine invertebrate groups may be better protected by a system of MPAs based upon ecosystem/habitat types. Other omissions in the protection of potentially rare and threatened species include (i) the wide variety of endemic fish, plants and invertebrate taxa found in South Australian waters, of which the conservation status is not known for almost all species; and (ii) ecosystems containing 'unique' species assemblages (the tropical relictual fauna of Upper Spencer Gulf; and the rare brachyopods, corals and large bryozoans of Backstairs Passage, for example). Edyvane (1999) provided detail about the distribution of other known threatened species that are currently not protected using MPAs.

Note that gaps in the protection of threatened biota in offshore environments are not discussed in this report.

5.3.4 Gaps in MPAs for conserving biodiversity

Under the NRSMPA, the representation of bioregions and ecosystem types is being used as a 'surrogate' for biodiversity protection. Given that there are still major gaps in the bioregional representation in SA's current system of MPAs (see Section 5.3.1), the national goal has not yet been met in this State. If 'hotspots' of species richness and endemism are also considered important in biodiversity conservation (and they are under the NRSMPA), then the existing MPA network in South Australia is again not meeting the national goal. For example, there is currently no adequate protection for (i) the species-rich habitats/ecosystems in some parts of South Australia, including rocky reef ecosystems which support globally significant levels of marine plant and invertebrate biodiversity (eg offshore islands of the west coast; reefs of the South East and southern Yorke Peninsula, among other areas: see Edyvane and Baker 1996b, 1996c, 1996d, 1998, 1999a, 1999b, 1999c for descriptions); and (ii) areas of high endemism and 'unique' assemblages of species (see Section 5.3.3).

At local scales, there are numerous pollution threats to South Australia's near-shore marine ecosystems, which have the potential to adversely affect biodiversity (Section 5.3.5). Unless these issues are given management priority, coastal marine MPAs for biodiversity protection are unlikely to be successful in the areas of impact. Note that this report does not discuss gaps in the protection of the biodiversity of South Australia's offshore environments (to the limit of State waters and further out into Commonwealth waters).

5.3.5 Threats to South Australia's near-shore marine ecosystems and biodiversity

South Australia's near-shore marine ecosystems serve significant ecological functions (see Appendix 2 for examples) and maintain biodiversity at all levels. However, there are numerous activities that currently continue to threaten the quality and biodiversity of South Australia's near-shore ecosystems, as described below. It is imperative that planning and management of the NRSMPA in South Australia address these concerns.

Note that this report does not discuss potential threats to off-shore ecosystems and the overlying waters, to the limit of State jurisdiction and further into Commonwealth waters. However, threats to these offshore areas exist, and require cooperative arrangements between State and Commonwealth agencies and stakeholders, to ensure that the impacts of potentially threatening activities (such as deep sea trawling, long-lining and other deep fishing, and mining exploration) are monitored and managed.

Near-shore marine issues that must be considered in the implementation of a NRSMPA in South Australia include:

- continued loss of habitat in (i) seagrass ecosystems, due to sewage and/or stormwater discharges and other pollutants; (ii) mangrove ecosystems, due to land reclamation, industrial developments, and human use (trampling etc); and (iii) saltmarsh, which is being degraded and removed due to agricultural, industrial and urban use and developments (Shepherd *et al* 1989; Steffensen *et al* 1989; Edyvane 1991; Edyvane 1996c;

Lewis *et al* 1998). Seagrass loss is a highly significant issue in South Australia (see Shepherd *et al* 1989; Edyvane 1996c; Lewis *et al* 1998 and Section 5.3.1).

- potential degradation of macroalgal reefs due to sedimentation from combined sources, including particulates from storm water, river catchment outflows, and other land-based discharges. There is some evidence to suggest that turbidity has caused changes to the species composition of near-shore reefs in the metropolitan area (Cheshire *et al* 1997). Damage to reefs from boats and anchors, and SCUBA diving activity, is another potential concern that has not been adequately monitored.
- inadequately controlled 'diffuse source' pollution from urban and rural areas, including (i) solid wastes, litter, domestic and agricultural chemicals, fertilisers, herbicides, pesticides, vehicle pollutants, oils, animal wastes, bacteria, soil and dust (Lewis *et al* 1998).
- ecologically-significant levels of heavy metal contamination (from both point sources and diffuse sources) in sediments and biota, particularly in the upper parts of both gulfs (Ward and Young 1982; Harbison 1984; Ward *et al* 1986; Kemper *et al* 1994).
- by-catch and benthic damage from trawling activities in the gulfs. By-catch in South Australia includes 'trash fish' that are not valued by industry; juveniles of commercially and recreationally significant fish species; blue crabs and other crustaceans; sponges; ascidians; echinoderms; mollusks and other taxa. By-catch ratios are smaller in South Australia than are typical for the tropics, but require more quantification. No objective studies of trawling-induced benthic damage have yet been completed in South Australia, but there is increasing evidence from other temperate ecosystems that benthic trawling can have significant negative impacts on habitat quality and benthic biodiversity (Collie *et al* 1997; Engel and Kvikek 1998; Prena *et al* 1999).
- introduction of pest species in ballast water (eg European fan worm, which competes with other species for space and food, has colonised several marine habitats in the central coastal area of South Australia since the early 1990s).
- the proliferation of inadequately regulated aquaculture operations, which discharge both organic and inorganic pollutants into marine ecosystems. Although very little work has been done in South Australia regarding this issue, some adverse effects on habitat quality and species composition at local scales have been documented (Cheshire et al 1996; Edyvane 1996c). Evidence from interstate and overseas indicates that intensive fish, mollusk and crustacean farming can be highly polluting activities, causing both acute and chronic impacts of physical, chemical and biological natures, in near-shore marine environments (Holmer 1987; Gowan et al 1990; Hansen et al 1990; GESAMP 1991; Ye et al 1991; Hanfman et al 1991; Martin et al 1991; Silvert 1992; Hallerman and Kapuscinski 1992; Sindermann 1992; Gowan and Rosenthal 1993; Roth et al 1993; Chua 1993; Tenore et al 1995).
- potential for acute impacts (from oil spills) and long-term impacts (from small continuous streams of hydrocarbons) on marine ecosystems and biodiversity in areas where oil tankers operate (see Grady 1999 for summary of frequency of oil spills in South Australia).

Edyvane (1996c), Lewis *et al* (1998) and the *South Australian Marine and Estuarine Strategy* (Government of South Australia 1998) discuss the legislative and administrative arrangements in South Australia that can be used to mitigate some of these impacts.

5.3.6 A framework for establishing a representative system

As shown in Sections 5.1, 5.2 and 5.3, the 10-year strategy for identifying biogeographically and ecologically-representative MPAs (under Ocean Rescue 2000 and the NRSMPA: see Chapter 4) is not yet being utilised for MPA planning and declaration in South Australia. There are several reasons for this, which are beyond the scope of this report. However, funding has recently been made available (from the Commonwealth Coasts and Clean Seas Program) to the South Australian Government, to implement the NRSMPA in this State. Also, the State Government's *Marine and Estuarine Strategy for South Australia* has endorsed the bioregional framework of the NRSMPA.

A potential framework for establishing a biogeographically and ecologically representative system of MPAs in South Australia has been documented in the SARDI research series report: *Conserving Marine Biodiversity in South Australia* (Edyvane 1999). This approach follows the Commonwealth directive to establish a representative system of MPAs in South Australia's marine bioregions (classified at national level by the IMCRA Technical Group), by representing areas in the smaller-scale 'biounits' within those bioregions. Under the directive of the Ocean Rescue 2000 Program and the NRSMPA, South Australia's 35 biounits have been classified at State level, from data supplied by a South Australian Marine and Estuarine Protected Areas Working Group, comprising government agency representatives, scientists and other technical experts.

It is clear from the South Australian report on *Conserving Marine Biodiversity in South Australia* (Edyvane 1999) that many currently unprotected areas in South Australia satisfy a number of the NRSMPA's criteria for MPA establishment (see Chapter 4). This includes, in many of the biounits, areas of ecologically significance (including ecologically representative areas and 'unique' ecosystem types); areas of major socio-economic importance; 'hotspots' of biodiversity; and areas containing biota of high conservation significance (endemic, rare, or endangered species, for example). Areas that require representation according to a NRSMPA in South Australia are discussed in Sections 5.3.1, 5.3.2, 5.3.3 and 5.3.4.

A recommended primary target for MPA establishment has been the representation of 10% of each bioregion by 2003, and representation in each of the 35 biounits by 2010 (Edyvane 1999). Potential difficulties with implementing a 'percentage representation' strategy for MPA designation are discussed in Chapter 6.

Implementing the NRSMPA in South Australia would include the designation of 'core areas' of high protection as part of the zoning system. Implementing the NRSMPA would enable the 'gaps' in the South Australia's current MPA system to be filled (see Section 5.3). This could include for example, fisheries MPAs (ie reserves); a more representative spread of MPAs for the protection of important coastal-marine ecosystem types (including regionally uncommon ecosystem types); MPAs for the protection of threatened species and habitats; some MPAs designated according to explicit criteria for biodiversity protection; and sites for scientific research and monitoring. Similar to a biosphere reserve or other multiple-use marine park system (Section 3.2.1), limited use zones outside the 'core' protected areas would continue to permit a variety of non-destructive activities. The third major zoning type (general use zones) would continue to accommodate a variety of industrial, commercial and recreational activities, ensuring that they do not compromise the integrity of the protected 'core' areas.

6 MPA DESIGN AND SELECTION: NUMEROUS APPROACHES, NO CONSENSUS

It is probably not possible to implement a standard, globally applicable set of design criteria to assist MPA establishment, due to:

- the fact that there are many goals of MPA establishment, such as bioregional and ecological representation, 'biodiversity protection' (which may be defined in many different ways), fisheries resource protection and enhancement, protection of critical habitats, endangered species conservation, protection of socially significant sites, among others. The 'optimum' design criteria for attaining each goal may not be compatible with other goals. For example, many reserve design criteria for both fisheries MPAs (Section **3**) and protection of endangered species, are often species-specific, and therefore based upon life history phases, population dynamics and demography of the species, their larval movement patterns, their habitat linkages, and their response to fishing. The design of potential MPA sites that can satisfy such species-specific objectives such as stock protection and management, do not always coincide with the same set of sites that would be chosen for an ecologically representative system of MPAs.
- the differences in the size, shape, distribution and dynamics of biogeographic regions, 'biounits' or 'ecounits', and ecosystems / habitats between coastal marine areas around the world, as well as the wide variety of different (and site specific) physical conditions and oceanographic factors affecting potential MPA sites in each region
- the great differences between countries in social and political systems, and in both natural and human-induced threats to MPA success, which can influence placement of MPAs
- the variety of different methods that have been used for MPA design within and between countries, including biogeographic and ecological approaches, single species approaches (particularly for fisheries MPAs and reserves for endangered species), 'Delphic' approaches (see Section 6.1.4), multiple-attribute mapping approaches, mathematical methods based upon terrestrial conservation planning principles, and *ad hoc* approaches, among others
- the conceptual and practical difficulties in achieving world-wide consensus about ways to impose essentially 'artificial' boundaries around (i) ecosystems whose spatial dimensions, linkages and biodiversity are often poorly known; and (ii) areas of water which are essentially continuous (despite discontinuities in the habitats and biota that underlie them), and whose hydrodynamics and productivity over space and time are also poorly understood.

To date, design of MPAs has involved several approaches that may be divided into the following broad categories:

MPA designs for biogeographic and ecosystem representation. This approach often incorporates the principles of landscape ecology and ecological hierarchy theory, and utilises spatial tools such as GIS, and multivariate and spatial statistics. These tools are used to order, categorise and map multivariate physical and biological marine survey data into 'representative' groups. Remotely sensed images and aerial photographs (for shallow marine areas where light penetration is adequate for resolution) have also been used in some of these classifications. Both biological / ecological survey data and physical 'surrogate' data have been used to define ecosystems and communities. Following the delineation of ecological units at various scales, other methods may be used to identify a 'representative' set of potential

MPAs, so that each unit is represented in the network. Some of these methods are discussed in the sections below on reserve design algorithms and decision modelling.

MPA designs based upon single-species population models and meta-population theory. This approach is spawning an increasingly large body of theoretical work, particularly reserve design models which seeks to optimise MPA size, shape, number and placement for the conservation and management of commercially or recreationally-significant species.

MPA designs based upon reserve selection algorithms. This approach originated in terrestrial systems, and utilises various iterative algorithms and other programming formulae to choose the optimum number, size and placement of MPAs, to satisfy a number of defined conservation criteria.

MPA designs based upon decision modelling. This includes GIS-based and/or model-based multi-attribute decision modelling, using either or both of (i) rule-based and arithmetic criteria in GIS modelling; or (ii) heuristic or mathematical optimisation methods such as those used in reserve design models of the type outlined above. Ranking methods and/or multivariate analyses have sometimes been used to classify the layers of spatial data that are used for input into such models.

Ad hoc MPA designs: size, placement and number of MPAs determined solely by socio-political factors, location of endangered species, attractive diving locations, practicality (eg marine extensions of terrestrial parks) and/or position of threatened habitat types (eg estuarine fish nurseries).

A summary of the approaches is provided below.

6.1 MPA DESIGNS FOR BIOGEOGRAPHIC AND ECOSYSTEM/HABITAT REPRESENTATION

6.1.1 International and national overview of mapping and classification

The biogeographic/ecosystem/habitat mapping approach to MPA design was first recommended by Ray (1975), who developed the concept of a hierarchical classification of coastal and marine environments, to assist marine conservation, research and monitoring. Ray and McCormick-Ray (1992) further refined this approach, and argued for the representation of biogeographic types in a comprehensive protected area system. Since that time, the hierarchical biogeographical ('bioregional') mapping approach, has been consistently recommended for the development of a national representative system of MPAs in Australia (Muldoon 1995; Thackway 1996; Thackway and Cresswell 1996; ANZECC TFMPA 1998 and 1999).

As with the creation of MPAs for fisheries management, work in New Zealand pre-empted the rise in the popularity of the bioregionalisation approach to MPA design. Ballantine (1990) discussed the biogeographic regions and coastal ecosystem types in New Zealand, and provided recommendations for MPA number and placement, based upon 'minimum set' representation of all combinations of bioregion and coastal type, which are currently referred to in New Zealand as 'ecological districts', based upon a biophysical classification. This classification of coastal marine types is based upon topographic features, geomorphic processes, bathymetry and sediments, and floral and faunal composition and abundance.

In several other countries, coastal marine benthic environments have been mapped using a combination of remote sensing technologies, aerial photography, GIS, and field-based survey methods. Examples include:

- the mapping of Caribbean benthic ecosystem types using (i) digitised and classified stereo pairs of aerial photographs (Sheppard *et al* 1995; Gill 1995); and (ii) commercially available remote sensors and / or digital airborne multi-spectral imaging (Luczkovich *et al* 1993; Mumby *et al* 1994 and 1995), combined with survey-based multivariate classification (Mumby and Harbourne 1999)
- in Florida, the mapping of manatee habitat (Reynolds and Haddad 1990; O'Shea and Kochman 1990), the modelling of 'essential fish habitat' (Rubec *et al* 1998), and the mapping of benthic communities / habitats in the Florida Keys
- advances in techniques for remote sensing of benthic ecosystems / habitat types, and for GIS-based classification of types (Bakker and van Kootwijk 1993; Deysher 1993; Pasqualini *et al* 1997).

Jones (1995) provided other examples of remote sensing and GIS technologies that have been used to map coastal marine environments in parts of the U.K., Europe and North America.

In Australia, the mapping of major biogeographic regions around the coastline has been undertaken as part of the Interim Marine and Coastal Regionalisation of Australia (IMCRA) (IMCRA Technical Group 1998), and is described in Section 4.3.2 of this report. At both 'bioregional' and smaller scales (ie 'biounits': Ortiz and Pollard 1992), researchers from several States have been mapping and / or classifying coastal marine biophysical data, marine ecosystems / habitats, and their component biota. This project has formed part of the 10-year Commonwealth directive to establish a national, biogeographically and ecologically representative system of marine protected areas (discussed in Chapter 4 of this report). Techniques of State-level bioregionalisation are discussed in the following section. More recent marine environmental classification work in Australia (that was conducted independent of the NRSMPA objectives), is the development of the Royal Australian Navy's laser airborne depth sounder, which has been used to map reef, sand and other benthic features of the Great Barrier Reef in 3D, to a depth of 50 metres (Abbott 1997). Other developments in marine benthic mapping that have originated in Australia include AGSO's work on depth-correction algorithms for classifying benthic imagery (Bierworth 1996).

Similar to the marine bioregional classifications used in Australia, a biogeographic classification system is now also used in British Columbia, to divide marine areas into 'ecosections' and smaller 'ecounits', based upon biophysical characteristics (Zacharias and Howes 1998; Zacharias *et al* 1998). This type of categorisation permits the current MPA system to be critically examined for adequacy and gaps, relative to the goal to proportionally represent each ecosection and habitat type (in each ecounit) in the MPA system.

Biogeographic zones and 'ecotypes' have also been mapped in South Africa, and fisheries researchers and coastal managers in that country have recommended the representation of biogeographic regions and ecotypes in MPAs, as one of the primary methods of conserving biodiversity (Attwood *et al* 1997). Other biogeographic work in this region that was carried out to assist the placement of MPAs has included a classification of shores and shallow water areas along the West Coast of South Africa (Emanuel *et al* 1992). The work entailed multivariate analyses of zoogeographic data (for 2000 different species), and mapping of coastal geomorphological units, based upon classification of aerial photographs.

The biogeographic approach to MPA design has also been recommended in California (Starr and Johnson 1997), as a way of 'integrating' and better representing different ecological regions in the currently disjointed MPA system. The current *ad hoc* MPA system in California comprises more than 100 parks, reserves and sanctuaries, which have been developed by a number of agencies, with each MPA having a different set of goals. The attributes of California's MPAs and have recently been mapped using GIS (McArdle 1997), which will enable planners and managers to determine biogeographical and ecological gaps in the current MPA network. For example, Bushing (1997) evaluated the ecological representativeness of a network of island MPAs in southern California using a GIS 'gap analysis'. This approach used physical and biological data layers (eg bottom relief, slope, wave height, kelp distribution) to consider the variation in regional ecology, disturbance regime, and meta-population dynamics of giant kelp, the major structural element in the ecosystem. The analysis also aimed to identify current 'gaps' in the protection of this major ecosystem type.

6.1.2 Bioregionalisation as a basis for MPA design

'Bioregionalisation' is a method of marine classification to assist identification of potential MPAs, and was recommended by the Commonwealth in the mid-1990s under the Ocean Rescue 2000 Program (Muldoon 1995). Bioregionalisation could be defined as the development of a nationally-agreed biogeographic classification scheme, at hierarchical scales, of marine areas whose 'boundaries' are clearly defined and mapped. The process is designed (in theory) to identify the full range of biogeographic regions and ecosystems in Australia's coastal marine zone, so that representative examples can be chosen as part of the National Representative System of MPAs (see Chapter 4).

Physical data sets incorporated into bioregionalisations include oceanographic data (temperature, salinity, currents, nutrients and upwelling zones); bathymetry; sediments and substrate geology and coastal geomorphological features; and climate (wind patterns, tides, wave patterns, rainfall, etc). Biological data sets may include (i) distribution of 'key species', fish and 'megafauna'; (ii) dominant and 'indicator' species for ecosystems; (iii) ecosystem / habitat maps (often delineated using aerial photography and remote sensed data in addition to survey data); and (iv) 'communities' or major 'assemblage' types. Numerous Commonwealth and State level organisations have contributed data to the bioregionalisation process in Australia (see Muldoon 1995; IMCRA Technical Group 1998).

Technically, bioregionalisation involves the production of GIS maps and / or spatial classifications, which combine the spatial pattern and variability in physical base maps (collated and classified from the data sets described above) with corresponding pattern in biotic groupings (also described above). These classifications and maps are produced at hierarchical scales, from biogeographic regions down to local 'biounit' (and sometimes ecosystem) levels. Ideally, the physical maps provide the structural base, in terms of size and shape of bioregional units, upon which biotic distributions can be plotted.

Area classifications under the bioregionalisation program aim to identify areas that represent the full variety of marine, estuarine and coastal environments, at hierarchical scales, so that MPA identification may proceed in a more biogeographically-inclusive and systematic way. Bioregionalisation is also considered to be a 'tool for conserving biodiversity' (Muldoon and Gilles 1995).

There is no doubt that bioregionalisation procedures can assist the MPA design process. Many decisions are being made about potential MPA sites, as well as coastal marine activity and resource use planning, without sufficient information about the variety of biogeographic regions and ecosystem types, the biota within these areas, and their vulnerabilities. Bioregionalisation can thus provide a more systematic, objective approach to conservation planning, including identification of potential MPAs.

6.1.3 State-level bioregionalisations

During the 1990s, progress in marine bioregionalisation was made in New South Wales, South Australia, Tasmania and Western Australia, as part of the Ocean Rescue 2000 Program (see Chapter 4).

In New South Wales, the method of bioregionalisation involved the production of an hierarchically scaled, three-tiered model of coastal, estuarine and marine environments along the entire NSW coastline (Ortiz and Burchmore 1992; Ortiz and Pollard 1995). The classification utilised a number of climatological, physiographic, oceanographic and biological data sets. The empirical data were classified with multivariate procedures to define the hierarchy of bioregions, which represented physical and biological processes operating at different spatial and temporal scales (Ortiz and Burchmore 1992). The NSW bioregionalisation revealed that only a small proportion of the range of coastal, estuarine and marine environments are were represented in MPAs during the 1990s, and that the majority (more than 60%) were concentrated in one bioregion (Ortiz and Pollard 1995).

The Tasmanian bioregionalisation was based upon State-wide surveys of reefs, which involved the collation of both existing data and purposely-collected survey data on the distribution of fish, invertebrates and marine plants, as well as physical data sets such as bathymetry and sea surface temperature. Multivariate analyses and GIS-based spatial statistics were used in the classification process (including an 'edge detection' method to identify gradients in species composition along the coast (Edgar *et al* 1995, Edgar *et al* 1997).

In South Australia, physical and biological data collated from participants in a Marine Protected Areas Working Group (which used the so-called 'Delphic' approach, as outlined in Section 6.1.4 below) provided the basis for the State level bioregionalisation process (Edyvane 1999). Statewide benthic surveys of the distribution and biomass of marine plants and sessile invertebrates (Edyvane and Baker 1995, 1996a, 1996b, 1996c, 1998, 1999a, 1999b, 1999c) also contributed to the bioregionalisation process.

In Western Australia, coastal geomorphology, bathymetry and other water mass data, geological maps, aerial photos and remotely sensed imagery, dominant biota, and other data from geomorphologists and biologists, were used in a 'geomorphic' analysis of regions (Wilson 1995). This procedure involved the 'Delphic' approach (see below) as well as more objective methods of analysis, and resulted in a hierarchical classification of 'habitat units' for each 'major coastal type' (such as island archipelagos, mangal stretches, gulfs, estuaries). Examples of habitat units (based mainly on coastal geomorphology) included (i) within mangals: ria shore, delta, gulf and sheltered bay ecosystems; (ii) within coral reef systems: atolls, barrier reefs, fringing reefs. Within habitat units, the categorisation of assemblage types was the finest level of classification (eg within coral reef habitat units: lagoonal, back reef, reef flat, reef-front slope) (Wilson 1995).

In Queensland, an initial 'meso-scale' (100s of km) bioregionalisation was undertaken during the 1990s (Stevens 1995), based upon existing published physical and biological data. The limitations of using available data were recognised by Stevens (1995), who recommended further survey work and the inclusion of a greater number of biological data sets, to improve the quality of the classification. More recently, GBRMPA in Queensland has commenced a five-year program to identify and select a range of representative habitats and ecosystems in the Great Barrier Reef, for inclusion in the NRSMPA. Although the classification process is based mainly upon physical data sets, it will ensure that a more even representation of habitats and ecosystem types are protected in the GBRMP, rather than limiting the 'protected' parts of multiple-use MPAs to dominant ecosystem types (coral reefs, in this case) or single, high-profile species (GBRMPA 1999).

6.1.4 The role of the 'delphic approach' in bioregionalisation

In South Australia and Western Australia, groups of experts in coastal marine oceanography, physiography, geomorphology, geology, biology and ecology, were involved in the State-scale bioregionalisation process. Experience and knowledge are used in the so-called 'Delphic approach' to arrive at a consensus about how regions should be classified (McArdle 1995). In South Australia, one of the aims of this approach was to develop biophysical classification units (eg 'biounits') within the IMCRA bioregions, and to determine areas of conservation significance within those units, using multiple ecological, social and economic criteria, to assist the development of the NRSMPA at a State level. Much of the State-level 'biounit' classification, which is currently being used in South Australia for MPA planning, was based upon the opinions of scientific experts, regarding the conservation value of marine features and biota in particular locations, rather than the use of more objective, systematic, and spatially inclusive data. Such data were not available at the time for many parts of the near-shore marine environment in South Australia.

There are some limitations and biases in the Delphic process, particularly when limited information from the experts, or from State-level marine literature and data sets, is available for particular areas that are included in the bioregionalisation process.

However, at a national level, the Delphic approach was considered a desirable complement to more 'objective' methods of bioregionalisation based upon data analysis. There are several reasons for this, including the facts that:

- MPA design cannot practically proceed using only biogeographic criteria
- the required data for more scientific methods of area classification are rarely available at a State-wide level, particularly biological data
- Delphic workshops can be used to help identify which areas need to be 'targeted' for more objective survey/data gathering and analysis, thus improving bioregional classifications
- the consequent MPA selection process usually involves 'Delphic' decision-making according to multiple conservation, social and economic purposes (Wilson 1995; McArdle 1995).

The multiple purposes mentioned in the last point above have been discussed in detail in other sections of this report, and include:

- 'conservation' (an umbrella term encompassing the maintenance of biodiversity and all its features, such as richness, rarity, endemism, etc, protection of breeding sites, conservation of endangered species, protection of ecosystems/habitats, etc)
- protection of commercially and recreationally-significant species
- protection of areas that are important for recreation and tourism
- protection of areas from increased human impact
- protection of sites for research and reference/monitoring of environmental and/or human-induced impacts and change.

As noted by Wilson (1995), biogeographic criteria are obviously inadequate to identify all such areas.

6.1.5 Problems with bioregionalisation

Bioregionalisation is a suitable technique for classifying and mapping large spatial scale units, but cannot realistically be used to define the smaller spatial scale variation over space and time in habitats and their component flora and fauna (Thackway 1995). In other words, bioregionalisation cannot practically be undertaken at scales that can identify the full range of habitat types along the entire Australian coastline, and it is at the small spatial scale of habitats that conservation and management decisions are made (such as placement of MPAs). Even at the larger spatial scales of regionalisation, biogeographic zones change geographically over time for example, due to changes in current patterns and temperatures, which can blur the distinct 'boundaries' given to such regions (Hamilton and Cocks 1995). Bioregionalisation is also useful for classifying units that assist marine management, but it may be difficult to link these units with ecological process and hierarchy theories (Cousins 1993, cited by Hamilton and Cocks 1995).

Bioregional classifications are not usually adequate for identifying areas of conservation, social and economic significance, for inclusion in the MPA planning process. One of the other major difficulties with the bioregionalisation approach was, and still is, the patchy nature of the data sets used, and the biases and omissions in particular data sets. Examples include the spatially and temporally uneven distribution in data such as fishing records, museum records, bathymetric data etc, and the sampling errors, omissions and non-representative nature of much of the available marine biological survey data.

Another concern is the lack of correspondence between the physical and biological patterns observed in bioregionalisation. There are several reasons for this, including:

- the lack of spatial biological data that is used in bioregionalisation, compared with physical data
- the fact that biological distributions often cross the 'boundaries' that can be defined by the distribution of physical features in marine systems, due to obvious reasons such as (i) dispersal and migration of marine biota through water; and (ii) the variability of marine plants and sessile benthos growing on the same type of substrate, and vice versa (Davidson and Chadderton 1994; SARDI, unpublished data 1993–1998).

Thirdly, defining 'boundaries' between parts of the marine environment at any scale is contentious, given the gradationary (rather than abrupt) nature of change between definable marine systems. Other technical concerns with the bioregionalisation process are discussed in Hamilton and Cocks (1995) and Muldoon (1995).

Lastly, there is a lack of standardisation between techniques used in each State. Various GIS-based techniques (such as gap analysis and 'ecotone' analysis: Peters 1990, cited by Edgar *et al* 1997) and multivariate clustering and ordination procedures (cluster analysis, multidimensional scaling, correspondence analysis) have been used in several Australian States, to classify the large data sets used in bioregionalisation. It would be useful to standardise between States the scales of bioregionalisation, the units of measure, the biotic content, and the techniques of analysis. However, standardisation between States may not be practical, given the differences in availability of data, and conflicting opinions regarding the suitability of the various technical methods used in bioregionalisation.

6.1.6 Design criteria for representing biounits, ecosystems and communities

The desired locations of MPAs to achieve conservation goals such as representation of biounits, ecosystems and communities can be assisted by ecological and habitat mapping work. Small spatial scale ecosystem/habitat mapping techniques, which are even more variable than large spatial scale methods of bioregionalisation, will not be discussed in this report.

It is worth noting that the ‘optimal’ sizes and locations of MPAs to represent the ‘biounits’, ecosystems and communities that are typical of a bioregion may conflict with the placement of MPAs that satisfy other objectives. For example, headlands may be important in retaining larvae of some species (Kripke and Fujita 1998), and thus the designation of a headland within a biounit could satisfy a fisheries management or enhancement objective. However, an adjacent bay may contain, for example, either a rare community type, or a far more diverse assemblage of biota compared with the headland. Therefore, preference of the former for MPA designation would not be the best choice for satisfying other MPA objectives, such as the conservation of biodiversity or rare habitats.

Early work in MPA design theory for ecosystem / habitat protection came from Bryceson (1981), who considered that ‘several small’ MPAs were preferable for some tropical systems, because distinct ecosystems / habitat types were ‘patchy’ (eg coral heads, mangrove stands), heterogeneous in composition, of variable shape (eg circular compared with elongated), and richer in biodiversity at their edges (‘ecotones’) compared with their interiors.

Regarding protection of critical habitat, there can obviously be no recommended ‘standard’ for MPA size, because habitat size, quality and connectivity are so variable in different marine ecosystems, in different countries. For example, a smaller MPA in an undisturbed, resource-rich habitat may be more effective than a large MPA which is (i) subject to numerous polluting influences which reduce habitat quality, or (ii) naturally low in resources (food, shelter, suitable substrate etc) and thus of low amenity to many species populations. The numbers, as well as sizes, of habitat patches that are protected in the MPA system are also important to consider.

There are many suggestions about appropriate MPA size, according to the goals of ‘representativeness’ at each scale. The problems associated with selecting MPAs based upon proportional representation of bioregions / biounits / ecosystems / habitats are discussed in the following section.

It is generally accepted that in order for MPAs to protect habitat, shape should be aligned with the natural geometry of habitat types and fragments. Important habitat types worthy of MPA designation may naturally be small (eg a rock reef), or narrow (an estuary, or a highly-productive boundary between two ecosystem types) or patchy (seagrass beds).

The relation between critical habitat and fish population dynamics and demography has also been examined in MPA design, and examples are provided in Section 6.2.2.

‘Whole system’ management (see Chapter 9) is obviously also important in planning the location of MPAs, especially for MPAs with ecosystem / habitat protection objectives. Location of actual and potential impacts must be considered in MPA design, because coastal marine habitats and species cannot be effectively protected without effective management and regulation of activities and impacts. These include both coastal building (marinas, seawalls, etc) and coastal marine pollutants (point source outfalls, poorly managed aquaculture facilities, and diffuse sources such as stormwater and urban and rural, runoff, etc).

6.1.7 The need for a network approach

Modern approaches to MPA design emphasise the need for networks of reserves, which can serve multiple purposes such as biodiversity conservation, ecosystem / habitat protection over space and time, fisheries enhancement and management, protection against periodic impacts, and numerous social benefits. The benefits of MPA networks have been debated since the

1980s (eg Kenchington 1988; Salm 1989), but few countries have developed networks to demonstrate the purported benefits that have pervaded the theory of reserve design.

In theory, MPA networks can be spaced to include all the critical habitats in a region, due to the broad spatial distribution of different habitats. By protecting a greater range of habitats and resources for various species, competition for space and resources is reduced, and population persistence is promoted. Furthermore, a network of protected areas can maintain a sufficient number of 'habitat patches' to promote recolonisation following local population decline due to demographic or environmental stochasticity (Verboom *et al* 1991). Networks of MPAs also assist an 'ecosystems' approach to coastal marine management (see Chapter 9.1) by protecting a mosaic of inter-connected ecosystem types/habitats, and their component biota. Furthermore, network designs enable replication of the representation of ecosystems/habitats. Replication is particularly important in areas that are subject to natural catastrophe and/or high levels of human-induced impact, and to assist scientific monitoring of MPA performance (Fairweather and McNeill 1993; Ballantine 1999).

In practical terms, if resources are available for managing all of the small MPAs, these areas may be easier to protect from damaging influences (poaching, physical destruction, etc) because each MPA has set 'boundaries' for management, and there are thus fewer 'entry points' for pollutants, illegal harvesting, etc.

In New Zealand, *networks of reserves* are favoured to maximise the number and variety of 'connections' (eg to permit continued movement of larvae and fish between MPAs), and to incorporate a variety of ecosystems/habitat types (Ballantine 1990, 1991a, 1991b, 1999). Ballantine (1990) argued that 'large' MPAs are not necessarily the best for satisfying a number of purposes, considering the dispersive phases of marine organisms, the variety of life history strategies in any one area, the need for networks to represent different bioregions and habitat types (see Section 6.1), as well as social and practical limitations.

Networks of MPAs that protect a variety of ecosystems (and communities/assemblage types, where possible) overcome the problems associated with optimising reserve design for single species, which has been the favoured approach of many scientists, but is not well supported by management and the public. A network approach, in which examples of all ecosystems/habitats are protected, is likely to be more successful in serving other goals in addition to fisheries management, compared with the long-debated academic argument about where to site individual MPAs to optimise their benefits for single species. Roberts (1998) discussed this important point in detail, reminding workers that any one area can act as both a source and a sink for a variety of species (see Section 6.2.2).

Regarding the role of MPAs in maintaining biodiversity, 'wholesale' biodiversity protection is obviously impossible, in coastal marine environments that are subject to numerous activities and impacts. However, the high protection of 'core' examples of ecosystems within large multiple-use MPAs is now considered to be an effective method of biodiversity conservation, as discussed in other parts of this report, and networks of small, high protection MPAs are becoming an increasingly popular conservation method on a global scale. Realistically, there can be no specific guide to the size and number of small MPAs required for biodiversity protection, although, according to Ballantine (1999), the entire MPA network should be large enough to sustain the natural processes operating in the region.

Obviously, there can be no standard, recommended size for MPAs in a network that aims to represent ecosystems/habitats and their component biodiversity, as discussed in the first section of this chapter. In contrast with modellers who aim to 'optimise' reserve network design, some workers (eg Ballantine 1999) consider that there may be no single 'best' network design to optimise benefits, and that a variety of different network configurations should be possible, depending upon the area and the circumstances.

As a general guide to size of multiple-benefit networks, Ballantine (1990) suggested that in most cases, a few square kilometres would be a minimum target size for MPAs to be useful, and that MPAs (called reserves in New Zealand) should be sized and spaced as part of an overall network. An existing MPA of five square km (the Leigh Reserve) serves as a useful example of the size of reserves that are considered for the network. The Leigh Reserve has provided many benefits other than replenishing fish stocks in that area. Apart from the well-documented fisheries benefits, that small MPA has played a role in 'habitat restoration', and primary production and kelp have both increased in the MPA since its declaration in 1977, amongst many other benefits (especially social ones) that are mentioned in several chapters of this report (Ballantine 1999).

More recently, Ballantine (1999) has suggested that a series of 'tiny' reserves would be inadequate to sustain natural processes and conserve biodiversity in the long term, even if there was a large number of such small MPAs. This suggests a critical minimum size for MPAs in a network, but there are few guides to sizing that are internationally applicable. The critical size of network MPAs will differ according to many factors, including the size and distribution of ecosystems/habitats, the degree of threat to those ecosystems, the existing uses and activities in an area, and pre-existing *ad hoc* MPAs that may form part of the network, among other considerations. Ballantine (1999) considered that the point of networks is not the specific size or number of MPAs in the network, but the size of the overall network, as a proportion of the total marine area in a country.

In recommendations for *shape* of MPAs (presumably in networks), researchers in South Africa considered biogeographic patterns rather than habitat requirements of species. Hockey and Branch (1994) proposed a 'middle/edge' arrangement of biodiversity reserves, linked to biogeographic regions, on the premise that reserve 'edges' represented areas of high biodiversity, and the 'middle' of reserves contained the 'representative' habitats and their component biodiversity. Although representation of biodiversity was the major objective in Hockey and Branch's (1994) design recommendation, they also suggested a 'second tier' of MPAs within the biogeographic regions, that were designed to protect areas of high productivity and improve yields of local species (ie fisheries MPAs).

6.1.8 Proportional representation of biounits/ecosystems/habitats: potential problems

The World Conservation Union has recommended that a minimum of 10% of every nation's marine environment be protected as a minimum assurance of 'ecological sustainability' (GBRMPA 1995). This approach is also being used in the South China Sea, where 20% of coral reef areas have been proposed as 'core' protected areas within strategically-placed, larger, multiple-use MPAs. Similarly, in North America, the National Academy of Science Report on sustaining marine fisheries in North America suggests 20% of the entire coastal area is a suitable total size for MPA establishment (Wilder, unpublished comment to California Marine Protected Areas Network, 1999).

Proportional ecosystem/habitat representation has been recommended for South Australian MPAs (Edyvane 1999).

In theory, proportional representation of areas at a given scale (from bioregions down to ecosystems/habitats) would protect the life history requirements of many exploited and unexploited species; protect biodiversity *per se*, and conserve critical habitats and ecosystem processes, rather than optimising MPA design for the protection and/or management of any specific species.

However, the MPA design approach that calls for proportional representation of each biogeographical region and the ecosystem types within them, has been criticised as inadequate and difficult to practically implement for the protection of terrestrial biodiversity and ecosystems (HORSERA 1993). The same concerns apply to use of that approach in marine systems.

For example, the 'real estate' method of selecting MPAs for ecosystem and biodiversity protection could be considered to:

- not have an established scientific basis, nor be proven efficient or effective by experience, despite its common recommendation by international conservation organisations
- not adequately provide for the conservation of a range of communities or habitat types, such as those that are rare, fragmented or highly susceptible to disturbance or degradation
- provide unequal levels of protection for some ecosystem types, since 10% of a very small, rare ecosystem type cannot be compared with 10% of a large, widely distributed, 'common' ecosystem type (note that 'common' or 'rare' are also subjective judgements according to the scale of analysis, and the composition of those ecosystem types, eg different examples of a single, common 'ecosystem type', which has been classified by biophysical procedures, may actually contain different biota
- not be achievable for some ecosystem types and the populations within them, of which the target level (eg 10% or 15%) might not be available for inclusion in the MPA network
- be overly simplistic for biodiversity conservation, because conserving 10% of an ecosystem type is unlikely to conserve representative examples of the biodiversity in the entire ecosystem (ie biodiversity occurs on many levels and at many spatial scales)
- add an element of inflexibility to the MPA network in the future; for example, knowledge of biodiversity in particular areas is likely to improve over time, but proportional representation of areas may foreclose future options of adding areas of high biodiversity to the MPA network
- foreclose options of improving the 'representativeness' of the system; for example, if one or a few large MPAs per bioregion is designated (taking up the '10%'), other ecosystem types within a bioregion may not have a chance of being represented in the MPA system
- increase the risk that the MPA design process can be manipulated by pressure groups, so that less 'worthy' areas (ie the degraded 'leftovers') are chosen because the satisfy the numerical target (eg 10%).

There are obvious problems when deciding which percentage of an area (eg bioregion) should be conserved, especially if the proportional approach is mistakenly combined with a single species conservation objective. In such cases, options become limited for a more 'equal' representation in future of the ecosystems/habitat types within that bioregion. For example, if '15% of a bioregion' is chosen as a conservation objective, and the 15% chosen is mainly bare sand, for the protection of a single, socially significant species and its breeding grounds (eg whale), then other highly significant ecosystem types within that bioregion (reefs, seagrass meadows etc) may have very limited chance of later protection in the MPA system.

Ballantine (1990) highlighted the difficulty of deciding what 'representative' actually means in terms of protecting bioregions and ecosystem types. In other words, considering the number of different ecosystems in bioregions and biounits, which of the many portions of a biounit is the most 'typical' of that unit, in terms of its physical and biological characteristics? This is a difficult question that cannot easily be answered, even with vast amounts of data.

Another practical concern with proportional representation is the calculation of MPA length. For example, '10% of the coastal length' will differ according to the scale at which the coast is measured, the fractal geometry of the coast, and the measurement units used. The practical significance of this example is the difficulty of (say) spacing MPAs at 100 km intervals, for coasts that are not linear (Ballantine 1990).

Deciding which proportion of each bioregion and biounit to be protected ideally requires knowledge of the diversity of ecosystems/habitat types within those units (especially if protection of habitat diversity is being used as a surrogate for biodiversity conservation at lower levels). At the habitat level, degree of threat and changes to habitat size and quality over time (due to both natural events and human-induced changes) are also important in the selection process.

6.2 MPA DESIGNS BASED UPON SINGLE-SPECIES POPULATION MODELS AND META-POPULATION THEORY

6.2.1 Background

Beverton and Holt (1957, cited by Guenette *et al* 1998) first suggested the use of models to assess the benefits of 'closed areas' for stock management, but the idea remained unpopular for several decades. Sluczanowski's work (1984, 1986) on closed areas for abalone fishery management preempted the current rise in popularity of marine reserve models for conserving and managing commercial species. During the early 1990s, there were comparatively few applications of population dynamics models to the design of MPAs for fished species (eg Polacheck 1990 for fish with cod-like population parameters; Die and Watson 1992 for prawns; Quinn *et al* 1993 and Rogers-Benett *et al* 1995 for sea urchins; DeMartini 1993 and Man *et al* 1995 for coral reef fish; Holland and Brazee 1996 for snapper).

However, marine reserve design modelling for fisheries MPAs has now become an active research area, particularly in North America. Recent examples include the work of Mangel 1998; Lauck *et al* 1998; Foran and Fujita 1999; Sladek Nowlis and Roberts 1995, 1997, 1999; Appeldoorn and Recksiek 1998 (unpublished); Mangel in review; St Mary *et al* in review; Stockhausen *et al* in review; among others. Apart from the global recognition of fish stock declines under current management regimes (see Chapters 3 and 10), the recent research activity in MPA design in the US may have been prompted by the reauthorisation of the Magnuson-Stevens Fisheries Conservation and Management Act in 1996, which recommended establishment of 'no-take' MPAs to assist fisheries management. Two major international conferences on marine reserve design were held in the US and Canada during the late 1990s: the Mote International Symposium in 1998, to discuss reserve design methods for protecting essential fish habitat and coastal marine fish species, and the University of British Columbia's 1997 conference on scientific principles for the Design and Monitoring of Marine Reserves (see Pitcher 1997).

Guenette *et al* (1998) provided a detailed review of models developed during the 1990s for fisheries MPAs, which specified reserve design criteria to maximise a number of stock and fisheries benefits, such as migration of fish or larvae out of the MPA; increased fisheries yield; or protection of stock size or numbers from the effects of over-fishing.

In addition to the modelling studies, criteria/principles for the design of MPAs to conserve and manage fished populations have been discussed since the early 1990s, based upon meta-population dynamics, modes and patterns of larval dispersal, and/or habitat linkages of life phases, amongst other considerations. (Davis 1998; Kenchington 1993; Carr and Reed 1993; Fairweather and McNeill 1993; Baker *et al* 1996; Sladek Nowlis and Roberts 1997; Roberts 1998; Allison *et al* 1998; Kripke and Fujita 1999a; Fogarty 1999). Summaries of reserve design criteria for fisheries MPAs are outlined in the following sections.

6.2.2 Data needs for designing fisheries MPAs

In theory, design of fisheries MPAs requires spatial knowledge of the various life history stages of the 'target' species, an understanding of which areas and processes are critical to those life stages, and an understanding of the response of population parameters to fishing patterns over space and time. In practice, knowledge of all these factors is (and is likely to remain) limited to very few species, mainly commercially valuable fish and invertebrates, which attract the necessary research funding. For many marine species populations, including almost all of those in South Australia and other Australian States, there is incomplete knowledge of population sizes, distributions, reproductive dynamics, dispersal and migration rates and routes for each life stage, and the relation of those stages to critical habitat and oceanographic features. The long time frames required to gather such data, in order to 'optimise' MPA design for even a few commercially and socially significant species, makes the approach impractical. Furthermore, optimising MPA design to suit the dynamics of one species may be entirely inappropriate for protecting many other species that do not share the same life history characteristics. There are also potential conflicts between the design of MPAs based upon biogeographic and ecological representativeness, and designs to maximise the protection of particular fished species. Also, any one location chosen for a MPA will contain numerous species with entirely different life history phases and population dynamics, and designs that are optimal for one type will not be suitable for another. Ballantine (1999) considered that single-species MPAs are unlikely to be self-sustaining, unless they are very large. All of these considerations make a single-species design approach of questionable value.

Whether or not MPA design should be centred on a few, commercially valuable species is a socio-political decision, which will not be debated in this report. However, specific 'no-take' MPAs for fish species are obviously an important part of an overall MPA network in many countries (see Chapters 2 and 3), particularly MPAs for over-exploited stocks. Although the benefits of fisheries MPAs are widely known and demonstrated (including benefits of *ad hoc* fisheries MPAs that were designated before design criteria were even considered), the design of such MPAs has been debated for well over a decade.

An example of the debate over which data are required before fisheries MPAs can be effectively designed and implemented, is the scientific argument over the importance of spawning and larval 'sources' and 'sinks' (which are discussed in detail in a later section). Some workers consider that knowledge of larval dynamics and dispersal patterns is essential in MPA design (eg Fogarty, 1999). Others (eg Roberts 1998) consider such knowledge to be:

- an unachievable goal for all but a few commercially valuable species
- a scientific ideal that can stall or even undermine the urgent process of MPA designation, because proof is so hard to provide
- of limited practical use, because (a) any one area can be both a 'source' and a 'sink' depending upon which species you are considering, and (b) a source area can become a sink, and vice versa, as patterns and processes in oceanography, fishing, and habitat quality change over time.

A related debate is whether a 'sink' area for incoming larvae or recruits should be protected. Whether or not 'sink' areas should be included in a MPA design network depends upon the objectives of those MPAs. From a fisheries perspective, protection of sinks may not directly assist fish yields, because they will not be supplying recruits to the fishery (Williams and Russ 1995). However, if one of the goals of MPA establishment is to replenish depleted populations, then protecting sink areas using MPAs might be a suitable strategy, so that incoming larvae can have the opportunity to reach adulthood, irrespective of fisheries enhancement considerations. In some cases, areas that currently operate as 'sinks' may have potential as 'source' areas if they are protected to the extent that critical numbers and densities for reproductive success are restored.

In many cases, the use of MPAs to protect the adult stock of major commercial species (eg lobsters, reef fish) from over-exploitation, has successfully replenished population numbers and in some cases provided 'fisheries export' (see Chapters 3 and 4), without the need for knowledge of source and sink dynamics.

To date, there appears to be no consensus regarding fisheries MPA design. Indeed, there may not ever be consensus, given the difficulty of generalising rules for application to numerous fish species with different dynamics, subject to different fishing patterns, in different countries with a great variety of ecosystem types, as well as different physical and oceanographic patterns and processes. Nevertheless, a theoretical 'wish list' of data needs for design of fisheries MPAs is provided below, for information.

'Seascape' ecology

Some knowledge of the scales, patterns, structure and function of coastal marine ecosystems and processes that affect marine populations has long been suggested as important in MPA design. The amalgam of various marine disciplines into the 'umbrella' science of *seascape ecology* could describe this endeavour. Meta-population dynamics (see below) are affected by the physical and biotic factors that are studied in seascape ecology. Examples include studies of the effects of current patterns and other oceanographic features, sea surface temperatures, coastal topography, and habitat distribution and type, upon population variables such as distribution, migration, larval movement and settlement of marine populations, at various space and time scales.

Understanding the influence of oceanography and other physical factors can assist the placement of MPAs to protect the biota and habitats that are affected by those factors. An example of the way in which seascape ecology can assist the design of fisheries MPAs is the knowledge that local coastal physiography, topography and habitat type determine how and where abalone larvae will concentrate and settle close to the coast. Kelp beds may attenuate bottom currents and thus help prevent abalone larvae from being carried away from the parental reef (McShane *et al* 1988; Eckman *et al* 1989, cited by McShane 1992). Thus, McShane *et al* (1988, cited by McShane, 1995) proposed that abalone larvae released near high-relief reefs or macro-algal stands are confined to areas near the natal site, and will settle close to the area where they were spawned, compared with larvae released into open waters with no physiographic 'obstructions'. In the latter case, larvae may be dispersed over large distances, such as hundreds of metres, according to Shepherd *et al* (1992) or several kilometres (Tegner 1992 and 1993), depending upon the species and the local oceanography, coastal topography and physiographic patterns between the two. Shepherd *et al* (1992) also proposed that the sheer zones, eddies and stagnation zones generated by bays, inlets and small islands in the presence of currents can concentrate larvae into sheltered bays.

Observations of similar significance for MPA design have been made for some coastal fish species. For example, in areas where current strength dissipates close to the coast, recruitment of juvenile fish out of the plankton can be stronger in sheltered areas than in adjacent headlands. Jenkins *et al* (1993) showed that post-settlement King George whiting in southeastern Australia were uncommon in seagrass beds exposed to strong tidal currents, but were abundant in beds where currents dissipated.

It is clear that such information has implications for single species MPA design, particularly the delineation of 'source and 'sink' locations (see below).

Meta-population dynamics

In the marine world, a *meta-population* can be defined as either as a genetically distinct or reproductively discrete population of members of one species, or a collection of interacting local populations of the same species which are reproductively connected (Shepherd and Brown, 1993). Meta-population dynamics originated in terrestrial conservation science, and is concerned with the persistence of populations over space and time, according to both random and predictable environmental variation and impacts; variation in habitat (patchiness, fragmentation, linkages, etc), and the influence of these factors on the size, demography and reproductive dynamics of the meta-populations. The concept proposes that ensembles of interacting populations with a finite lifetime (Hanski and Gilpin 1991) are constantly subject to external and internal controls, which determine population stability, extinction, and re-establishment ('turnover').

Meta-population dynamics investigates the effects upon population size and persistence of such external population-controlling factors as:

- *Environmental stochasticity*: the effects of random or at least unpredictable physical and ecological factors (Shaffer, 1987).
- *Natural and human-induced 'catastrophes':* in the coastal marine environment, this could include algal blooms, viral or bacterial-related disease outbreaks; temperature-induced fish or invertebrate kills, all of which can cause large decreases in population numbers over short time frames. Overfishing could also be termed catastrophic for various populations. The effects of El Nino/ENSO events could also constitute a catastrophe for some populations. Mangel and Tier (1994) considered catastrophes to be major determinants of population persistence.
- *Habitat variation over space (including patchiness, fragmentation* and *linkages*).

The 'external' factors listed above interact with (and affect) 'internal', population-specific variables such as the following:

- *Demographic variation:* including differences between populations in age structure, and in birth, death and reproduction rates.
- *Reproductive dynamics:* rates of reproduction, dispersal, immigration and emigration determine 'source' and 'sink' locations for larvae and/or adults, and can vary considerably between populations comprising a meta-population.
- *Genetic variation*: which permits populations to maintain themselves and to 'adapt' in the face of constant environmental change at various spatial and temporal scales.

An example of the link between habitat, physical variables (depth, in this case) and meta-population dynamics, is the work of Rogers-Bennett *et al* (1995), who showed that sea urchin morphology, densities and recruitment rates varied according to habitat type (eg presence of 'rock bowls'), food availability and depth (deep versus shallow). This has implications for the size and location of MPAs for that species: protecting the highly fecund shallow water meta-population (the 'source') could assist 'replenishment' of the deeper water, heavily harvested meta-population.

Population parameters such as growth, natural mortality, migration rate, age at maturity, and fecundity can also influence whether or not a MPA will achieve its various aims. The effect of fishing on these parameters over space and time, and implications for MPA design, has been the subject of an increasing number of modelling studies. Two of the earliest (and most cited) models, from Polacheck (1990) and DeMartini (1993), have shown that fast growing species will yield higher biomass per unit time and area in a MPA than will slower growing fish. However, this generalisation is complicated by migration rates: the benefit of increasing spawning stock biomass in a reserve will not be realised if dispersal rate out of the reserve is high (DeMartini, 1993). Regarding age at maturity and relative fecundity, a MPA cannot be expected to increase spawning stock biomass if much of the population outside the reserve is intensively fished prior to age at sexual maturity (DeMartini, 1993), and thus relatively few animals reach spawning size or maximum reproductive potential (which, for many marine species populations, is associated with older age classes).

Minimum viable population size

Estimates of minimum viable population (MVP) and meta-population sizes (Gilpin 1987; Shaffer 1987; Shafer 1990) was a popular topic in terrestrial theoretical ecology during the past decade, and such work has application in the design of fisheries MPAs. Data used in such work are closely related to meta-population dynamics studies, and include population parameters (such as reproductive success and mortality rates); estimates of habitat fragmentation and linkages; and the response of population dynamics to environmental and human-induced stresses over space and time. Generally, (i) larger populations have larger ranges, which increases the chance of population persistence, despite environmental or human-induced stresses (Shafer, 1990); and (ii) reducing population size increases the chance of population decline, due to loss of reproductive potential and reduced genetic fitness. MVP sizes are difficult to estimate for marine species populations. Several obvious reasons for this include poor knowledge of total population size and distribution; lack of understanding of population dynamics, habitat linkages, and the impact of fishing and other influences on populations over space and time; the fact that many marine species populations are far less vulnerable to extinction compared with many terrestrial species; and sampling difficulties.

However, given the basic premise that a ‘larger’ population is more likely to persist in the long term than a ‘smaller’ one (unless a catastrophic event significantly reduces the population), fisheries MPA can be more successful if their design is considerate of the need to:

- maintain population size where possible
- maintain the links between meta-populations, in terms of demography, migration, dispersal paths and other reproductive linkages.

Population demographics (growth, mortality, age structure) can significantly affect minimum viable population sizes (Lande, 1988). There is increasing evidence for some marine species populations that alterations to the age structure has affected spawning aggregation numbers and densities (see references in Chapters 2, 3 and 4), which ultimately affects viable population size. This is detrimental to the long-term persistence of populations of long-lived species, particularly if the most fecund age classes in the population are heavily fished, as has occurred in a number of snapper, grouper and grunt fisheries throughout the world.

Maintaining population age structure is thus another important factor in fisheries MPA design. This can be achieved by protecting spawning adults, particularly ‘sedentary old fish’ which are often the ‘megaspawners’ in the population; by designating MPAs in which fish population numbers and biomass can naturally be replenished over time.

Source and sink dynamics

Many marine species populations are perpetuated by movement of larvae or adults from what Pulliam (1988) has termed 'source' areas (from where the propagules or animals originate), to 'sink' areas (where those propagules or animals settle). Pulliam (1988) has defined sink areas as reproductively-insufficient areas with high mortality-to-recruitment ratios, whose populations persist due to immigration or replenishment with either larvae or older animals from more productive 'source' areas. According to meta-population dynamics theory, small 'sink' populations which are subject to 'environmental stochasticity' (such as processes causing consistently poor recruitment) and/or 'catastrophe', suffer a high probability of decline (Shaffer, 1987; Stacey and Taper, 1992). However, immigration (eg by incoming larvae from another meta-population) can 'rescue' the small population from decline, which highlights the importance of maintaining larval source populations. Small, reproductively 'isolated' meta-populations are particularly vulnerable to decline: they may have no reproductive linkages with other meta-populations, and use their own 'internal dynamics' to maintain themselves (Hanski and Gilpin, 1991), but cannot persist easily when faced with factors which reduce population size or reproduction potential.

Source and sinks may be habitat-specific (eg seagrass beds in which larvae settle), or be controlled by physical factors such as the location of upwellings, eddies, sheer zones, offshore jets, and other oceanographic features. Source areas may contain abundant resources, or particular oceanographic features that promote reproduction (eg via spawning aggregation). For coastal species, sink areas may range from (i) tens to hundreds of kilometres from the source of the larvae (in the case of migrating temperate fish, tropical reef fish, and many macro-crustacea) or (ii) the same reefs from which the larvae were spawned, or areas in the vicinity of tens to hundreds of metres (eg abalone, and many mollusk and echinoderm species).

The importance of sources and sinks of recruits in MPA design has been considered during the past decade (Carr and Reed 1993; Tegner 1993; Quinn *et al* 1993; Roberts 1995; Williams and Russ 1995; Baker *et al* 1996), particularly for meta-populations of benthic marine species that:

- have spatially separated spawning groups
- require critical densities to be maintained for fertilisation success
- display different population dynamics within the one species (eg see Shepherd and Baker 1998).

Carr and Reid (1993) displayed common models of larval replenishment, according to (i) the location and number of larval sources (eg spawning adult populations); (ii) the critical distance between one or more larval sources and the corresponding larval sinks; (iii) 'self-recruitment' potential within sinks; and (iv) the presence or absence of a major larval 'pool' which replenishes each population contributing to that pool. Protection of larval source areas is considered to maximise fisheries benefits, especially within a network that is designed to protect sites that act as 'source' areas for the maximum number of other sites (Williams and Russ 1995).

Knowledge of (i) reproductive dynamics (such as size, density and spacing of populations, and their reproductive connections), and (ii) the oceanographic variables and patterns affecting larval timing, movement and duration, can assist in determining important source and sink areas. These factors are more easily studied in some mollusk and echinoderm populations, particularly of those species with short-lived, pelagic, lecithotrophic (yolk-feeding) larvae. In such populations, adults and juveniles are much more closely coupled in space and time than those with long-lived planktonic larvae. Abalone, for example, have short-lived, pelagic larvae which are often (i) retained near their natal reef or bay, or (ii) subject to limited dispersal, due to local area hydrodynamics and topography (McShane *et al*, 1988; Shepherd *et al* 1992). Thus MPAs, which are designed to protect local adult spawning populations associated with a particular bay or coastal reef, are quite likely to also indirectly protect larval recruitment into the same area, or into reproductively-connected adjacent areas. Results from abalone studies have shown that in addition to reduced spawning stock sizes, critically low *density* of spawners (which affects aggregation potential) has been implicated as a major factor reducing the long-term recruitment success in two greenlip abalone populations (Shepherd and Brown, 1993; Shepherd and Partington 1995). Work has also focussed upon the reproductive connections between abalone sources and sinks, based upon larval dispersal directions and distances. Wells and Keesing (1990) showed that one depleted abalone population on a Western Australian reef was repopulated by larvae from a neighbouring reef several kilometres away. McShane *et al*'s (1988) modelling studies also showed probable reproductive connectivity between separate reef meta-populations of abalone, depending upon the position of reefs and the strength of along-shore currents. Tuck and Possingham (1994) also described theoretical relative exporter (source) populations and relative importer (sink) populations of abalone, based upon large and small larval production quantities within a meta-population, and recommend that populations which produce high numbers of larvae should be 'conservatively harvested'.

For most fished species with planktonic larvae, source and sink dynamics are not known. Many tropical reef fish, for example, disperse recruits to reefs hundreds of kilometres away from the source spawning areas (Williams and Russ 1995). For some migratory coastal fish species (eg see Lenanton *et al* 1991, for Australian salmon; and Kailola 1993, for tommy ruff), larvae can move long distances between the 'source' (southern Western Australia, in this case) and the sink(s) (the estuaries of southern WA, South Australia and Victoria, in the case of tommy ruff). Annual abundance of juveniles in one region depends upon the variable annual strength of major currents and their timing of peak flow. Whether or not the 'sink' areas for those incoming larvae should be protected from over-exploitation is a socio-political decision. In any case, knowledge of larval source, transport routes and sink areas, where known, can assist management.

If the relative abundance of recruits and their areas of settlement are known, MPAs can be designated to protect the outcome of this process, either as permanent reserves (eg in 'nursery' areas with high concentrations of juvenile fish) or temporary MPAs (eg seasonal closures in years of particularly strong recruitment which may be later transferred to successful fishing years).

A number of MPA studies have demonstrated (i) protection of spawning 'source' population sizes and densities, and recruitment processes (Chapter 3); (ii) successful export of larvae to population sinks (Ebert and Russell 1988, cited by Fairweather and McNeill 1992); and (iii) supply of recruits to replenish depleted areas (see Chapter 3). Often recommended in MPA design is the theoretically simplistic but practically difficult task of designating networks of MPAs that are spaced closer than the critical dispersal distance of the populations in question.

Compared with source and sink areas, the larval movement process can obviously rarely be protected by MPAs. However, engineering works, such as inlet stabilisation, breakwater construction, dredging of navigational channels, etc, can disrupt the passage of incoming larvae near estuaries and shorelines, and ultimately affect recruitment. Therefore, any new proposals of this sort should be evaluated in light of potential effects upon local and regional fisheries (Epifanio 1988; McConaugha 1988; Fairweather 1991). One example of engineering works affecting the larval stage comes from Canada, where closure of a strait disrupted migration of larval lobsters, and is partly attributed to the decline in the lobster stock in that area (Fairweather 1991).

Scales, distribution and site dependence of life history stages

In theory, knowing the population dynamics of the 'target' species can significantly aid the design of fisheries MPAs. Some theorists consider that MPAs can only protect key life stages if there is knowledge of (i) the scale and distribution of those stages; (ii) their habitat connections; and (iii) how those life stages are affected over space and time by various natural and human-induced impacts. For example, some species may be long-lived, migratory, and have spatially-separated age groups, which are fished in different localities and at various times by differentially-selective gear types, throughout the species life history.

Regarding the scale of life history processes, Kenchington (1990, Figure 3.1, p. 34) addressed conservation potential for critical life stages according to the spatial extent of (i) the territorial or site dependent phase, such as juvenile fish dependence upon a particular estuarine habitat for food and shelter; and (ii) the distribution or migratory phase. For example, the spawning area of some coastal fish species may be hundreds of kilometres from the residential or schooling territories where those fish usually live. Kenchington (1990) has identified four types of life history phase relationships, based upon the mixes of site dependence, territoriality, and/or migratory behaviour displayed by various species life stages. Fisheries MPAs can be useful for three of these life history types.

Depending upon the dynamics of the particular species, long-term persistence may require the use of MPAs to (i) protect estuarine habitat for juveniles (see examples in Chapters 3 and 4); (ii) protect the spawning potential of older, highly fecund adults (eg snapper, grouper, grunts and the like); or (iii) protect large groups of animals from being opportunistically overfished or caught as bycatch during major migrations or spawning aggregations.

In South Australia, for example, there is potential for MPAs to protect the populations of (and enhance the fisheries for) species displaying:

- limited adult territory and a planktonic larval phase (eg blue crabs, lobsters)
- a fixed, habitat-dependent phase, with planktonic larvae or a large migratory range during some part of the life cycle (eg snapper, and some other coastal fish species such as whiting)
- dispersed adult and juvenile phases, with seasonal spawning aggregation of adults (eg cuttlefish)
- a site-specific adult phase with a limited larval and adult range (eg abalone).

Spawning source areas for major coastal fisheries species in South Australia can be concentrated or dispersed, and can vary from local areas close to shore (eg cuttlefish), to offshore sites out of the gulfs (eg whiting), to waters thousands of kilometres from the site of the fishery (eg tommy ruff, Australian salmon). Designing fisheries MPAs to protect aggregated spawning stock is obviously easier than efforts to protect more dispersed stock, because aggregations are now easy to locate and map due to the increased use of GPS, echo sounders, and benthic plotting equipment. Similarly, larval 'sinks', such as coastal nursery areas in estuaries and bays, are also easy to locate.

It is obviously easier to protect coastal shallows (with which juveniles of many fisheries species are associated, and which are spawning grounds for other species) and mid-water areas (eg reefs for 'site-attached' adult fish), compared with offshore areas (which are spawning areas for some coastal fish species)[8]. For species with limited larval dispersal, habitat for juveniles is often close (or identical) to habitat for adults. Fisheries MPAs for such species could thus help to protect the numbers and densities of adults, and the quality of their habitat.

Spatial and temporal factors affecting species population dynamics are considered relevant to design of fisheries MPAs. For example, the distribution and age structure of migrating and/or schooling species, together with knowledge of the duration and location of migration, can determine how useful a MPA would be. There may be little point in designating an area as a MPA if key populations are only present for part of their life cycle (or part of the season), and are subject to major impacts at other times, or in other areas away from the MPA. This aspect of MPA design is important on local, regional and even national scales. For example, the area and boundaries of politically-defined States may rarely correspond to the spatial boundaries of coastal species distributions. If a stock has a broad coastal distribution ranging across politically-defined States, a MPA in one State may eventually fail if the stock periodically migrates interstate and is not appropriately conserved in the adjoining State.

Habitat patchiness, fragmentation and linkages

Protecting a variety of critical habitat types and sizes to prevent species and population loss has been widely accepted in terrestrial conservation for several decades (Shafer 1990; Hanski 1991). Although the importance of protecting critical marine habitat was stressed as early as the 1970s (Ray 1976, 1978), marine habitat received little attention on a global scale until more recently, with regard to MPA design. The 1992 Australian Conference for Fish Biology stressed the importance of protecting critical habitat for fish stock protection and fisheries management, and MPAs were recommended as a major method of achieving this aim (Hancock 1994). Protecting critical habitat using MPAs was also the subject of a recent international conference in North America (Mote Symposium 1998).

There are numerous examples in the marine literature, demonstrating the physical, chemical and ecological importance of various marine habitats/ecosystem types (see Appendix 2). For example, juveniles of many coastal fish and crustacean species (such as King George and yellowfin whiting) depend upon particular habitat types (such as seagrass-dominated estuaries) during their critical growth and development phase prior to dispersal.

Ecosystems/habitats within the same general category (eg ‘seagrass’) each have specific ecological values, which is relevant to the concept of ‘representativeness’ in MPA design. For example, different species rely upon specific seagrass habitats within and between different estuaries. In other words, seagrass estuaries are not functionally all the same at a specific level (Bell and Pollard 1989; Ferrell *et al* 1993). Furthermore, the type and location of seagrass beds that assist recruitment of commercially important fish species may differ from those which act to maximise the *diversity* of many different fish and invertebrate species (Jenkins *et al* 1992).

Faunal diversity can be higher in the presence of marine plants compared with unvegetated habitats, as shown by Edgar *et al* (1993) in Victoria, who collected significantly more species from seagrass in Westernport Bay than from bare sand, and calculated more than twice the total production of small fishes from seagrass compared with unvegetated substrate. There is

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[8] Note that offshore spawning areas could ostensibly be protected using seasonal or area closures against offshore fishing, although surveillance could be difficult.

also trophic connectivity between seagrass and bare areas. For example, seagrass detritus reaching bare sand supports great abundance of prey for fish (Jenkins *et al* 1993).

It is now commonly known that loss of seagrass beds can adversely affect fisheries, as shown by Jenkins *et al* (1993) and Edgar *et al* (1993), who associated the declining catches of whiting and leatherjacket fish in Victoria with seagrass loss. Kelp beds are also critical habitat for many species, and their removal can affect fish abundance, including that of ocean-going pelagic species, because juveniles of many fish species use seaweed to shelter and feed in (Lehtinen *et al* 1988, cited by Fairweather 1991). Jenkins *et al* (1996) showed that species richness and abundance of fish, including that of commercial species, were much higher in macroalgal reef habitat than on bare sand. Macroalgal reef beds have been described as 'keystone' ecosystem types, which provide conditions for continued recruitment of fish and invertebrates (Fairweather 1991).

Within the range of a species that has habitat linkages, population sizes will differ between habitats, due to variation in size, structure and resource content of those habitats. Some habitats contain more food, substrate for settlement, or area for shelter, compared with others. This *patchiness* or heterogeneity of habitat has been considered important in population regulation and maintenance, according to meta-population dynamics theory. The various factors that cause population decline will affect some patches and not others, permitting some populations to act as sources or 'recolonisers' (Hanski and Gilpin 1991; Doak *et al* 1992). Furthermore, larvae are also patchily distributed, so the size and proximity of suitable settlement habitat (eg estuarine seagrass) can 'make or break' an age class of larvae.

The spatial extent of resource use in habitat patches is also considered important in MPA design. Populations of some species may be well provided for by the resources available in limited and well-defined habitats, whilst others may utilise a variety of habitats throughout their life. Some fish species are effectively 'habitat-bound', and will only reproduce on their 'home' reefs. Other species will not move away from the shelter of dense seagrass and macro-algal stands, perhaps due to the higher likelihood of being preyed upon if they traverse bare sandy areas in which they are not camouflaged.

Populations that migrate over local, regional or continental scales have more complex relationships with habitat: the persistence of some populations may depend upon a diversity of habitats being available in a region (Pulliam 1988), rather than depending upon the resources in any one critical habitat. In marine systems, many migrating species are independent of habitat (eg feeding and reproduction occur in the water column), but others rely upon specific habitat types (which may be separated by hundreds or thousands of kilometres) at certain life stages. For example, some fish species (eg mulloway: Kailola 1993) may migrate between estuaries that are dozens or hundreds of kilometres apart.

Decreases in both total area and density of seagrass patches or reef cover (or other marine habitat type) could thus indirectly affect population abundance or dynamics in various ways. Species populations that have 'habitat linkages' can be protected using MPAs, especially if the critical habitats that provide resources for those populations are well separated, and thus alternative habitats are too distant for the population to reach.

Bell *et al* (1988) considered that settlement success of larvae (and consequent recruitment to the fishery) will be assisted by conserving as many patches of seagrass in as many different locations as possible, within any seagrass estuary. A challenge for single species MPA design would be to devise a system of MPAs to ensure that critical estuarine, seagrass and reef habitat is maintained at sufficient size and spacing for the protection of larval and juvenile fish over a broad region. (Obviously, concomitant ecosystem management must also occur, to ensure that the factors causing seagrass decline, such as nutrient pollution and siltation, are controlled.)

Fragmentation of habitats, either by natural processes or human-induced impacts, can increase patchiness. This is also regarded as a major determinant of population persistence or decline, according to the size and spatial arrangement of fragmented patches, and considering that the chance of death is greater for population members that must move greater distance (and for a longer time) to reach a habitat fragment, compared with an unfragmented area (Doak *et al* 1992).

In the marine environment, temporary fragmentation can result from natural processes (storms etc), but of far greater concern is the fragmentation that can result from the physical and chemical degradation of habitats which is continuously occurring in many regions, particularly coastal waters adjacent to areas of human settlement. Examples include nutrient-induced seagrass decline, physical damage to reef ecosystems, and physical damage and chemical pollution of bays and estuaries, etc. Such 'fragmentation' of these habitats could:

- diminish the habitat 'carrying capacity', thus necessitating the move by some animals to other habitats where resources are in greater relative abundance
- possibly alter some population dynamics parameters, such as growth and reproductive success, if migrating members of the population cannot locate habitat fragments that are sufficient in size and resources
- increase the risk of death due to natural predation or to non-target fishing mortality (eg as bycatch; undersized catch), for animals which need to move long distances to find sufficient habitat for feeding, shelter, reproduction, etc.

Populations which persist in poor quality habitats may depend upon neighbouring habitats for certain resources (Pulliam 1988). Examples of the relation of habitat fragmentation, patchiness and linkages to MPA design would be the need to protect the habitat of a spatially 'isolated' meta-population, because the chance of larval dispersal or recolonisation may be lower than for populations in geographically closer, more 'connected' habitat patches.

In theory, MPA design will be assisted by determining which habitats (including 'fragments') are likely to be important patches for various coastal species. Practically, this could only be achieved for a very few species, particularly sedentary species with strong habitat linkages, and a few commercial fish species whose life history phases and habitat linkages are well studied.

Recent spatial models that incorporate critical habitat type and distribution in the design of MPAs for commercial fish include the work of Lindholm *et al* (1997). Model results suggested protection of critical habitat for newly settled juvenile fish (based upon correlations between fish settlement strength, and habitat complexity), and considered that migration rates away from critical habitat are also an important factor in MPA design. Similar recent work on MPA design for tropical marine habitat conservation (Appeldoorn and Recksiek 1998 (unpublished)) has emphasised the relationships of species diversity to habitat diversity and area, correlation of habitats in space, and known movement patterns of fishes relative to habitat use.

6.2.3 Design criteria for fisheries MPAs

Effective sizes, numbers, shapes and locations of fisheries MPAs obviously depend upon the purposes of the MPA. The design criteria for single-species fisheries MPAs differ greatly, according to a number of factors that have been discussed in the previous section. These include:

- the life history phases, population dynamics and demography of the species in question
- oceanographic and other physical factors affecting the life stages
- larval movement patterns
- habitat linkages (including the size, shape and connectivity of critical habitats to which any of the life phases are related)
- fishing patterns; and the population response to fishing.

One of many examples is the recent modelling work of Stockhausen *et al* (in review) in which MPA configurations were simulated to enhance or maintain fisheries catches, according to criteria such as hydrodynamic patterns, fishing intensity and larval transport, which are used to simulate appropriate sizes and locations of MPAs to satisfy these fisheries objectives.

Some design criteria for fisheries MPAs are obvious. For example, (i) the location of spawning aggregations on reefs or in relation to specific oceanographic features; and (ii) estuarine 'nurseries' that function in the feeding and rearing of larval and/or juvenile stages, and 'export' the grown fish to adjacent fisheries, are clearly important candidates for MPAs. Such areas may be easy to locate and map, and thus MPA designs to include such features are theoretically and practically simple, and will often receive high social and political support. However, most design criteria for fisheries MPAs rely upon a variety of different oceanographic, geomorphological, ecological, biological and population dynamics data (see Section 6.2.2). It is difficult to 'optimise' MPA design to suit a variety of species, given the differences in life history patterns and processes, habitat linkages, and responses to fishing and other impacts that affect population dynamics.

Although design criteria (size, shape, numbers and locations, amongst others) for fisheries MPAs are related, they have been separated below to facilitate their description.

Size

It was recognised more than 20 years ago that the optimal sizes of fisheries MPAs can usually only be species-specific, which satisfies a limited number of MPA objectives (Rooney *et al* 1978). Clearly, given the variety of life histories, population dynamics and distribution patterns of fish, there can be no set guide to MPA size that satisfies the objectives of conserving and managing many different fisheries species in an area of any given size.

Davis (1989) implied the difficulty of determining optimum size for fisheries MPAs. Davis suggested that MPAs must be large enough to be productive (and assure that target species populations are perpetuated), yet small enough to optimise the boundary to volume ratio, and assure maximum export of propagules and juveniles, and even (in some cases) adult fish.

Modelling work has provided some useful guidelines for MPA sizes to satisfy single species conservation and management objectives. For example, Polacheck (1990) and DeMartini (1993) emphasised mobility of stocks in MPA design, stating that the higher the movement rate of fish out of reserve areas, the larger the reserves need to be to (i) maintain increases in spawning stock biomass relative to the surrounding fished area, and (ii) to maintain the differentials in both fish density and fishing effort inside, compared with outside, the MPA. Carr and Reid (1997) also suggested that 'larger' MPAs are more likely to be self-replenishing, considering that many marine species have highly dispersive life stages. Recent examples of suggested MPA sizes for single species, based upon population modelling, include the work of Sumaila (1998), Mangel (1998) and Guennette and Pitcher (1999). Mangel (1998) has modelled MPA size according to the proportion of stock protection required to sustain a fishery, and explored the 'trade-offs' according to MPA size, and harvest regulation outside the MPA. French researchers have also modelled the effects of MPA size on fisheries yields, using the SHADYS simulator (Maury and Gascuel 1999). The logical conclusions from that work are that 'larger' protected areas are suitable for 'diffusive' or migratory species, and can act as 'source' areas to supply the fishery, whereas similar sized areas for 'resident' species can adversely affect the yields in adjacent fisheries, and therefore smaller areas are preferable for such site-attached species. Sumaila (1998) reached a similar conclusion, using a dynamic bioeconomic model, namely, that large MPAs are bioeconomically beneficial when net 'transfer' rates of fish are high: they assist the fishery, they can 'mitigate against biological losses', and they protect the stock against the 'shock' of severe recruitment failure.

Compared with suggested sizes from theoretical modelling studies, there is a global paucity of empirical evidence pertaining to the appropriate size of MPAs for specific purposes (such as fish stock protection and fisheries replenishment). However, there are some empirical examples of the ways in which size determines the success of a MPA for satisfying single-species conservation / management objectives, such as the following:

- Shepherd (1991) and Shepherd and Brown (1993) showed that recruitment dynamics, spawning stock size and location, and broad patterns of larval location need to be considered, so that a designated MPA is large enough to protect the reproductive dynamics and spawning stock of a regional meta-population.
- High 'site fidelity' and limited daily movements of particular fish species mean that quite small reserves can be used to effectively protect populations of mature adults. This was demonstrated by Holland *et al* (1993) for white goatfish (*Mulloides flavolineatus*) in a Hawaiian reserve, and by several of the MPAs for site-attached reef fish species, discussed in Chapter 3 of this report.
- Recent work in Tasmania (Edgar and Barrett 1999) has shown that, compared with fished reference sites, improvements in biomass, abundance and densities for several commercial fish and invertebrate species, are positively correlated with MPA size. Few improvements were noted in the smallest MPAs (one km coastline), more in the two- km MPAs, and the largest number of improvements in the biggest MPA (seven km).
- Work in a temporary MPA on the Great Barrier Reef has recently shown that protected reefs of several square kilometres permit population densities of the heavily harvested black teatfish (a sea cucumber) to recover, and that smaller MPAs were not effective in achieving population replenishment (Australian Institute of Marine Science, unpublished media release 1999).

Some examples of the recommended size for coral reef MPAs to protect the spawning dynamics of coral, other invertebrates and reef fish varies by an order of magnitude (eg from 300 ha to 3470 ha, according to Talbot 1993, cited by Brunckhorst 1994). There are other examples, too numerous to cite here, of the variety of suggested MPA size for specific purposes, and most of those examples are species-specific.

MPA size has been debated in terms of protecting spawning aggregations, larval sources and sinks, site-attached reef fish and invertebrates, and maintaining population dynamics of commercially and recreationally significant species populations (see Chapter 3 and Section 6.2.2). Even small MPAs of less than one square kilometre can be effective for protecting strongly site-attached species and spawning aggregations, depending upon the spatial limits of the population and the density / spatial concentration of the spawning aggregation. Kripke and Fujita (1998) suggested specific MPA sizes for the protection of particular commercial fish species in North America, (such as rockfish, thornyheads, and Dover sole) according to MPA goals, such as protection of spawning biomass and age structure.

In New Zealand, the famous Cape Rodney-Okakari Point (Leigh) Reserve is only five square kilometres, yet it has been very successful in replenishing fished populations, providing fishery export, and restoring habitat (Ballantine 1999). This MPA was established long before workers considered what an 'optimum' MPA size should be to manage particular fished stocks. In Australia, many small MPAs of less than one square km have been established (McNeill 1994), and some of these have satisfied the limited number of objectives under which they were declared.

Size of MPAs for the protection of site-attached species populations is easier to determine than that for more wide-ranging species. Study of the short and long-term movement patterns of site-attached reef fish in Tasmania, has shown that temperate reef fish in the Labridae and Monacanthidae families do not move far from their 'home reefs' (Barrett 1995). The existence of natural 'boundaries' of open sand between reefs, into which these reef-attached species no not move, means that the size and shape of MPAs that are designed for protecting such site-attached species should accord with the positions of the reefs. In such cases, even small MPAs can be effective in achieving specific aims, such as protecting a portion of the stock of heavily fished reef species. Roberts and Hawkins (1997) showed that a small (2.6 ha) MPA in St Lucia (Caribbean) protected heavily fished snappers and grunts – large individuals resided in the MPA, and the species were found nowhere else along the heavily fished coast. However, small MPAs are obviously vulnerable to 'edge effects' such as fishing at the MPA boundary, which can reduce their effectiveness in practice.

In South Africa, both small and large MPAs have effectively protected fish stocks, depending upon the objectives of those MPAs. Fifty kilometres has been considered an appropriate size for the protection of fish species targeted by shore anglers (Atwood and Bennett 1994). Smaller MPAs may be effective for 'site-attached' species (eg commercially harvested mollusks) with limited larval dispersal and adult movement. In general, South African authorities have supported large MPAs, partly for practical reasons, including the ineffectiveness of protecting small MPAs without buffer zones from impacts, and the difficulty of policing small MPAs in remote areas (Atwood *et al* 1997).

Some workers have devised 'rules of thumb' for MPA size, according to the population dynamics of major marine groups. Barrett (pers. comm., cited by Major 1998), for example, has suggested that abalone or lobster may only require MPAs as small as a few hundred metres for most individuals to be protected, and for larval export to the surrounding fisheries. In the case of lobster, benefits to both local and regional fisheries can occur, due to the existence of long-lived, widely dispersed larvae. Barrett suggested that reef fish require 10 times the scale of their home range to be protected. In Tasmanian ecosystems, this would equate to at least five km of coastline.

The Relation between Size and Number

Size and number are intimately related in MPA design. Whether populations are better protected by a single large or several small reserves ('SLOSS'—Simberloff 1988; Shafer 1990) has been debated for well over a decade in terrestrial conservation science. Much of the terrestrial discussion of the optimal size and number of reserves to prevent population extinction is not particularly relevant to marine systems, due to the greater connectivity between marine populations, habitats and processes than occurs on land, and the lower levels of human-induced fragmentation. However, a few general principles are transferable, and these are discussed below. Factors relevant to the SLOSS debate in marine ecosystems include meta-population dynamics, habitat quality and linkages, fishing patterns over space and time, and degree of threat (from both over-fishing and other impacts). Depending upon the species and the location, any or all of these factors may be important determinants of MPA size.

The following paragraphs discuss the theoretical and practical advantages and disadvantages of (i) single large MPAs, and (ii) networks of smaller 'no-take' MPAs. In reality, both 'single large' MPAs (eg marine parks, biosphere reserves) and 'several small' (networks of 'no-take' MPAs for fisheries, recreation, research, education, cultural purposes etc) usually make up the system of MPAs in many countries, including Australia and all its States. Both large and small MPAs serve a variety of valuable purposes (see Chapters 2, 3 and 4 of this report), and 'real-world' decisions about MPA design and placement are rarely based on the choice between one large or several small. The practical significance of the SLOSS debate could thus be considered relevant mainly for MPA designs that serve a specific purpose (such as conservation of threatened species, or protection and enhancement of fisheries populations), rather than multiple purposes, which an entire 'representative system' seeks to satisfy.

Single large MPAs

'Large' is a relative term, and if MPAs are large enough for activities and uses to be zoned, then the fisheries protection and management function may either be assisted or hindered, depending upon the number and size of 'no-take' zones within the large MPA.

The zoning of uses and activities in large MPAs has obvious social and economic benefits that may not be available in some smaller MPAs, as discussed in earlier chapters of this report. Larger MPAs also permit protection of biogeographically and ecologically significant 'core' areas through use of buffer zones, whereas sufficient area for creating buffers may not be available in smaller MPAs.

For species with low migration rates, and high dependence on specific habitats, modelling results indicate that large MPAs more effectively increase the spawning stock biomass of fish in the reserve (de Martini 1993). Larger MPAs are also more likely to supply ('export') stock to replenish depleted areas, either through natural emigration, or deliberate reintroduction to the depleted area. Large MPAs might remain more genetically diverse than small reserves because they often contain (i) larger numbers and variety of species, and (ii) greater numbers of reproducing individuals of certain populations, if habitat quality is maintained. These features can help to maintain reproductive potential and prevent 'critically' low numbers and densities of the spawning adult animals that maintain genetic variation. This is particularly important for heavily fished species, which have a high risk of reduced genetic diversity.

Large protected areas usually have a diversity of habitats, and, according to terrestrial conservation theory, they can also help to prevent the occurrence of 'genetic bottlenecks', which may occur in small (and remnant) habitat patches (Saunders *et al* 1991). Therefore, in larger MPAs, there is a greater chance of 'minimum viable populations' (MVP) being protected (Shaffer 1987; Shafer 1990), through the provision of essential resources that assist reproduction and survival. Larger MPAs are advantageous for species that require large habitat areas to carry out their life processes, or for those requiring a number of different habitat types throughout their life. For example, a large coastal marine MPA might contain protected estuarine area for juveniles as well as adjacent reef area for adults of the same species.

Despite the advantages discussed above, it might not be practical or feasible in some areas to designate large MPAs, representing several critical habitat types and protecting particular coastal processes. There are several reasons for this, including the fact that in some regions habitat damage and fragmentation though human-induced impacts has reduced the area that can practically be designated as MPA. There are also socio-economic limitations to MPA size, including:

- pre-existing (and sometimes competing) multiple-uses, which limit the available area for protection as MPAs

- public concern that the high economic value of particular areas (eg for fisheries) may be compromised if those areas are declared as MPAs
- restrictions to available MPA area, based upon pre-existing hectarage of MPAs in a State or country (even if those pre-existing MPAs were designated in an *ad hoc* way, and serve limited uses)
- negative public perceptions that large areas of the coastal marine environment will be 'locked up' if MPAs are designated.

Even the proponents of MPAs express concern about the design of large MPAs, and much controversy can be caused regarding the size and spacing of each zone. For example, the proposed multiple-use marine park at Port Phillip Bay (Environment Conservation Council 1998) was criticised by the Victorian National Parks Association (Major 1998) as providing for a 'business and usual' approach, without enough area designated as 'no-take', compared with the provision for recreational and commercial activity.

There are other concerns with the function and management of single large MPAs. Regarding the protection of critical habitat types such as estuaries for juvenile fish and invertebrates, single large MPAs may be less appropriate because (i) different estuaries within any one State are physically and ecologically dissimilar enough to support different fish assemblages (Bell and Pollard 1989; Ferrell *et al* 1993); and (ii) recruitment levels of any one species to different estuaries within or between bioregions can vary over space and time. Therefore, designating 'one large estuary', for example, as a MPA, is unlikely to be very useful unless it attracts an inordinate amount of the total incoming coastal larval assemblage.

Single large MPAs may also be inappropriate for protecting several meta-populations that are spatially separated, and exhibit variable population dynamics. Further, in large MPAs it may be difficult to prove that increased population and ecological benefits accrued due to the MPA, or due to other factors, because there are no 'replicate' or comparable reserves to corroborate the evidence. Lastly, Bryceson (1981) was probably the first marine worker to state in print the obvious fact that designating single large MPAs is impractical in areas that are subject to catastrophic events or high levels of human-induced disturbance.

Networks of Small MPAs

Networks of small MPAs can assist fisheries stock protection and management, particularly if species have spatially separated life history phases (for example, juveniles in an estuarine habitat, and adults on a rocky reef). In theory, small MPA networks are also useful for protecting and managing species that comprise different meta-populations, each with variable population dynamics. Designating a number of small fisheries MPAs can help to protect meta-populations against the effects of natural or human-induced impact. For example, some marine fisheries species have separate meta-populations, which have somewhat 'independent' dynamics. Even if the persistence of one meta-population suffers due to disease, over-fishing, or habitat destruction, the entire stock will not be jeopardised if MPAs are placed to protect spatially separated meta-populations, because those meta-populations are unlikely to all suffer the same 'disaster' at the same time. Therefore, genetic variability, migration potential, and/or spawning potential in the populations can be maintained (Quinn *et al* 1993; Baker *et al* 1996; Shepherd and Baker 1998). MPA networks are especially useful for protecting small or discrete local populations with well-defined habitat requirements and limited dispersal of the adult phase, such as some molluscs and crustaceans, and fish species with strong habitat linkages. At a larger scale, if a species has populations that range across separately-managed regions, it may be important to protect a portion of those populations in each region, so that genetic variation of the whole species will not be compromised due to critically high regional fishing levels over a long period. Networks of MPAs can be used to achieve this aim.

A network of MPAs can help to maintain a representative source of recruits from undisturbed habitats (Kenchington 1988, cited by Fairweather and McNeill 1993). Networks of small MPAs can also be used to link ‘source’ areas (in which larvae and other propagules reproduce) with ‘sink’ areas in which propagules collect, thus acting as ‘stepping stone’ areas that can assist dispersal and immigration (Goeden 1979, cited by Fairweather and McNeill 1993).

Small, numerous reserves, which may or may not be designated as ‘no-take’ zones within larger multiple-use MPAs, can effectively export biomass to adjacent fisheries. Networks are considered especially useful for fish populations scattered over numerous reefs; for example, many small MPAs maximise the potential for a portion of the stock to move out of those MPAs into the adjacent fisheries, according to Roberts and Polunin (1991), due to the greater ‘perimeter-to-area ratio’ of networks.

Perhaps the greatest advantage of networks versus single MPAs is that networks can increase the chances of fish stock protection and enhancement due to their greater number, spatial spread, and protection of habitat variety. Over time, it might not be known which reserves in the network were most responsible for protecting and enhancing stocks, but that information may not be important in practice (Ballantine 1990).

There are comparatively few disadvantages to properly designed networks of MPAs, compared with single large MPAs. However, two potential impediments to success include the fact that:

- the information regarding the critical connections between habitats, and between populations dynamics of different meta-populations, may not be available (Fairweather and McNeill 1993)
- the greater perimeter-to-area ratio of many small MPAs can increase the ‘transfer rate’ of mobile species out of the MPAs (de Martini 1993), depending upon the location of the MPAs and the mobility of the species. This can be advantageous to the fishery, but detrimental to heavily fished stocks that may require the use of MPAs for population replenishment.

Shape and Buffers

The discussion of reserve shape (see Diamond 1975, cited by Saunders *et al* 1991; Shafer 1990, for terrestrial examples) is pertinent to marine systems only as far as they can be described in terms of their ‘boundaries’, ‘edge permeability’ and ‘boundary-to-volume ratio’ (Davis 1990; Fairweather and McNeill 1993; Stamps *et al* 1987, cited by de Martini 1993). For example, long ‘thin’ reserves have a greater area of ‘edge’ compared with more evenly-proportioned reserve shapes, and may provide less protection for species which move ‘over the edge’ (eg become vulnerable to fishing outside of the reserve). However, habitat type, linkages and population behaviour within and between habitat types complicate the theoretically-simplistic argument for evenly-proportioned reserves. For example, if a ‘thin’ stretch of rocky reef surrounded by sand is designated as a MPA, it may be effective in protecting reef-associated species that do not forage out on the sand, but it is unlikely to assist the conservation of ‘free-ranging’ species that forage over a wide space and a variety of habitat types, and are not ‘confined’ to the MPA.

MPA shape should be aligned with the natural geometry of habitat types and fragments: important habitat types worthy of MPA designation may naturally be small (eg a rock reef), or narrow (an estuary, or a highly-productive boundary between two ecosystem types) or patchy (seagrass beds).

Regardless of its shape, the spatial area of a MPA may not be effective in protecting or enhancing populations of sedentary species (such as mollusks and crustaceans), if daily or seasonal movements (related to feeding or other habitat use) cause most of the population to move out of the MPA and into an adjacent area where they are vulnerable. Such an occurrence was recorded by Hunt (1991) in Florida, where spiny lobster moved out of the MPA every night to feed on the unprotected reef flat, and many were removed by fishers' traps during the evening. Provision of appropriate buffer zones around MPAs (or increasing the size of the MPA to encompass critical foraging, migration or breeding areas) can improve the success of MPAs in protecting sedentary stocks. Obviously, the same design principles do not apply to the foraging, breeding and / or migrating areas of far-ranging animals which are not 'bound' to one habitat type.

Starr (1998, cited by Kripke and Fujita 1998) suggested that shape for fisheries MPAs should minimise the perimeter through which ocean currents carry larvae outside of the MPA 'boundaries'.

Computer simulation models have also been used by some workers, to determine the 'optimum' size, shape and spatial arrangement of MPAs for sustaining fishery yields of particular species. Sladek Nowlis and Roberts (1995), for example, predicted optimum reserve sizes that were usually significantly larger than the size of existing reserves, and showed that if fishing intensity and MPA size were controlled, catches remained higher than would occur in a situation with under-fishing and no MPAs.

The appropriate shape of MPAs can also be influenced by the availability of area for 'buffer zones' around the core protected area. Batisse (1986, 1990) discussed the importance of reserve buffers, utilising globally accepted and successful 'biosphere reserve' design principles, which encompass a core protected area, a buffer zone, and a transition / multiple-use managed zone (see Section 3.2.1). Although such terrestrial designs are not always suitable for marine areas due to greater connectivity in marine systems (Kenchington 1991), the concept of buffered, protected core areas in a matrix of 'whole system' managed areas has great utility for marine conservation and multiple-use management. The core / buffer / multiple-use model is becoming increasingly popular in MPA design all over the world, as discussed in Section 3.2.1. Large, multiple-use MPAs in Australia usually incorporate buffers around core protected areas, through a zoning scheme that regulates activities in each of the MPA zones (eg 'core', 'buffer' and 'general use' zones). For example, the 'core' no-take component of the new Macquarie Island Marine Park is a single, highly protected central corridor (approximately 5.71 million ha) flanked to the north and south by two buffer areas (defined as 'highly regulated species / habitat management areas'), of 2.7 million ha and 7.7 million ha respectively. The boundaries of the Macquarie Island Marine Park are based largely on a combination of available oceanographic data; the ecology of species in the region which have high conservation value (eg feeding and breeding areas, behavior and migratory routes of seabird and seal species); and current fishing grounds (Environment Australia 1999, unpublished comment to Californian Marine Protected Areas Network). This zoning approach combines physical data with biological, ecological and social considerations.

Location

The most appropriate locations for fisheries MPAs will depend upon their specific purposes (Roberts and Polunin 1991). For example, if the aim is to protect critical sites for larval settlement, then information at the appropriate scale regarding larval 'source areas' and larval dynamics should be available (see Roberts 1997b for Caribbean examples of 'source' areas for fisheries replenishment). Similarly, MPAs for provision of juvenile recruits to fished areas obviously must be situated to maximise replenishment potential, and knowledge of the distribution and movements of the target species can assist such MPA placement. If the aim is to protect 'high quality' habitat where abundant resources result in higher densities and abundance of biota than in 'low quality' habitats (Buechner 1987, cited by Roberts and Polunin 1991), then the distribution of such habitat must be known. This aspect of MPA design has received increasing attention (eg 1998 Mote Symposium).

Recent models have suggested ways of optimising MPA locations for the protection of 'essential fish habitat' whilst simultaneously sustaining or enhancing fisheries yields (eg Mangel in review, St Mary *et al* in review). Results from a model of MPA design for the protection of juvenile groundfish (Lindholm *et al* 1997) suggested that design (eg location, size and boundaries) should be sensitive to the distribution of critical benthic habitat, the impacts of mobile fishing gear upon such habitat, and the juvenile migration rates out of the protected area. Section 6.2.2 provides a more detailed discussion of the relation of critical habitat (and its patchiness and fragmentation) to MPA design.

Temporary versus Permanent MPAs

Temporary MPAs (eg seasonal closures) can be used to protect larval source areas if the seasonal patterns of supply and settlement are fairly consistent. Temporary MPAs can also protect population size and density during a critical spawning period. In abalone, for example, fertilisation is enhanced by the aggregation behaviour of animals during the several weeks in which spawning occurs, according to Shepherd (1986), and temporary MPAs could be used to protect meta-populations during this season. Another example would be the closure of seagrass beds to trawling during periods in which recruitment is high, or the seasonal closure of reefs during spawning aggregations (eg sedentary reef fish, cuttlefish). Temporary MPAs can therefore be useful when the distribution and population dynamics of local populations are well known. Seasonal closures may also be effective for migratory populations, or those with spatially separated life stages (eg spawning aggregations in reef habitat). For example, an area could be seasonally protected during the seasonal spawning periods in which a population visits a particular site.

Seasonal closures may also be rotated over a broad region, according to the movements of population numbers and their interactions with the fishery. This could be useful for populations with widely scattered individuals that utilise a variety of habitat types (ie rotating reserves could benefit fisheries by maximising 'contact' of the MPA-enhanced population numbers with local fisheries (Roberts and Polunin, 1991). In contrast, 'fixed' MPAs might not 'export' fish to the fishery if the population was wide-ranging with no strong habitat linkage.

However, there is some evidence that seasonal closures to protect (for example) spawning populations may be ineffective if the location and reasons for the closed areas are not well accepted. Keesing and Baker (1998) provided an extreme example where temporary closure of an area for abalone meta-population replenishment drew fishers' attention to the site, and resulted in higher yields in the region during the closure than at any other period in which yields have been recorded. Obviously, temporary closures will be most effective if fishers appreciate the rationale for the closure, and comply with those conditions. Another concern with temporary MPAs, such as rotating seasonal closures, is that although specific fisheries benefits might accrue over time, such a system can be detrimental to the goal of biodiversity protection (Fogarty 1999), if the areas in the rotating system are opened to various destructive fishing techniques that target many different species. Clearly, rotating closures may be more suitable for protecting single species that are targeted by low-impact techniques (eg mollusk or urchin harvesting).

Permanent MPAs are most appropriate for critical habitat types in which population dependence is known. Examples include the strong link between juvenile fish and estuaries, or sedentary species with reef habitats, or locations associated with a significant process that affects the life stages of many different species (eg local current patterns in a bay, causing the larvae of many different species to concentrate and settle out of the plankton).

6.2.4 Proportional representation for fisheries MPAs: potential problems

The 'ideal' size of MPAs for single species conservation may not accord with objectives for bioregional / ecosystem representation (see Section 6.1). For example, single species modelling work may recommend that 20% of a spawning stock be reserved using a MPA, but this does not imply that conserving 20% of the habitat / ecosystem type in which the species spends part or all of its life will serve the same conservation purpose for that species. Sladek, Nowlis and Roberts (1997, 1999) suggested that in order to enhance fisheries, MPA sizes of 40% of the fish populations' management area would be required, for fish with strong 'site-attachment' that do not cross the MPA boundaries. The modelling results of Lauck *et al* (1998) suggest that MPA sizes that protect 50% of the population will enhance the total catch as well as protecting the stock. In practice, protecting 50% of a population would result in a wide variety of MPA sizes, depending on the population in question, and some of these MPA sizes may be completely infeasible. For example, protecting 50% of a population of an over-fished, site-attached species of limited distribution may require a MPA of an area equal to a few square kilometres. In contrast, conserving 50% of a wide ranging, broadly distributed, migratory stock with catholic habitat connections could equate to a few thousand square kilometres.

In addition to the 'no-take' question, another problem is the decision about what proportion of an area should be 'limited take'. For example, it has been suggested that 10% of certain 'areas' (unspecified size) should be closed to destructive fishing methods such as trawling and netting (Barrett pers. comm., cited by Major 1998). It is a difficult and contentious issue to decide which 10% of an area to close, or to even decide the actual percentage of an area to close, to achieve benefits in fisheries' sustainability and other goals such as biodiversity protection.

Socio-political and other pragmatic constraints to theoretically ideal single-species MPA sizes will likely preclude such a system from being established. For example, some regional fisheries require that fishers be restricted to operate in certain parts of the region and not others. Closing off 40% (or even 20%) of a country's total fishing grounds for any single species will favour some fishing operations over others, and is unlikely to be publicly accepted.

6.3 SINGLE-SPECIES FISHERIES MPAs VERSUS MPAs FOR ECOSYSTEM REPRESENTATION

Some marine taxa share similar life history patterns; the same coastal processes affect their population dynamics and persistence, and they utilise the same habitats. Therefore, MPAs designed to protect specific populations (such as commercially valuable species) can also protect many other species and thus assist in overall 'biodiversity conservation' goals.

The opposite is also true: many taxa within one ecosystem / habitat have entirely different life history phases and population dynamics patterns, different levels of mobility and modes of dispersal, and different critical habitat requirements throughout their life phases. Some taxa may be confined to that habitat; others may 'pass through' seasonally or irregularly. Thus, the single-species MPA design approach is obviously implausible for all but a few species, and is one of the reasons why representation of bioregions and ecosystems / habitats is favoured by many policy-makers as a sensible basis for MPA design.

Another disadvantage of single-species fisheries MPAs is that they are often designated for the most economically and socially 'valuable' species: if circumstances change, and those species become less favoured, then the long-term viability and management of the species-specific MPAs may become socially and politically contentious.

Conversely, habitat-based MPAs that are designed to protect many different species (such as area closures against trawling, etc) may not adequately protect particular species populations, depending upon the habitat requirements and linkages, mobility and larval dynamics of those populations. Chapter 11 provides an example of a multi-species MPA that failed to protect a specific, economically important species population from decline.

Although single-species MPAs can be used to protect 'habitat-bound' populations, and those with limited mobility and / or well-defined ranges and well-understood population dynamics, such MPAs serve limited goals. Ecosystem / habitat-based MPAs that can be used to conserve a larger number of species are now becoming more common, particularly as the basis for MPA networks (see Section 6.1.7). Such 'multi-species' MPAs are appropriate in areas which (i) function as critical habitats, serving the resource needs of a variety of species, and (ii) are centres for critical regional or local processes that affect a variety of species. An example of the latter would be a bay in which the larvae of many different species tend to concentrate, due to local topography and current patterns.

Ultimately, the choice of designating single-species MPAs versus ecosystem-based MPAs usually depends upon a number of social, economic and political decisions. For example, the value of protecting a particular fisheries stock using single-species reserves may be contentious, compared with the protection of a greater number of species and habitats, through designating a similar sized MPA in a different location. Further, it may be infeasible or social unacceptable to close specific areas in which competing uses and interests exist (eg aquaculture, commercial and / or recreational fishing, conservation).

A well designed network system of MPAs (see Section 6.1.7), would include representative ecosystem types as well as protected areas for mobile fauna with specific habitat linkages. The large number of areas that are required to serve all the conservation objectives for particular economically and socially valued species, inevitably results in some compromise to an ideal network. Furthermore, many species (particularly highly mobile species) require other conservation measures in addition to MPAs, such as adequate management of fishing and bycatch levels (see Chapter 9).

6.4 MPA DESIGNS BASED UPON RESERVE SELECTION ALGORITHMS

This approach originated in terrestrial systems, and utilises various iterative algorithms and other programming formulae to choose the optimum number, size and placement of MPAs, to satisfy a number of defined conservation criteria. During the past decade, the application of both heuristic and optimising reserve selection algorithms has burgeoned in terrestrial conservation science (eg Kirkpatrick 1983; Margules *et al* 1988; Cocks and Baird 1989; McKenzie *et al* 1989; Pressey and Nicholls 1989; Pressey *et al* 1993 and 1996; Possingham *et al* 1993; Margules *et al* 1994; Price *et al* 1995; Williams *et al* 1996; Woinarski *et al* 1996). One of the main aims of this approach is to choose 'minimum sets' of reserves to satisfy a number of conservation objectives, and, in some cases, to specify the design criteria other than the number of those reserves (size, shape, location, adjacency, etc) that can achieve the objectives. One of the commonest conservation objectives in algorithm-based reserve design during the past decade has been the representation of biodiversity, although in practical terms, protected area placement usually considers many other criteria, as discussed in this report, and few reserves for biodiversity protection *per se* have been designated to date. The 'minimum set', 'optimal set' and 'maximum coverage' approaches operates according to principles such as:

- 'complementarity' between sites or taxa in the network (eg in terms of species richness, family-level richness, species rarity, representation of habitat types, etc
- 'efficiency', or minimum redundancy, of reserve selection algorithms, to represent any of a number of objectives (such as biodiversity conservation) in the minimum number of reserves
- 'irreplaceability' of taxa or sites in reserve networks.

Pressey *et al* (1993); Williams *et al* (1993); Williams and Humphries (1994) and Forey *et al* (1994) discuss these concepts in more detail.

There is debate amongst workers in this field about the comparative merits of various heuristic algorithms and optimising routines, to determine which can best satisfy the principles of efficiency, complimentarity, and irreplaceability. Some of the simpler reserve selection algorithms can be used in GIS, and some of the most widely-known examples are the automated routines for biodiversity summation and spatial mapping available in the WORLDMAP program from Britain (Williams *et al* 1993; Pressey *et al* 1993 and 1996). To represent biodiversity, such programs require knowledge of (i) the number of species or other taxonomic units per sample ('richness') and / or the degree of taxonomic difference between the taxa, or (ii) the number of ecosystem / habitat types, if these measures are being used as a 'surrogate' for biodiversity estimation. WORLDMAP has been used to iteratively select a complementary set of potential reserve sites which satisfy a number of biodiversity goals, such as maximum species representation (usually within a small taxonomic group eg Kershaw *et al* 1994), or maximum representation of taxonomic diversity, with or without site weightings according to presence of rare / vulnerable / endangered, uncommon or endemic species. The program can also be used to assess 'tradeoffs' in representing various goals in the minimum reserve set, considering that the distribution of species-rich sites and sites with high numbers of rare or threatened species may not often coincide.

Although the use of reserve design algorithms has been a popular theoretical approach to terrestrial reserve design problems since the early 1990s, only recently have some workers applied this approach to the design of MPA networks. The application of reserve design algorithms in MPA siting on will not be discussed in this report. As is the case with an alternative data-intensive scientific method of MPA design and selection, discussed in Section 6.5, the utility of objective scientific methods of reserve design such as these, is highly contingent upon the choices and quality of input data types. If those data are poor quality, then an advanced numerical analysis for reserve design may not necessarily provide worthwhile results.

6.5 MPA DESIGNS BASED UPON MULTI-CRITERIA DECISION MODELLING

6.5.1 Background

Multi-criteria (also called multi-attribute) decision modelling has been used terrestrial environmental planning since the early 1990s (eg Eastman 1993; Eastman *et al* 1993) to help solve land allocation problems, including the placement of protected areas and buffer zones. In simple terms, decision modelling refers to a set of numerical procedures, by which various criteria are combined to satisfy one or more objectives. The objectives may be complementary, or conflicting, and in the latter case, some form of prioritisation must be involved with the solution.

During the past decade there have been rapid and significant developments in methods of multi-criteria evaluation and decision modelling. Some of these methods, based upon linear and mixed integer programming (eg Janssen 1992) were restricted to the domains of mathematicians and computer programmers, and have not been suitable for the large spatial data sets (eg rasterised grids of remotely sensed data) that typify land-use analyses. Also, most mathematical programming methods can satisfy a limited number of competing objectives. Furthermore, some developers of heuristic methods (such as Eastman, cited above), have considered that in land-use planning, mathematically optimal solutions based upon linear programming may not always provide 'realistic' solutions that are acceptable to various decision-makers. Other techniques, that are more suitable for analysing data sets in GIS, have now become accessible to a wider audience, particularly through the commercially available GIS package IDRISI from Clark University in USA. For example, decision-modelling techniques incorporated into the Multi-Criteria Evaluation module of IDRISI are based upon Boolean analysis, and / or (i) weighted linear combinations or (ii) ordered weighted averaging of standardised 'input' factor grids, with or without the use of Boolean constraint maps to limit areas under consideration. Various weighting, ranking and classification methods are available in IDRISI for resolving the 'spatial conflicts' that occur with multi-objective problems. When few objectives are involved, Environmental Systems Research Institute's (ESRI)'s ArcView Spatial Analyst also has some capacity for performing multi-criteria decision analysis, using simple mathematical preference criteria and rule-based restraining criteria on classified and ranked suitability maps. One of the technical challenges during the next few years will be the development of improved methods for using vector and point data (eg of socio-economic criteria) in decision-analysis modules that were originally designed for use with raster data sets (eg remotely sensed physical or biological base maps). Despite the sophistication of these computerised aids to decision-making, there is still subjectivity involved; for example, in the choice of 'importance ranks' for input data, the set of weightings assigned to each criterion, and the order of priorities in allocation decisions.

6.5.2 Potential application for MPA design

Marine and coastal spatial data were gathered in many countries during the 1990s, and for applications requiring few data sets and maps, simple visual inspection of overlaid maps that highlighted 'areas of conflict' in land use were often sufficient for management decision-making.

However, for decisions that require consideration of numerous coastal marine environmental types, uses, activities, interest groups and issues, more sophisticated assessment methods are usually required. To date, there appears to be limited knowledge and use of GIS-based multi-criteria decision analysis amongst marine workers. At the time of writing this report, there were no available published examples of GIS-based multi-attribute decision modelling specifically used to assist the siting of MPAs. In some instances, related techniques have been used in marine planning applications, but there are very few examples:

- GIS-based spatial marine data analysis has been used to identify map suitable sites for aquaculture development (eg Ross *et al* 1993), by selecting sites that satisfy a set of 'combined factor' criteria, based upon physical, biological, and socio-economic data sets.
- In the US, the National Oceanographic and Atmospheric Administration has monitored coastal benthos, wetlands and adjacent upland areas every two to five years since the early 1990s, using remotely sensed data to establish a dynamic land cover database. The database also documents historic change, and can be used to evaluate previous, current and future coastal management strategies. The GIS-based data enable researchers and managers to link developmental change in the coastal zone to ecological effects and economic productivity in fisheries (Klemas *et al* 1993).

- 'Exclusion mapping' using multiple spatial constraints (eg existence of ecologically sensitive areas, bottom-trawling grounds, shipping lanes, etc) has also been used for the siting of artificial reefs (Gordon Jr 1994).
- In France, the coastline, marine sediments and vegetation of the Loire Estuary have been classified using SPOT and Landsat TM images, and data pertaining to commercial biota (fishes, mollusk distribution, etc) and socio-economics have been overlayed and classified, to form an index of 'resource sensitivity', for use in oil spill contingency planning (Populus *et al* 1995).

An Australian example of a comprehensive spatial data set that can be used for MPA design is the application developed for Jervis Bay (Dutton *et al* 1994), in which 30 physical, biological and socio-economic data layers have been assembled. Most layers are rasterised, but there is also provision for vector-based inputs (such as point and polygon coverages of socio-economic data such as heritage sites, or important commercial and recreational zones). Some of the physical data sets include geology, coastal watercourses and depth contours, and examples of biological data include coverages of seagrass and macro-algae, seabird habitat, distribution of sediment fauna, and location of rare or threatened species. Socio-economic data include proposed aquatic reserves (ie MPAs), location of Aboriginal heritage sites, popular dive sites, commercial and recreational fishing zones, and recreation resources, among others. The database combinations in the Jervis Bay GIS can be used to assess various conservation options, according to spatially explicit criteria such as habitat connectivity, vulnerability to impact, and integration with adjacent land use. In this case, the site suitability assessment was based upon a goals-achievement matrix method, and synthetic 'output' layers can be derived for conservation management, including 'core protection' sites (ie such as no-take sanctuary MPAs), recreation zones and multiple-use zones.

Determining the location of suitable MPA sites can be based upon similar principles used in the 'combined factor' / 'exclusion mapping' examples cited above. Multi-criteria evaluation is particularly relevant to the solving of 'real-world' siting problems, in which pre-existing (*ad hoc*) MPAs may exist, and also, a number of conflicting user groups and activities must be considered in decisions about siting new MPAs. During the 2000s, it is likely that many of the comprehensive, multi-attribute data sets that have been collected in marine systems during the 1990s, will be better utilised, with the aid of multi-criteria decision analysis, for the selection of suitable MPA sites.

Section 6.1.1 provides examples from several other countries of ecosystem / habitat-mapping using remote sensing and GIS methods. Such maps can be used as key components of the set of the base-maps that can be overlaid with socio-economic data in point and polygon format, as inputs to decision models that assist in MPA site selection.

Multi-attribute GISs are data-intensive, and rely upon high quality input coverages, which can limit their application to those areas for which such data are available. In Australia, given the wide acceptance of the MPA selection criteria employed for the national representative system, it should be possible to spatially classify and map areas according to a number of recognised conservation criteria, such as those devised by the IUCN (see Chapter 4). This could include maps of biogeographic regions, ecosystems / habitat types (derived from analysis of both remotely-sensed and field-based biological and physical data), overlayed with various socio-economic coverages in polygon or point format (according to the spatial area of the attribute). Such coverages could include cultural and historical features, recreation zones (eg significant locations for diving), sites of scientific importance, sites of economic importance (eg for fishing or aquaculture), and location of threats (such as sewage or industrial discharge points) among others. Such coverages could be used as inputs to GIS-based multi-criteria decision models, using methods discussed earlier in this section.

Some MPA workers are now accepting techniques of spatial analysis such as those outlined above. For example, Fogarty (1999) considered that remote mapping and the 'new analytical approaches to quantifying marine habitat suitability' have important implications for the siting of MPAs.

6.6 OTHER SCIENTIFIC APPROACHES TO MPA DESIGN

BIORAP (Margules and Redhead 1995; Faith and Nicholls 1996; Hutchinson *et al* 1996; Noble 1996) is another set of GIS and spatial analysis tools that can be used in 'priority areas analysis' (ie reserve selection). Amongst other modules that assist in mapping biodiversity, BIORAP includes modules for (i) multivariate pattern analysis of biological and physical variables; (ii) interpolation methods for environmental data sets; and (iii) tools for choosing sets of potential reserves, and calculating the 'costs' associated with choosing alternative sites and placements for reserves within the set. The utility of automated biodiversity mapping and reserve selection tools is highly contingent upon the quality of available data sets, and the ecological assumptions that must be accepted when using the techniques. There is potential for error in application, particularly when extrapolating data to unsampled areas, aggregating disparate data collected at various scales (and using different techniques), and when undertaking multivariate and/or priority area analysis without considering the sensitivity of results to chosen data sets, units of measure, sampling scale, and other factors associated with both data 'inputs' and techniques of analysis. Another concern, particularly when trying to identify potential reserve areas to represent biodiversity, is the questionable validity of using mapped environmental variables as 'surrogates' for biodiversity, particularly when the concordance between the physical and biological patterns is not known (which is usually the case in marine ecosystems). Vanderklift *et al* (1998) and Ward *et al* (1999) provided the first published example of the use of the BIORAP tools for potential placement of MPAs.

Commercially available ecosystems models, such as ECOSIM and ECOSPACE, have also recently been used to assess the utility of MPAs in the context of (i) the trophic interactions required to maintain exploited fish populations and (ii) the habitat preferences of fished species, and effects of degraded habitat quality (Walters *et al* 1998). Recently, Walters (in review) also used ecospace to evaluate MPA design criteria (such as size) for conserving and managing fisheries species and protecting local scale biodiversity. ECOSPACE modelling work has been used to demonstrate the ecological consequences of alternative MPA design strategies, incorporating both biological and socio-economic data.

6.7 *AD HOC* MPA DESIGNS

Little will be stated about this approach, even though it was the most common way of identifying and selecting potential MPAs until the early 1990s. It is likely that many countries now have reduced opportunities, in a socio-political sense, for designing a more representative network of MPAs, due to the *ad hoc* placement of MPAs during the past two decades, to satisfy specific objectives. Such objectives often included preservation of attractive dive locations; conservation of socially-valued endangered species; and declaration of estuarine nursery MPAs for fish stock conservation. *Ad hoc*, single-objective MPAs were often designated irrespective of habitat quality, present impacts, future threats, or management potential. Chapters 2, 3, and 4 discuss some successful MPAs that resulted from *ad hoc* placement, and Chapter 11 discusses some failures resulting from this approach.

6.8 MPA SELECTION

Volumes could be written about MPA selection criteria, and an exhaustive analysis is beyond the scope of this report. Each State and country has different goals to achieve by establishing MPAs, and therefore both design and selection criteria can differ greatly at national and international levels. Selection criteria for Australian MPAs have been detailed in Ivanovici *et al* (1993), Brunckhorst (1994), Thackway (1996) and the Commonwealth's *Strategic Plan of Action* and the *Guidelines for establishing the National Representative System of Marine Protected Areas* (ANZECC TFMPA 1999). In South Africa, guidelines for MPA establishment (including selection criteria) were formulated as early as 1976, by the Marine Reserve Committee (Attwood *et al* 1997).

In some parts of the world, it has been proposed that community 'stake-holders' be entirely responsible for the identification of potential MPAs, as well as their selection / placement (e.g. Brody 1998, for the Gulf of Maine). Community-driven MPA establishment eschews the 'top-down' approach of government-driven MPA nominations, which are often based upon biogeographic and ecological classifications, and other scientific data that supports an ecosystems-based approach. Although the merits of a 'bottom-up' approach driven be stakeholders are obvious, regarding public sense of 'ownership' and 'stewardship' of MPAs, as well as the potential for greater acceptance and compliance with MPA regulations, this has not been the approach recommended by the Commonwealth Government for establishing the NRSMPA in Australia, for several reasons. Although community groups provide important cultural, social, economic, and local ecological knowledge for the MPA designation process, and the input of stakeholders is now widely accepted as an essential part of that process, it is unlikely that an entirely community-driven approach would find wide acceptance, in countries which aim to achieve a comprehensive, adequate and ecologically representative network of MPAs. Community nominations are often based upon sectorial interests; local-scale environmental concerns; or sites with particular socially-valued or economically-valued features or resources. Although such concerns should be considered in the design and adequacy of the final MPA network, they are not generally accepted as being suitable driving forces for systematically establishing networks of MPAs for ecosystem and biodiversity protection. An entirely community-driven approach to new MPAs can perpetuate the *ad hoc* system of marine protection that many countries have tried to overcome during the past decade. Furthermore, locations (and ecosystems) that are not socially valued, but nevertheless have important physical, chemical, ecological and biological functions in the marine environment, are likely to be ignored if the process of MPA designation is entirely driven by community interests. It is also likely that many countries and States, which have very low levels of MPA-based marine environmental protection, do not have a similarly low level of community / sectorial group representation in the uses and activities that currently occur in their marine environments. In effect, ecologically representative MPA networks aim to redress that imbalance, by protecting some areas from the extractive uses and other damaging activities that already widely occur, thereby assisting the goal of protecting marine ecosystems and their component biodiversity. Therefore, a 'socially and economically representative' MPA system is more appropriately achieved through the equitable zoning of limited-use and multiple-use areas, and should not be the overriding rationale for MPA designations, at the expense of ecosystem and biodiversity protection.

In Australia, the Commonwealth Government (ANZECC TFMPA 1998 and 1999) has adopted what is hoped to be both an ecologically considerate and socially acceptable approach, that does not entail either extremes of the 'top-down' or 'bottom-up' approaches. Potential lists of network MPAs that satisfy the primary goal of the NRSMPA have been identified on ecological grounds (eg as part of the past decade of work that has been undertaken in Australian States for the NRSMPA), but the final selection of MPAs from those State lists will consider social, economic, cultural / indigenous and other interests, following detailed phases of community participation in the MPA selection process in each State. Obviously, in the development of a comprehensive, adequate and representative (CAR) system of MPAs, selection of the final set of MPAs on socio-economic grounds, cannot ignore the ecological imperatives that were used to identify the candidate set. Therefore, in reality, the ecological identification criteria and socio-economic selection criteria (as advocated by the Commonwealth) cannot clearly be separated in the development of the NRSMPA based on CAR principles.

Conflicting priorities during the MPA selection process are discussed below.

6.8.1 Insufficiency of design criteria

To date, MPA selection processes have often been hindered by competing decisions about which design criteria are most relevant, considering that such criteria are often specific to particular MPA objectives (representation of ecological units, for example, compared with fisheries stock protection and enhancement, as discussed in previous section). It should be stressed that any of the major methods of MPA design discussed in previous sections may, by themselves, be insufficient to guide the MPA selection process.

For example, although bioregional maps can be used to identify regions of biogeographic and ecological significance, protection of such areas is only one of the many goals of MPA establishment. Similarly, optimally-positioned reserves for protection and management of commercially or recreationally valuable species, also satisfy a limited number of objectives regarding MPA designation. For example, optimising MPA design for single species, according to any specified criteria, such as larval sources and sinks (see Section 6.2.2), is an unworkable approach for a representative system of MPAs, even if a large number of no-take MPAs (each optimised for different species life history strategies) are implemented. More realistic approaches may include a number of single-species 'no-take' MPAs, within larger, multiple-use MPAs (eg marine parks, biosphere reserves), or networks of ecosystem-based MPAs that can serve a number of conservation and management objectives (see Section 6.1.7).

New methods of 'objectively' considering multiple objectives in MPA design may offer some improvement (ie the reserve design algorithms and multi-criteria decision modelling approaches discussed above), and have blurred the distinction between MPA design and MPA selection processes. Although these methods may be efficient for prioritising sites for selection, according to competing objectives, one of the major concerns is the high sensitivity of results to the type, format and amount of input data, particularly 'ecological criteria'. It is difficult to adequately categorise and compare ecological criteria between sites, considering that there is often unequal sampling between sites that are being compared. Therefore, well sampled sites (for which many biological and ecological features are known and recorded) usually score above those sites which actually may be of equivalent ecological value, but have been poorly sampled and thus have few features scored and considered by the model. Other concerns include the scale and adequacy of ecological mapping used as a base layer in such models, and the fact that important reserve design considerations currently cannot be adequately incorporated into such multiple-objective methods. These include source and sink connectivity, multiple habitat usage, and other ecological connectivity between habitats; benthic versus pelagic conservation decisions for each area; the need for multiple reserve sizes within a single network, to meet different ecological conservation objectives; buffers around core protected areas and other aspects of nested designs; and designs for effective threat/impact management.

Apart from data concerns, even a theoretically 'ideal' network of MPAs that can be modelled to satisfy multiple objectives and minimise conflicting uses and activities (see Sections 6.4 and 6.5) may not easily be designated in practice. That is, the final 'optimum' list or network of MPAs that satisfy the greatest number of objectives, and minimise the largest number of conflicts, can be completely unrealistic for actual MPA designation, even if 'practical' criteria (such as adjacency of selected sites) are included. The number of designated 'no-take' MPAs (for whatever purpose) is usually a socio-political decision, and consequently the number and type of ecosystems and communities and populations that MPAs protect usually fall short of the ideal specified by design criteria.

It is not unrealistic to expect that in many countries the implementation process may revert, through practical constraints, to one of incremental placement of (say) one or two MPAs over several years, considering (i) the severe social, economic and political limitations to implementing theoretically ‘idealised’ networks, and (ii) the long lag between planning and placement (see Chapter 8).

6.8.2 The role of ‘representativeness’ in MPA selection

IMCRA (see Chapter 4) and the State level bioregionalisations (see above) are useful in identifying the biogeographical and ecological gaps in Australia’s current MPA network, and for compiling a candidate list of potential MPAs to ensure a comprehensive, adequate and representative system. However, the selection process for potential MPAs cannot (and should not) be based solely upon biogeographic or ecological representativeness.

Wilson (1995) recognised that *representativeness* is only one objective of MPA establishment, even though it has been the driving reason behind MPA planning by Commonwealth and State Governments in Australia during the past decade (see Chapter 4). The selection process brings many other ‘non-ecological’ considerations into play. Wilson (1995) aptly summarised the distinction between the technical aspects of bioregional classification or other methods of identifying ‘representatives areas’, and the process of MPA selection:

> *‘I have great faith in the power of informed people to make good judgements . . . once the available information on the biota and geomorphology has been processed by whatever method, the time comes when knowledgeable people have to debate the choices and make decisions. Machine analysis and objectivity then dissolve into the shadows.’ (Wilson 1995, p. 174).*

There are many social, economic and political constraints in MPA selection that often override ecological issues such as ‘representativeness’ (Salm 1989, McNeill 1994), and these constraints can reduce suggested design criteria (in terms of size, shape, number, buffers, linkages, etc) to a theoretical ideal.

Social, economic, cultural, and other data relating to marine uses and activities, user groups and management are required for MPA selection. Consultation with stakeholders is now considered a key part of the MPA selection process (see Thackway 1996; ANZECC TFMPA 1998 and 1999). The social, cultural and economic criteria that must be considered during the selection phase of MPAs should not be underestimated. Such data are highly relevant for the practical designation and management of all kinds of MPAs, including those that have the major objectives of representing ecological units; protecting areas for fish replenishment and fisheries management and enhancement; or for any other major purpose of MPA designation. Detailed discussion of these criteria is beyond the scope of this report.

6.8.3 International and national examples of MPA selection processes

In South Africa, Hockey and Branch (1997) recommended a scoring approach for MPA selection. This approach, called the COMPARE ranking system, matched 17 distinct criteria (such as regional representativeness, habitat diversity and quality, conservation value, satisfaction of social needs, and effectiveness for management, with a set of MPA objectives (encompassing species and habitat protection, fisheries management objectives, and ‘utilisation’ objectives, such as recreation, education, research and monitoring, among others).

In New Zealand, Dr Bill Ballantine has devised a set of principles for designing MPAs (called reserves in that country), which espouse ecological representation, geographic and ecological network design, replication, conservative classification levels, and natural sustainability of reserve sites. The basic premise for the system of reserves in New Zealand is that reserves be 'no-take', permanent, minimally disturbed, and available for low-impact uses such as non-exploitative recreation, scientific research and teaching/education. Since the late 1990s, New Zealand's Leigh Laboratory has run a course in marine reserve design and selection, which is the first program of its kind in the world.

In Australia, not all States have completed the selection phase of the NRSMPA, and there is much debate as to the methods that should be used (eg scientific methods such as multiple-criteria decision analysis or reserve selection algorithms, versus the 'Delphic approach', community consultation, and ranking of ecological and socio-economic criteria, as applied in Western Australia).

The approach to selecting a candidate list of MPAs in Western Australia was based upon the desire to represent, at least once, each of the distinctive coastal types (and all geomorphologically-defined habitat types within those coastal types). Where there were significant regional variations in physical and climatic features, examples of a specific habitat type at several points along the coast were selected, to cover the likelihood of regional variations in biota. (Note that biotic distributions were not used for the bioregionalisation process, and geomorphological classification was used as a 'surrogate'—see Section 6.1.3.) The more 'subjective', but equally important, phase of the selection process was to canvass 'anyone and everyone' (Wilson 1995) to identify conservation hotspots, important breeding and nesting sites, feeding areas for socially significant taxa (eg dugongs), socially-significant diving spots and other recreational areas, areas of industrial and commercial importance, etc. More recently, the Department of Conservation and Land Management (CALM) in Western Australia has utilised a multi-criterion prioritising framework for determining suitable MPA sites, based upon existing biophysical, ecological and socio-economic data.

The approach to MPA selection in Queensland will utilise cultural, ecological, economic, social, legal and practical criteria, gathered during a detailed process of public consultation. These data will be used to determine the 'least cost' to the community of alternative MPA site selections (with 'cost' being defined in a variety of social, economic and conservation terms) (GBRMPA 1999). Both representative (major habitat types) and unique areas (eg spawning grounds, nursery sites) that were nominated during the identification phase, will be considered in the selection phase, and the number of 'no-take' MPAs ('green zones') will be increased to better protect site-specific values that may be biological, social or cultural (GBRMPA 1999).

6.9 TEMPORAL CHANGE

Coastal ecosystems and species populations are not static, and are continually subject to numerous physical, chemical and ecological variables (and impacts) that promote change, and this also has implications for MPA design and performance over time. Over time, for example, the distribution of certain species could change according to gradual increases in average water temperature, salinity or nutrient loads close to the coast. If the area subject to that change is a MPA, and the population(s) it previously protected move to another area, then the boundaries of the MPA need to be 'flexible' over time. In some cases, the location of the entire MPA may need to be revised.

An example of physical coastal change that could affect the long-term viability of a MPA would be the geomorphological and locational change of river mouths, which may be especially marked in non-urban, high volume estuarine areas. (Carey, pers. comm., 1995). If a MPA is sited close to such a dynamic estuarine system, then the MPA boundaries need to be responsive to such change in the long term.

Methods of assessing change must be utilised, and MPA location may need to be periodically reviewed in light of analyses of temporal change. An example of assessment technique is the habitat mapping from the Dominican Republic, which used remote sensing and GIS technologies (Stoffle *et al* 1994). In this work, a historical suite of Landsat images and aerial photographs were analysed (and ground truthed using sonar and GPS-based) to highlight temporal changes in the distribution and density of key marine biotic coverages, such as seagrass, corals and mangroves. The changes, which had occurred over 20 years in these habitats, were ecologically significant. Such results have implications for MPA placement and performance over the longer term.

7 USER GROUP CONFLICTS IN MPA ESTABLISHMENT, AND POTENTIAL SOLUTIONS

As coastal populations increase and marine environmental quality declines, near-shore marine areas have become increasingly coveted by a variety of often competitive interest groups, including commercial fishers, recreational anglers, tourists / recreational users, indigenous fishers / gatherers, conservationists, coastal developers, and aquaculture industries, the latter establishing an increasingly strong presence in many countries since the early 1990s.

If MPAs are proposed or situated near centres of human activity, conflict inevitably arises between user groups, and often between establishing agencies and user groups collectively. Vested interests in marine environments are even more problematical than land tenure (Elder 1991), because legal and institutional arrangements are still ill-equipped to deal with competing groups that vie for exclusive use, or at least, exclusion of most other uses.

Conflicts arise through the exclusion of activities in MPAs, particularly those of high economic value, and also from restrictions that are perceived as ‘unfair’, by favouring the activities of some user groups over others. For example, opponents of coastal marine MPAs that permit non-extractive (‘look but don’t touch’) activities might object to the fact that one economic activity (fishing) is being replaced by another (diving and associated services). Proponents of such MPAs might state that fishers have been reaping benefit from almost all parts of the near-shore marine environment over large space and time scales, and that other, non-extractive users of the marine environment should be able to enjoy some areas that are undamaged by fishing. Conservationists might state that neither extractive nor non-extractive activities should be permitted in MPAs. Indigenous users might state that marine area ownership and tenure, and sustainable use of marine resources have been part of coastal Aboriginal culture for millennia, yet these indigenous marine management systems are not recognised neither by Australian marine law nor by the groups responsible for MPA planning and establishment.

In some cases, the rationale for prohibited activities in MPAs is obvious; for example, the need to exclude extractive activities from MPAs that are designed for habitat protection, or the need to exclude fishing from MPAs that are designed for fish stock replenishment. However, some MPAs exclude commercial fishing, but permit recreational fishing and recreational usage, such as boating and diving. If the number of recreational fishers is high, then the same proportion of fish could possibly be removed periodically from the MPA as would occur from commercial activity. High rates of visitor use could also potentially damage MPA habitat to the same extent as would gear from the excluded commercial activity. In such cases, the rationale for favouring one user group over another is often not justified of publicised, and the achievement of MPA goals through such management decisions are rarely demonstrated.

Conflicts within user groups are as difficult to solve as conflicts between groups. For example, fisheries MPAs that are preferentially placed in near-shore environments for convenience, ease of management or recreation / tourism benefits, can disadvantage and displace near-shore fishers who do not have access to offshore areas. If other fishers have the equipment to target the same species further away from the coast, conflict arises because one fishing group is being favoured over the other. MPA planners should obviously be aware of the fact that size and choice of location for MPAs can have significant social consequences.

It is particularly difficult to ‘sell’ to many user groups the concepts of using MPAs for bioregional representation, ecosystem protection, biodiversity conservation, fisheries management, and ecological sustainability, even if those groups support the concepts in principle. In New Zealand, for example, a survey of two groups of people (living near, and away from, a proposed MPA) showed that nearly 94% of all respondents from both groups were in favour of more MPAs being established in New Zealand, but only 57% percent of respondents from towns near the potential MPA supported the proposal for 'their areas' (Wolfenden *et al* 1994).

Most examples of user group conflict have involved MPAs for fisheries protection and enhancement. Apart from the concern about foregone income from the establishment of MPAs, some fishers and other user groups oppose MPAs on the grounds that benefits cannot often be seen in the short term. Although some MPAs have demonstrated benefits over short time frames (eg one to five years, in Tasmania – see references by Edgar and Barrett), closures of longer terms (years to decades) are sometimes required for fish stocks to be replenished, diversity to improve, or degraded habitat to be restored. There is no standard answer to this problem of 'delayed benefits', because the time taken for a MPA to be successful will differ considerably, according to its size, location, purpose(s), contents, and management, as well as the environmental variables and level of human activity surrounding and affecting the MPA. In other cases, MPAs will be successful in achieving the goals of protecting marine ecosystems/habitats and biodiversity, for example, but these benefits may not be obvious to the public, either through failure of proposed monitoring plans, or no monitoring provision at all, often for economic reasons.

The 'NIMBY' ('not in my back yard') argument about MPAs has become a major international issue, particularly for proposed MPAs involving fishing prohibition. The polarisation of views between MPA advocates and MPA opponents stems from the MPA advocates' view that:

- the sea and its resources are common property, and that fish harvesting is a privilege, not a right, (and thus no compensation is necessary)
- current fishing harvests are not sustainable, with declines and commercial collapses of stocks becoming more widespread in many countries
- non-extractive users of the marine environment have a right to demand that fish stocks are not depleted by fishing
- MPAs have been demonstrated to assist fisheries stock recovery and enhancement, and fisheries sustainability
- MPAs are required to ensure that marine ecosystems and their biota persist for perpetuity and long-term social benefit, without being damaged and depleted for short-term economic gain

versus the anti-MPA fishing lobby's contention that:

- traditional access equals ownership, and thus compensation is warranted
- fishing is a right, not a privilege
- fishing is a traditional livelihood, often with inter-generational significance
- recreational fishing is a major activity involving millions of people in many countries
- fishing is a major contributor to national, State and local economies.

There is obviously no clear solution to these opposing points of view, because both can be seen as valid under the system of constitutions, legislation, government policy, and management structures that operate in many countries. It is notable, however, that in most countries, marine water and its contents are not legally 'owned' in the sense that terrestrial land is leased, bought and sold. In North America, for example, marine waters have long been held in public trust, and cannot be conveyed to private parties (Wilder 1999, unpublished comment to Californian Marine Protected Areas Network).

Despite the abundant evidence (cited in this report) that MPAs have been successful in achieving a number of marine conservation and management aims that benefit society as well as the marine environment, resistance is still high within some sectors. To date, it has been easier to convince user groups of the need for MPAs when alternative marine management measures have been known to fail, and when it becomes obvious that delaying the implementation of MPAs could have long-term environmental and socio-economic impacts. For example, fishers are more likely to accept the need for fisheries MPAs when fish stocks decline, and their catch levels are reduced. This situation occurred in the 1980s in Tasmania, where MPAs were only accepted as a management option by the fishing industry when fish stock declines became serious (Kriwoken 1993). The same situation is presently occurring in some parts of North America, where MPAs are considered by many to be one of the last effective management options available for curbing the serial depletion of vulnerable stocks (see Section 3.4.1). Similarly, divers and other recreational users are more likely to accept the need for increased protection of marine ecosystems using MPAs, once they see the visible impacts of overuse on marine benthic habitat quality.

Section 8.3 discusses ways in which conflict within and between user groups, and between MPA proponents and opponents, have been overcome in some regions (eg parts of North America, Philippines, New Zealand and Australia). Some of the methods that have been used to overcome user group conflicts have included:

- a clear definition and public explanation of the goals of the MPA, and the management arrangements required to support those goals
- detailed public education programs, through the use of public meetings, talks to industry and community groups, mixed media (such as radio, newspapers, television, computer web sites) and surveys
- consultation between all MPA ‘stakeholders’, through the formation of committees and working groups, prior to MPA establishment
- a study/survey of the socio-economic and environmental values of proposed MPA areas, (with valuation in terms which are as equal as possible, when comparing between interest groups)
- formation of a MPA management plan, prior to MPA establishment, with involvement of all stakeholders in the production of the plan, and a public comment period for the plan
- zoning of areas and activities in multiple-use MPAs, so that most uses and activities can potentially be accommodated in some way, whilst ensuring that those uses and activities are compatible with the MPA goals
- joint management and monitoring arrangements, often involving user groups such as fishers and divers.

Ideally, if the rationale for MPAs can be widely promoted through consultation and education, then the long-term benefits of MPAs can be made obvious to opposing groups. In examples that involve the creation of large, multiple-use MPAs, zoning uses and activities can sometimes solve conflicting interests, so that a variety of user groups can be accommodated within the MPA, without compromising its goals. Allaying user group concerns about foregone opportunity in ‘no-take’ MPAs is more difficult, but various measures are becoming successful in some countries (see Sections 7.1, 7.2, and 8.3).

Demonstrating the benefits of MPAs to specific interest groups can also reduce opposition to MPAs. This has been a somewhat successful strategy in New Zealand and Tasmania, where many fishers, for example, are now more supportive of MPAs because their fisheries have benefited from the establishment of MPAs. Demonstrating MPA benefits is a ‘catch-22’ situation; however, MPAs need to be designated before the benefits can be seen, but some user groups will not accept the designation of MPAs until they can see the benefits.

Public support for MPAs has been often been high in areas where the public has ‘directly experienced’ MPA benefits (such as increased fish stocks, improved habitat quality, recreation / tourism / education opportunities, etc). User group conflicts were overcome in this way in parts of New Zealand, where non-extractive, non-damaging uses of MPAs are encouraged. In New Zealand, both non-extractive users (divers / snorkellers, scientists, educators, etc) and fishers benefited from the establishment of the Leigh Reserve (see references by Ballantine). However, it has been more difficult in many other countries for MPAs to gain the acceptance of both groups, especially prior to establishment. Even in New Zealand, which has a relatively long history of successful (although spatially limited) MPA establishment, it is likely that the current proposal for a representative system of MPAs around the coastline will result in much conflict between user groups, particularly those involved with property rights-based commercial fishing.

Socio-political compromises to scientifically (biogeographically, ecologically) ‘ideal’ MPA designs are becoming more common. Mutually agreeable compromises about the size and placement of MPA have occurred in a number of countries that have engaged the cooperation and support of fishing communities, such as the South Pacific, and, recently, part of the Florida Keys in USA (Molyneaux 1999). However, the situation is very difficult in countries where fishing is a major income generator and export earner. Methods that are being considered in some countries to deal with the issue of fisheries displacement are discussed below (Section 7.1), and chief amongst them is financial compensation.

With the changes in user group activities that follow MPA designation, it is important to review the management of activities outside of the MPA. For example, it is considered desirable to reorganise resource allocation outside the MPA, to relocate displaced fishers where possible, and also prevent increased levels of unplanned overfishing in the adjacent areas—ie ‘effort transfer’).

7.1 INTERNATIONAL EXAMPLES OF USER GROUP CONFLICTS AND SOLUTIONS

Recent experience of MPA workers in North America has shown that it is difficult to establish MPAs that are large enough to demonstrate benefits to fisheries due to (i) resistance to new fisheries management approaches; and (ii) intense political opposition of local special interest groups (Roberts *et al* 1995; Bohnsack and Ault 1996; Bohnsack 1997, 1998).

Recently there has been heated debate in North America about compensation rights for fishers who are (or have potential to be) displaced from MPAs. An attempt to solve the dispute has been made by America’s National Research Council, which is currently engaged in a two-year national study of MPAs for fisheries. The council has a committee of 21 international experts, who are conducting a series of public forums with scientists and fishers all over North America (Molyneaux 1999). As part of the process, the National Council is reviewing data from closed areas (temporary MPAs). Although a ‘fragile compromise’ on the location and size of a ‘no-take’ MPA for fisheries replenishment has been made in the Gulf of Mexico, fishers in other parts of North America are still very antagonistic towards MPAs. Suggestions to set aside 20% of fished areas in some parts of North America as ‘no-take’ zones have been met with much resistance (Debenham 1999). Alternative measures to fisheries MPAs have been suggested for areas where fisheries management regulations have not been strong in the past. Some of these alternatives include the introduction of quotas for long-lived, site-attached, over-fished species such as groupers, and prohibitions on bottom trawling gear (particularly in rocky habitats), which has demonstrably flattened out bottom topography, decreased habitat complexity, altered species composition, and reduced biodiversity (Engel and Kvikek 1998).

Financial compensation is also being considered in North America, for displaced fishing activity that would occur following MPA designation, coupled with job replacement and training schemes. Other measures have also been suggested, such as phasing in MPAs over time, to allow the accrued fisheries benefits (see Chapter 3) to eventually compensate for the loss in fishing area (Sladek-Nowlis and Roberts 1997, cited by Murray *et al* 1999).

The opposite approach to compensation is also being considered in some North American MPAs, whereby a group that directly benefits from MPA use, at the 'expense'[9] of other uses/values, is required to pay for the privilege. Lease programs for aquaculturists are being suggested, with revenue used to manage and monitor the marine sanctuaries (multiple-use MPAs) in which the aquaculturists operate. This is seen as one way of compensating the public for essentially 'private use' of seabed and water that are considered to be a public resource (Barr 1997).

Increased multi-sector participation in MPA planning and management is becoming an important way of allaying user group conflicts in some areas of North America where new MPAs are being proposed (Ehler and Basta 1993). Multi-sector participation includes fisheries managers, resource user groups, other managers of economic development, local and regional coastal land use and water quality managers, conservation representatives, scientists, indigenous users, recreation/tourism representatives and other 'stakeholder' groups.

The approach to conflict resolution in Central American MPAs (such as Belize), in which tourism activities have impacted on the fishing sector, is a 'participatory planning method'. Some parts of the MPAs are closed to fishing but available for recreational activities, and other parts are open for small-scale, non-damaging forms of fishing as well as tourism. All stakeholders are involved with the development of an equitable zoning system, under the guidance of three government resource agencies, and this system is considered to be very effective in resolving user group conflicts, since it has the approval of all stakeholders (George and Nicholls 1994; Wells 1998).

More recently, a similar zoning approach was adopted for the Galapagos Islands, (Bensted-Smith and Bustamante, 2000, unpublished comment to Californian Marine Protected Areas Network), following three years of negotiations between management boards and authorities, and national and local stakeholders. The zoning scheme provides for 20% of the coastline to be zoned as 'no-take' protected areas within each biogeographical region, including buffer zones and designated sites around core protected areas, in which recreation/tourism activities will still be permitted. Compensation, redistribution of fishing effort, and alternative employment schemes for fishers are being organised.

Conflicts between user groups in MPAs in both Kenya and the Philippines are also solved by a system of zoning, in which recreational activities and fishing are spatially separated (Wells 1998).

With sufficient public participation in the designation process, it is possible for MPAs to be acceptable and workable in social terms, without compromising their ecological integrity. Fiske (1992) provided an example in which the very public designation process for a marine

[9] 'Expense' in this case would include the often unquantified costs of 'externalities' such as environment damage.

reserve in American Samoa eventually pleased all parties, including the commercial fishers who were initially the strongest opponents of the reserve.

7.2 NATIONAL EXAMPLES OF USER GROUP CONFLICTS AND SOLUTIONS

As in the international cases, different sectors of society in Australia have opposing views regarding the purposes, locations, and effectiveness of MPAs. Many fishers are concerned that a national representative system of MPAs will restrict their access to resources. Conservationists are concerned that multiple-use MPAs will not provide adequate protection for marine ecosystems and biota if extractive activities such as trawling and mining are permitted[10]. Indigenous representatives are concerned that the bioregional focus of the NRSMPA ignores human interests in (and use of) marine environments, and makes no provision for clan estates, which extend into the sea, and have traditionally been defined and used by Aborigines to control access to, and use of, marine resources (Roberts and Tanna 1999).

The Australian situation regarding user group conflicts is beginning to resemble the fisheries compensation issue that is currently raging in North America, with the parallel but seemingly opposing developments in Australia throughout the 1990s of (i) the National Representative System of MPAs; (ii) an industry-driven move towards fishing property rights (eg International Fish Rights Conference held in Western Australia in 1999); and (iii) increased public knowledge of the need to consider in Australian marine law and management, the existence of indigenous sea rights (eg 1999 National Indigenous Sea Rights conference), such as clan estates.

Although this report has attempted to show that a national representative system of MPAs can be compatible with a variety of coastal marine activities, a more thorough and site-specific assessment in the context of user group conflicts is beyond the scope of this work.

There are isolated examples of Australian MPAs being supported by fishers after their establishment, particularly if those fishers directly benefit from the effects of the MPA on fished species. For example, Brenthouse (1990), reported that trawl fishers who initially considered a ‘no-take’ MPA in the northern Great Barrier Reef to be an imposition on their livelihood, changed their opinion two years after the closed area was established. Trawl catches near the closed area had increased, and a subsequent survey of the trawl fishers showed 80% compliance with the regulations of the MPA. A number of fishers even expressed the wish for more permanent closures.

In Queensland’s multiple-use GBR Marine Park, creation of a zoning system, as well as dissemination of MPA management information and long-term promotion and education about MPA values and impacts have been crucial methods of reducing user group conflicts. GBRMPA’s commercial fisheries consultation program, and more recent co-operative research and management ventures with the tourism industry, also assist in reducing user group conflicts, by directly involving stakeholders in management issues for the GBR. However, the interests of all groups are apparently not equally accommodated in the GBR. The formulation in 1992 of a 25-year strategic plan for the Great Barrier Reef offered some scope for recognising Aboriginal and Torres Strait Islander rights to marine resources and

[10] In Western Australia, for example, the current system of MPAs has been criticised for providing inadequate protection of species and habitats, particularly endangered species, which may be affected by fishing operations, seismic surveys, and oil and gas drilling, all of which are permitted in some large, multiple-use MPAs (Preen 1998).

participation in management, but the plan has been criticised by some indigenous groups as not adequately considering traditional aboriginal interests in tenure and management of some areas of the GBR.

New Zealand owes much of its success in reducing user group conflicts in existing MPAs to long-term efforts in public promotion of the MPAs. The MPA promotion strategy includes zone maps, brochures, other written material, information sessions, and even videos, to keep MPA user-groups informed about the MPA, its goals, operation and management arrangements. However, it remains to be seen whether the early success of MPAs in small parts of New Zealand's coastal marine areas can be expanded, with the proposal for a national representative system of MPAs in a country that has native and non-native fishing property rights established in law.

8 MPA PLANNING, LEGISLATION, MANAGEMENT AND MONITORING

8.1 MPA PLANNING: THE LONG ROAD TO PROCLAMATION

Years of planning and consultation usually occur before MPAs are finally designated, even if the proposed MPA is a comparatively small area. The Leigh Reserve in New Zealand is a classic case. When the MPA was first proposed in 1965, there was no international or national legislation that existed to permit the sea to be 'reserved' in any way, and the proposal was strongly discouraged by both the New Zealand government and commercial and recreational fishing groups. It took 11 years of campaigning, educating, promoting, consulting, and collecting evidence about MPA benefits, before the reserve was established (Ballantine 1979). Several years after the MPA was established, it was hailed as a resounding success for:

- conservation of local biodiversity
- habitat protection
- replenishment of depleted fish and invertebrate populations
- recreation / tourism
- marine biological and ecological research, and marine education
- fisheries management (eg local rock lobster fishers increased catches near the reserve, were convinced of its value, and became vigilant about poaching in the reserve).

The success of the Leigh Reserve, coupled with a long-term commitment to public education and communication by New Zealand's key MPA proponent (Bill Ballantine), has resulted in the creation of 16 more MPAs in New Zealand since the 1980s. New Zealand now has a plan for a national representative system of no-take reserves, covering every ecosystem type around the coast.

Many recent MPA proclamations around the world still suffer a comparatively large lag time between proposal and declaration. For example, The Western Sambos Ecological Reserve, proclaimed in 1997, is considered to be the first 'no-take' MPA in US waters to be declared for fish replenishment. It is hoped that Western Sambos will 'export' fish, larvae and eggs, to restock the surrounding area that is heavily fished, polluted and subject to heavy tourism. The Western Sambos MPA took six years to be established, following extensive consultation between scientists, fishers, divers, aquarium fish collectors, local business leaders, and county, State and Federal officials (Schmidt 1997). Despite abundant evidence that 'no-take' reserves can help to sustain fisheries and replenish declining stocks, fishing groups in the Florida region are, apparently, skeptical of the declaration of Western Sambos. Such a response is expected, in the early stages of MPA declaration, given the time it takes for a MPA to prove its value for fisheries management.

Current MPA planning by the IUCN provides an international example of the long-term planning associated with establishing and managing a system of MPAs. The IUCN has recently been involved with project proposals for establishing MPAs in countries that currently do not adequate protect and manage marine ecosystems and resources, including Tanzania, Vietnam and Samoa. The proposals have the strategic, collaborative approach that is considered necessary in many countries for successful MPA establishment. Such proposals include:

- an assessment of the biophysical information for each proposed MPA
- identification of potential local-level strategies to address the major threats to MPAs
- legislation, regulations and zoning provisions

- institutional arrangements and mechanisms involving appropriate government agencies
- mechanisms to achieve ongoing and effective public involvement
- education, extension, surveillance and enforcement provisions
- research targets
- management and evaluation plans
- economic assessment of the proposals.

It is likely that achieving these aims on the planning 'wish-list' will take several years, at the very least.

The situation in Australia is similar. One of many examples is Victoria's first MPA (the Harold Holt Marine Reserve), which took seven years to establish, following acrimonious debate that eventually provided for commercial and recreational fishing activities (netting, angling, spearfishing, etc) in most of an area that was originally proposed as a 'no-take' reserve (Malcolm 1993). In New South Wales, it took even longer (20 years) to establish the first real MPA following the initial proposal.

There is no 'quick fix' available to overcome the lengthy process from MPA planning to MPA proclamation, which often involves protracted planning phases, legislative confusion, inter-agency conflicts, and (in many cases) acrimonious public debates. Methods of streamlining the process are many and varied, and are discussed in the following section.

8.2 MPA LEGISLATION

The declaration of MPAs under more than one Act, and MPA management by more than one agency, has previously created public confusion, management and monitoring impediments, and even legal disputes, in most Australian States. The legislative problems associated with MPA establishment should not be underestimated. If legislative and management arrangements are not made clear prior to MPA declaration, legal problems can ensue from (among other reasons) an unwarranted lack of communication between the government and the public, resulting in conflict. For example, during the 1980s, the Victorian government wished to exclude fishing from inside the Wilson's Promontory Marine Reserve, but was unable to do so legally because the MPA had been declared under a different Act. This resulted in a Supreme Court case between professional abalone fishers and the State government (Malcolm 1993).

It may be preferable to have MPAs declared under one piece of legislation, preferably one that is devised specifically for marine protected areas. There are currently Marine Park Acts in Queensland (both Commonwealth and State Acts) and New South Wales. New Zealand was the first country to create an MPA Act *(Marine Reserves Act 1971*) and, although other countries have been slow to follow, marine park Acts are now starting to be developed in other parts of the world to assist MPA planning (eg in East Africa: SARCDC 1997).

It may also be preferable that MPAs be managed by one agency, or by coordinated management arrangements between the relevant agencies. This would help States to overcome the current problem of differing legislative and management provisions under each Act, for suites of MPAs within one State that are declared under different Acts. The strength of legislation for designating MPAs currently differs between Acts: MPAs can be revoked under the fisheries Acts in New South Wales and South Australia but, if declared under any of the national parks and wildlife Acts, MPAs can be revoked only under parliamentary decision (McNeill 1994).

In South Australia for example, the two agencies that manage MPAs (Primary Industries and Resources, and the Department of the Environment and Heritage) do not have the same legal provisions to monitor and control activities outside the boundaries of the MPAs which are declared under their respective Acts (*Fisheries Act, 1982* and *National Parks and Wildlife Act, 1972-1982*). Implementing a representative system of MPAs would require some standardisation of management practices between agencies.

8.3 MPA PLANNING AND MANAGEMENT

During the past decade, much has been written about MPA planning and management, both nationally and internationally. The government reports cited in Chapters 3 and 4 provide detailed discussions of effective planning and management procedures for MPA establishment, and such detail will not be recounted here. During the 1990s, the formulation of responsive, adaptable management plans is becoming prevalent for MPAs all over the world. They are especially useful for large MPAs that contain a variety of ecosystems/habitat types, in which multiple-uses are permitted. Management plans are best formulated prior to the establishment of the MPA, but must be responsive and adaptable after MPA establishment. For example, when the Tanzanian Government became aware that conflicting (and some very damaging) practices were affecting the variety of economically-valuable ecosystems in the Mafia Island region of East Africa, they formulated a management plan for a marine park around the island, *prior* to its establishment under a new marine parks and reserves Act (SARCDC 1997).

Socio-economic surveys of MPA users are also being used in MPA planning, with valuable results for planners and managers. During the early 1990s, a survey was conducted in New Zealand of 800 potential MPA users from socio-economically comparable 'high-impact/opinionated' groups (living near two proposed MPAs) and 'low-impact/neutral' groups (away from the proposed MPAs) (Wolfenden *et al* 1994). Nearly 94% of all respondents from both groups were in favour of more MPAs being established in New Zealand, but only 57% of respondents from towns near the potential MPA supported the proposal for 'their areas'. More than 80% of respondents stated that they would like to have received more information about the MPA proposal from the Department of Environment. Opposers of the MPA were a more active group compared to the supporters, and worked harder to lobby the government to quash the MPA proposal, even though both supporters and opposers originally belonged to the same community action group[11]. More detailed analysis and questioning of respondents led the surveyors to conclude that a vocal minority of opposers, rather than a 'silent majority' of supporters is more likely to engage the attentions of MPA planners. It was therefore recommended that MPA planners should also seek out the opinions of the less obvious 'public majority' during MPA planning.

Surveys have also been used for MPA planning in the Caribbean (Mascia 1999), where site managers are employed for each MPA. The survey of 42 MPA managers highlighted gaps in biological and socio-economic information prior to MPA establishment, public participation in the planning process, and compliance with MPA regulations. Such information is useful for management of current MPAs as well as for planning new MPAs.

Increased participation in the MPA planning and management process is now seen as critical to MPA success (Ehler and Basta 1993). Participants should include fisheries managers,

[11] At the first community meeting to discuss the proposed MPA, the supporters felt intimidated by the eloquence of the opposers, and left the group, whereupon the opposers established leadership and developed a strong financial base with which to lobby against the MPA (Wolfenden *et al* 1994).

resource user groups, other managers of economic development, local and regional coastal land use and water quality managers, conservation representatives, scientists, indigenous users, recreation/tourism representatives and other ‘stakeholder’ groups. Multi-sector participation is particularly important in the management of large, multiple-use MPAs. This is the new approach being used in North America, Central America and Britain, where reliance on statutory power and government-led consultation/interpretation is considered to be much less effective than meaningful stakeholder participation and joint planning arrangements (Jones 1999).

In the Bahamas, for example, the Department of Fisheries began working in 2000 with a national environmental education foundation, local government representatives, the National Trust, and a number of scientists, to establish a network of no-take MPAs throughout the Bahamas. Through this process, the government has recently approved five no-take marine protected areas in that country, as part of a network approach to MPA establishment (see Section 6.1.7). As is currently the case in many parts of the world in which new MPAs are being established, full community participation is purportedly occurring in the Bahamas, to decide the boundaries for the MPAs, the governing legal framework, and the details of assessment, management and monitoring plans.

Involving all stakeholders during the MPA planning process prevents situations like that described in Section 8.2, in which critical management decisions concerning MPAs fall into the hands of the legal system, well after conflict has arisen.

One apparently successful Australian example of multi-user, multi-agency participation in the MPA planning process is the recently declared Macquarie Island Marine Park. The national fisheries management authority (AFMA), State fisheries managers, scientists, fishers, conservationists, and other stakeholders and interest groups were involved with the development proposal for the MPA (Dahl-Tacconi (1999, unpublished comment to Californian Marine Protected Areas Network). These groups also jointly assessed the conservation values of the waters around Macquarie Island, and assisted with the design of zonings of the park.

Public participation is desirable in both the formulation of MPA management plans, and in tactical, ‘on-site’ management activities. This is likely to reduce user group conflicts, and encourage stewardship and active public participation in management and monitoring of the MPA. State and local governments must cooperate closely, engaging public participation wherever possible, for effective MPA management. For example, the sanctuary management plans for North America’s large, multiple-use MPAs provide for continuous management involving Federal, State and local governments. In North America, management strategies for MPAs are evaluated for their costs, likely consequences, and ease of implementation, so that priorities can be established (Ehler and Basta 1993).

In Australia, although State agencies are (and should continue to be) primarily responsible for MPA management, local government involvement in the management process is also desirable, since councils may be more aware of (and more directly responsive to) the local-scale management issues that can arise for MPAs near their coastal electorate.

Management plans for MPAs must be responsive to changing conditions in the MPA over time. For example, although five-yearly zoning reviews have been largely successful for the Great Barrier Reef Marine Park, they have been criticised as being unresponsive to rapid changes in particular areas (such as sporadic occurrences of local fish aggregations), that require swift response (for example, in fishing effort controls) (Shorthouse 1991).

MPA promotion and education is a crucial part of the management process. The GBRMP and the marine reserves of New Zealand owe part of their success to long-term public promotion efforts, in the form of zone maps, brochures, other written material, information sessions, and even videos, to keep MPA user-groups informed about the MPA, its goals, operation and management arrangements. Similarly in Queensland, the GBRMP Authority started a commercial fisheries consultation program during the 1990s, using a commercial fishing representative in the consultation process, and also provides much management information to tourists/recreational users of the GBR, using guides and educational materials. Such strategies are considered important to the successful management of the GBRMP.

MPA planners in New Zealand have been engaged in a comprehensive public education program to promote the value of a representative system of MPAs (called marine reserves in that country). The public consultation and education strategy includes public meetings to discuss MPA proposals; television programs; radio talk-back sessions; magazine articles; computer web sites; support for conservation groups; diver surveys; and MPA monitoring programs (Ballantine 1999). The favoured approach in New Zealand, which appears to have met with success, is for government agencies to persistently promote the benefits of MPAs, and to actively engage the public in the decisions about MPA locations and sizes, rather than prescriptive tactics that involve designing the MPA and 'selling it' to the public afterwards.

The promotion strategy must match the goals of the MPA. For example, if a country tries to establish a biogeographically and ecologically representative system of MPAs, there is an obvious need to promote the critical roles of *marine ecosystems*, and how protection of these ecosystems will benefit society. With the exception of coral reefs, much of the general public does not often consider the importance of marine ecosystems as a whole. This contrasts with the great public interest in whales, dolphins, bizarre sea animals and beaches.

However, it is unreasonable to assume that the public will accept a MPA system planned exclusively around ecological criteria. Likewise, it is reasonable to assume that most people would place more value on MPAs if the establishment were couched in terms of social benefits. This approach has been successful in Tasmania, where community awareness and public education programs were used prior to the establishment of the four MPAs at Maria Island, Governor's Island and D'Encastreaux Channel (Kriwoken 1993). Public meetings and discussions with local councils and interest groups were held. Interestingly, commercial and recreational fishers were initially opposed to MPAs in Tasmania, but supported the concept of MPAs after declines in many of Tasmania's traditional fisheries were noted during the 1980s (Kriwoken 1993). It became evident to the fishing sector that MPAs were one way in which fish could be naturally 'propagated' to replenish depleted fisheries.

There is little point in establishing a system of MPAs if they will not be managed in some effective way, and will thus be subject to numerous detrimental influences, or require regular, expensive and socially undesired enforcement procedures. Suggested approaches to maximise MPA acceptance and compliance, and to minimise the need for enforcement, include the following:

- the rationale for MPAs should be well publicised, through public education/liaison programs
- the potential locations of MPAs should be well promoted, and thoroughly discussed with all 'stakeholders'
- the 'boundaries' of the MPA should be simple enough for the general public to recognise them (eg based upon geographical markers such as headlands and bays, or on specific reef patches)

- where possible, a management and monitoring program should be established, preferably with community participation in the monitoring, and involving a MPA education component.

The following two sections provide examples of successful management strategies for MPAs.

8.3.1 International examples of MPA management

Multi-group participation in MPA management is the norm for North America's large marine sanctuaries that cover thousands of square kilometres. Groups involved in the formulation of management plans include State-based natural resource agencies, local governments, the United States Environmental Protection Agency, the National Parks Service, water management authorities, and the US Fish and Wildlife Service, among others. This working group has a 'Citizens' Advisory Council' that provides the critical link to local user groups and the general public, and the council is an important part of the MPA planning and management process. The sanctuary plans emphasise the need for ongoing data collection from public, scientists and resource managers, and 350 individuals are involved on a long-term basis with this process to assist MPA management (Ehler and Basta 1993).

It has been recently stated (de Vogelaere and Green 1998) that MPAs in North America for which scientific advisory committees have been appointed are more successful in producing and implementing MPA management plans, compared with MPAs for which such committees do not exist. The scientific advisory committees were also considered to have a broader range of resource management alternatives, larger research budgets, and engage in a wide variety of management-oriented research activities in the MPAs. There has consequently been a call to expand the role of scientific advisory committees to the management of other MPAs.

A new approach to MPA management directly involves fishers in data collection to aid management decisions. An unprecedented act of cooperation in 1999 has resulted in a plan for MPA scientists and commercial fishers to work together in the MPA around the Channel Islands in North America. The fishers will be trained and paid to gather data during their 'off-seasons', including information about where fish are located, how they behave, and habitat conditions in the MPA. If successful, government agency funds will be supplied for fisher-based monitoring in 11 other MPAs in North America (Polakovic 1999).

Ideally, major user groups would be involved directly with the management of MPAs in all countries. This approach is successful in the Philippines, where local fishers, for example, supported the MPAs in principle, because they have seen the benefits of increased catches several years after MPA establishment (ICLARM 1997). However, given the differences between the artisinal fishing of Philippine islands compared with commercial business nature of many fisheries in other nations, the goal to engender propriety and close stewardship of small MPAs will be challenging.

Other data collection needs for MPA management at local and regional scales include the mapping of marine biogeographic regions and resources. Geographical information system (GIS) mapping of the marine biological resources in the Gulf of Fallorones National Marine Sanctuary (West Coast of USA) has recently been undertaken to assist MPA managers (Ford and Bonnell 1997). The Channel Islands National Marine Sanctuary also benefits from a GIS-based management strategy that allows staff to map physical and cultural features of the sanctuary, which includes aerial survey data for marine mammal sightings and fishing vessel activity, and historic shipwreck monitoring (NOAA 1997). Several State agencies and one university contribute and exchange data for the GIS, and the regularly updated maps are tailored to the needs of each agency, and shared among all resource- and area-management groups. Some of the research and monitoring programs that assist managers of the Channel Islands Marine Sanctuary are listed in Section 3.4.5.

South Africa's approach to MPA management has recently involved an evaluation system which compares the criteria and objectives for each MPA (the 'COMPARE' program), including biodiversity protection, fisheries management and human use and activity (Hockney and Branch 1997). This scoring system is used to (i) assess the effectiveness of current MPAs, in relation to each other and also according to the objectives of the national MPA system; (ii) evaluate the effects of changes in either legislation of management of MPAs; (iii) identify issues in MPA management that require action; (iv) assist in the development of management plans; and (v) identify gaps in the existing MPA network, so that South Africa can develop a rationally planned and defensible national system of MPAs.

8.3.2 National examples of MPA management

Cooperative MPA management programs between MPA interest groups and government authorities are now developing in Australia. Despite accumulative damage to the Great Barrier Reef from land-based nutrient pollution and siltation (among other impacts), the GBR Marine Park has long been hailed both nationally and internationally as an example of successful integrated management of marine ecosystems, resources, activities and user groups. The success to date has been largely based upon the zoning system, which was developed with extensive public consultation during the 1970s and 1980s. Commercial and recreational fishers, boaters, conservationists, scientists, tour operators, developers, and other interest groups were involved with the production of zoning plans for each section of the park. Thorough resource inventories and analyses of uses and 'stakeholder' activities (including economic valuations of uses) were prepared prior to zoning each section, and the zoning plans are subject to review every five years (Rigney 1990). The Great Barier Reef Marine Park Authority has also developed consultative committees to improve communication between 'stakeholders'. Membership of the committees includes tour operators, recreational and sports fishing groups, commercial fishing representatives, recreational dive groups, representatives for indigenous users, conservation groups, local government, port authorities, and State government agencies. As an example of cooperative management in the GBR, reef tour operators are now working with the marine park authority, the Department of the Environment, and the Cooperative Research Centre, to develop codes of practice, monitoring programs, and other management arrangements to minimise impacts, and ensure the long term sustainability of major activities (Green 1996).

Management plans are also prepared for the Ningaloo Marine Park in Western Australia (CALM 1989). These plans have a 10-year review date, and cover the legislative and administrative arrangements; values of the park (conservation, recreation, commercial, historical/cultural); zoning of the park; commercial and recreational activities; land management adjacent to the park; development; potential impacts; research; and education. Maps of the conservation and management zones assist the multiple users of this MPA.

Management plans have been (or are currently being) developed for other large, multiple-use MPAs in Australia, and are considered essential to the long-term successful operation of these areas, given the large spatial scale, multiple uses, multiple potential impacts, and numerous user-group conflicts associated with large MPAs.

8.3.3 South Australian examples of MPA management

MPAs in South Australia are ostensibly managed by the State agencies under whose Acts the MPAs were declared, namely the Primary Industries and Resources South Australia (formerly the Department of Fisheries) and the Department of the Environment and Heritage. Section

8.4.3 below discusses monitoring programs that were carried out in three of South Australia's MPA during the 1980s, to assist management.

A detailed management plan has been prepared for the Great Australian Bight Marine Park (Government of South Australia 1998). There are no management plans for the other existing MPAs, apart from the aquatic extensions of terrestrial national parks and conservation parks (which currently offer no subtidal protection, and are thus not considered here as MPAs). A management framework has been provided for potential new MPAs in South Australia (Edyvane 1999).

8.4 MONITORING MPA EFFECTIVENESS AND IMPACTS

Monitoring in the broad sense refers to repeated measurements taken at the same site, on the same subjects, over a specified period of time (Noble and Norton 1991, cited by Phillips *et al* 1993). Monitoring in MPAs is conducted to investigate patterns of activity by user groups, to monitor impacts of activities, and to assess the effectiveness of the management aims, such as maintenance of habitat quality, species replenishment, and biodiversity conservation. The results from monitoring are used to assess the effectiveness of the MPA, and to predict the likely impacts of changes to the management and use of that MPA. Monitoring results can therefore have significant political, socio-economic, scientific and other repercussions.

Ideals have been set in Australia for MPA monitoring programs (Philips *et al* 1993; Fairweather and McNeill 1993), espousing the scientific principles of pilot studies, control sites, 'before / after' comparisons for MPA and non-MPA areas; and site replication in the experimental design.

Much monitoring of MPAs has been questioned and / or criticised for:

- not being adequately replicated (ie not including enough MPAs and non-MPA areas in the comparisons)
- being short term
- not comparing statistically significant numbers of MPAs versus non-MPAs
- being carried out *a posteriori* (ie only after the MPA was established) and not having any *a priori* data for adequate experimental comparison (ie most monitoring studies do not adhere to BACI (before-after-control-impact) experimental protocol, and are thus 'pseudo-replicated' in space and time)
- not considering the potential for experimental error due to differences over space and time in the skill, accuracy or technique used by various surveyors
- ignoring sources of sampling error, such as 'diver-positive' behaviour of fish in MPAs (ie less cryptic and evasive than their fished counterparts outside the MPA)
- ignoring small-scale variability over space and time in abundance and distribution of plants and in the MPAs, which may invalidate short-term or unreplicated survey results
- not considering, in degraded, exploited systems, the 'natural state' with which to compare changes due to MPAs
- discounting other factors that might be responsible for effects observed in MPAs (eg Roberts and Polunin 1992 stated that some of the differences that they observed in a comparative study were probably due to habitat and oceanographic differences between MPA and non-MPA areas, rather than the effects of protection against fishing).

(Stewart-Oaten *et al* 1986, cited by Fairweather and McNeill, 1993; Ballantine 1988; Cole *et al* 1990; Fairweather and McNeill 1993, Phillips *et al* 1993.)

These limitations can lead to equivocal or inconclusive results, ie scientifically flawed studies may not be able to detect differences which are actually due to the MPA. To overcome these limitations, workers in Australia have recommended statistically valid and scientifically meritorious monitoring programs for MPAs (Phillips *et al* 1993; Fairweather and McNeill 1993). Carr and Reid (1993) also recommend more statistically robust analytical methods for determining the effectiveness of North American MPAs.

However, despite the inconclusiveness of some of the studies that demonstrate the benefits of MPAs, proponents could argue that:

- Even if the benefits cannot easily be demonstrated experimentally, it is unlikely that MPAs will have long-term adverse effects upon resource populations, fisheries or society—indeed the evidence attesting the value of MPAs clearly outweighs the disadvantages (as discussed in other sections of this report), and the number of examples clearly demonstrating the benefits of MPAs (particularly for fisheries management and enhancement) is growing rapidly.
- Often it is politically, feasibly, practically, or economically impossible to conduct the kind of exemplary, replicated monitoring studies advocated by the critics of MPAs. In many cases, spatially replicated MPAs of the same type, which are desirable for statistically valid comparisons, do not exist in an area. Further, where replication is possible, political and economic factors make it unlikely that several similar MPAs could be designated purely for experimental rigor.
- Even if specific benefits to fisheries, such as increased size or abundance of resources, cannot be demonstrated in the short or medium term, MPAs can be used to protect critical habitats from destructive influences such as benthic habitat damage and point-source pollution, and this indirectly benefits local and regional fisheries.

Regarding the monitoring of MPAs for biodiversity protection, there are no recorded examples of baseline data on biodiversity and population abundance estimates being gathered in proposed MPA areas prior to their establishment (McNeill 1994), apart from isolated examples of 'before-and-after' studies in fisheries MPAs (see Chapter 3). Such 'baseline' data are highly desirable, and would obviously assist MPA research, management and monitoring. However, compared with monitoring a MPA to assess its effectiveness in protecting a single species (eg for fisheries management), it is difficult to determine how effective MPAs are at protecting biodiversity. Reasons for this difficulty include the facts that: (i) studies of biodiversity in MPAs are very costly and time consuming; (ii) it is difficult to measure biodiversity, unless all taxa are known and can be assessed by those conducting the study; and (iii) traditional numerical measures of species diversity are often unreliable, and prone to spatial error (Phillips *et al* 1993).

In reality, limited government resources are rarely used for monitoring MPAs in most States of Australia. It is a long and costly process to establish MPAs, and differing funding priorities after MPA establishment usually makes regular monitoring a luxury, despite expressions of its scientific necessity. It will be especially difficult for jurisdictions to implement a performance assessment system for the role of MPAs in biodiversity conservation – despite the call for scientifically-valid surveys before and after MPA establishment, as well as long-term studies of specific protected sites, such monitoring strategies are unlikely to be given priority funding. Currently, the status of (and activities and impacts occurring in) most of Australia's MPAs are not monitored over time. There is no reason to believe that monitoring will occur in new MPAs which are established, unless there is clear directive, a mandatory requirement for regular reporting, and a dedicated source of funds and personnel.

The Commonwealth Government's *Strategic Plan of Action* for establishing the NRSMPA (ANZECC TFMPA 1999) has recommended the use of performance assessment criteria, and environmental assessments and audits on a regular basis, using environmental indicators (eg Saunders *et al* 1998; Ward *et al* 1998). It is proposed that each State develop a performance assessment system of MPA monitoring, consistent with a nationally-agreed reporting framework. Done and Reichelt (1998) have developed a novel approach to monitoring coastal marine ecosystems and the activities that affect them (such as fishing). This promising approach involves the construction of numerical indices to reflect the trophic status, conservation status, and species composition / biodiversity (amongst other indices) of coastal marine ecosystems. Where data are available (but often they are not, as discussed below), a set of performance standards such as this holds much promise for MPA monitoring.

One successful monitoring approach that is being used internationally (see section below) is the use of public volunteers in monitoring programs. This has apparently been successful in areas where it has been attempted, and is especially beneficial when data are not traditionally collected by resource management agencies of universities. The example sited in Section 8.3.1, on the use of fishers to assist MPA management, also extends to regular monitoring, since many fishers are obviously at sea more often than scientists, and can work with researchers to collect data for monitoring changes in the MPA. Such data might include temporal catch rates surrounding the MPA, and recording incidences of poaching and other damaging influences.

8.4.1 International examples of MPA impacts and monitoring

There are many examples of monitoring programs to assess effectiveness of MPAs, particularly MPAs for fisheries replenishment, as discussed in Chapter 3. It is especially useful to monitor MPAs from the time they are established, and this is the approach taken in some of the new MPAs in North America.

For example, baseline data on fish species composition, densities and length frequencies are being regularly recorded by divers within and adjacent to the Big Creek Ecological Reserve in California, an area where highly valuable rockfish (*Sebastes* spp.) have been depleted by sports and commercial fishing (Ven Tresca *et al* 1998). Local fishers regularly provide data on rockfish catches outside the MPA, to managers, who assess the effects of the MPA in local fishing, and vice versa (Pomeroy and Beck 1998). The entire Big Creek MPA has also been mapped acoustically, and the ongoing survey results are being used to compare the efficacy of the MPA compared with other rockfish MPAs, as well as stock levels in other areas that do not have MPAs.

Monitoring impacts in MPAs is becoming increasingly necessary, due to the cumulative impacts of MPA use. Significant impacts affecting multiple-use MPAs in tropical and sub-tropical regions include:

- physical damage to coral reefs from heavy use by SCUBA divers, and from boat anchors
- physical damage and chemical contamination of MPAs from land-based pollution sources
- destructive fishing methods such as bottom trawling (which occurs in North America's large national marine sanctuaries, for example)
- illegal use of destructive fish-harvesting methods such as dynamite-blasting (which has occurred in some south-east Asian MPAs)

- trampling and harvesting plants and animals in inter-tidal MPAs.

Monitoring of diving impacts is receiving increasing attention due to the widespread and cumulative nature of the impacts. In North America and Hawaii, divers from the Coral Reef Alliance regularly monitor impacts by divers and boat anchors.

Studies of SCUBA diving impacts in both tropical and temperate MPAs that are popular dive sites indicate that damage to attached benthic organisms is usually greater than impacts on fish populations. Such monitoring work has shown:

- stability of fish assemblages over a two-year study period at sites heavily utilised by divers in the Key Largo National Marine Sanctuary near Florida (Greene and Shenker 1993)
- significant coral damage in MPAs from boat grounding (Tilmat and Schmal 1981, cited by Davis and Tisdell 1995)
- ‘aesthetically striking’ differences between dive-damaged and undamaged areas in Egyptian resort dive areas (Hawkins and Roberts 1992, cited by Davis and Tisdell 1995), including damaged and broken coral colonies, lose fragments of live coral, and partially dead and abraded corals in heavily dived sites
- damage to *Acropora* corals (including breakage) by divers and anchoring in South African MPAs (Riegel and Riegel 1996)
- damage to the physical structure of corals from trampling, resulting in reduced size and distribution of coral cover, particularly hard corals (Hawkins and Roberts 1993, cited by Davis and Tisdell 1995)
- a 60-fold increase in diving activity, and a consequent significant decrease (-50%) in the densities of bryozoan colonies in a Spanish MPA, one year after the installation of a diving buoy to assist divers (Garrabou *et al* 1998).

Some authors have concluded that, because diving impacts are cumulative, a ‘critical threshold’ of diver usage exists, beyond which the amenity values and ecological values of MPAs are compromised (Phillips 1992; Dixon *et al* 1993; Davis and Tisdell 1995). These ‘critical levels’ have been quantified in some MPAs (eg 4000-6000 divers per year, per site, is considered the carrying capacity for the Bonaire Marine Park in the Caribbean, according to Dixon 1993). Diver education, rotation and spacing of permitted dive sites, banning photography tripods which encourage divers to stand on reefs, and monitoring of park users and activities, are recommended as ways of managing this problem (Dixon 1993). Zoning schemes for MPAs, in which vulnerable and more resilient areas are ranked, and consequently subject to different permissible levels of diver use, have also been suggested (Riegel and Riegel 1996).

The seriousness of threat to Central American and African MPAs that are heavily used by tourists, has recently been recognised by the IUCN, which has devised a monitoring program to assess the extent of reef damage by tourism, and to determine the effects upon biodiversity in those regions.

8.4.2 National examples of MPA impacts and monitoring

Zann (1995) summarised some of the major human-induced impacts in Australian MPAs, such as land-based nutrient pollution (eg GBR Marine Park); heavy metal contamination; threat of oil spills (particularly in tropical MPAs); direct and indirect effects of both legal and illegal fishing and harvesting, and over-use of MPAs by tourists. Other, relatively new threats include introduction of foreign pest species from ship ballast water, and potential impacts of aquaculture operations on nearby MPAs.

Adequate monitoring programs are usually only undertaken for large marine parks in Australia. For example, tour operators in the Great Barrier Reef Marine Park are working with the Reef Cooperative Research Centre to develop cost-efficient monitoring systems for reef impacts due to tourism use (Green 1996)[12].

The effects of user impacts are not monitored in most of Australia's smaller MPAs. However, some studies have been undertaken to assess the effects of trampling and collecting in inter-tidal MPAs, (Phillips *et al* 1993). Such studies have shown that macro-algal mats can be easily damaged by trampling, and may take more than a year to recover, and that significant reductions in sizes and densities of inter-tidal organisms (such as mollusks) have occurred.

Work has been undertaken to assess the impact of diving, such as the effects of kicking (with dive fins), trampling, holding, kneeling or standing on benthic organisms. Anchor damage from dive boat mooring can also cause significant local impacts. Phillips (1992, cited by Davis and Tisdell 1995) concluded that the 'carrying capacity' of the Julian Rocks Aquatic Reserve in New South Wales was being exceeded by tourist usage, with an average of 43 000 dives occurring per year in a small MPA of 80 hectares. Wilks (1993, cited by Davis and Tisdell 1995) estimated that 85 000 uncertified 'resort dives' and up to one million recreational dives per annum were made in Queensland's GBR Marine Park during the 1990s. A study by Harriot *et al* (1997) of four major dive-sites in Eastern Australian MPAs, showed that the average number of benthic contacts made by divers during a 30-minute dive, ranged between 35 and 121, according to site; most contact was made by dive fins; and a minority of divers caused coral breakage from this activity.

Education programs, controlled access, supervised dives conducted by MPA officials, and increased use of designated boat mooring sites can all assist in reducing the damage from diving. All of these methods are used in the GBR to minimise impacts. A national mooring system has recently been devised in Australia, to establish a greater number of boat moorings and thus assist in reducing the impacts of anchor damage.

8.4.3 South Australian examples of MPA impacts and monitoring

Barker Inlet is the site of a long-term fish monitoring program conducted by SARDI (formerly the Department of Fisheries) (Jones 1984; Jones *et al* 1996; Jackson and Jones 1999). Changes in the fish species composition and abundance in the estuary over more than a decade were considered to be influenced by thermal effluent from the Torrens Island Power station, and the spread of the seaweed *Ulva* (Jones *et al* 1996).

During the 1980s, fisheries enforcement officers conducted regular visits to Aldinga Reef Aquatic and Port Noarlunga Aquatic Reserves to monitor activities, and the area was also included in helicopter patrols (Ivanovici 1984). A monitoring survey was also conducted by the Department of Fisheries at the American River Aquatic Reserve, which showed that the distribution of organisms had not significantly changed since the 1940s when the ecology of the site was first studied (Ivanovici 1984). However, survey results from the 1980s are now outdated, given the increased use and impacts upon most MPAs since that time.

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[12] Although regular monitoring of activities occurs in some parts of the Great Barrier Reef Marine Park, it is ironic that coral assemblages in the waters around Australia's first unofficial MPA (Green Island, declared in 1938) have declined in quality, partly due to 'overuse' as a tourist destination, and consequent hydrocarbon pollution from tourist motor boats.

Apart from Barker Inlet, West Island (discussed in Section 5.2) and more recently, at Noarlunga Reef, aquatic reserves and other MPAs in South Australia are currently not monitored to assess impacts or MPA effectiveness. During the 1990s there was concern amongst fishing groups that the estuarine and shallow marine MPAs of upper Spencer Gulf were not being monitored, despite a government declaration at the time of MPA establishment that monitoring would be undertaken. Monitoring and environmental impact assessment work has been undertaken since the late 1990s at Noarlunga Reef, involving university researchers and community divers.

Until recently, one of the problems that exacerbated MPA impacts in South Australia was inadequate public knowledge of their existence. In 1986, for example, the South Australian Department of Fisheries commissioned a visitor survey of the Port Noarlunga Reef Aquatic Reserve, in which 43 000 observations were recorded during a 12-month survey period (Sutherland 1987, cited by Johnson 1988b). The survey showed that (i) fewer than 50% of visitors to the State's most popular aquatic reserve (Port Noarlunga / Onkaparinga Estuary) were aware that the area was a MPA; and (ii) fewer than one third of respondents knew which government department was responsible for managing the MPA. Despite the lack of knowledge of the MPA's existence, the Port Noarlunga Aquatic Reserve was popular (particularly with local people, divers and jetty fishers), with over half the respondents returning to visit the MPA more than 10 times per year (Johnson 1988b).

More recently, the agency responsible for managmeent of aquatic reserves in South Australia (Primary Industries and Resources South Australia) has provided signage near the reserves. Each sign describes why the reserve was established; the spatial boundaries (a map is provided); and what the management regulations are, for that reserve. Signage, in addition to production of MPA brochures, assists in the management of South Australia's aquatic reserves, by alerting more of the public to their existence, and the management regulations that should be followed.

8.4.4 Use of volunteers in MPA monitoring programs

Given the global trend in Western society during the 1990s for a reduction in public agency expenditure on monitoring and surveillance activities, alternative resources have been sought in some countries. In North America, volunteers ('citizen scientists') are collecting data that will be used to protect the resources of marine sanctuaries off the west coast (Roletto *et al* 1998). The volunteers' efforts are considered a valuable contribution to the information base for MPA management, including MPA inventory, species population research and risk assessment. Other benefits of utilising volunteer monitors includes public education (eg the volunteers from the west coast sanctuaries have helped to educate hundreds of locals about marine biology and the benefits of the MPAs, and the positive effects of this program reach thousands of visitors to the sanctuaries every year).

Also in the US, at the Flower Garden Banks Marine Sanctuary near Texas, 'non-expert' volunteers were recently involved in a fish-monitoring program for the MPA, involving 789 visual surveys. The program has shown that the volunteers produced survey results that were comparable with those of scientific experts (Pattergill 1998, 1999). Furthermore, the larger number of volunteers compared with scientists resulted in bigger sample sizes, which increased the statistical power of abundance estimates for some fish species. Apart from being a valuable data source for MPA management, volunteer monitoring invoked a sense of stewardship in the MPA.

Use of volunteers to monitor impacts and MPA effectiveness is likely to increase in the 2000s, particularly in countries that have few government resources for monitoring and managing MPA activities and impacts.

8.4.5 User fees for management and monitoring

Another avenue for monitoring in a climate of limited government funding for such activities is the use of MPA user fees. In Australia, the reef tax in the GBR Marine Park is used for marine monitoring and management, and has been accepted by the multitude of visitors. There are also entrance fees for the Jervis Bay National Park, which includes a marine component (Gentle 1994). In the Caribbean, the '$2-per-dive' fee charged in the Netherlands Antilles' Saba Marine Park yielded US$17 500 in 1992 to assist management (Dixon 1993), and, when combined with nominal fees for yachting, souvenir sales and donations, the MPA management became entirely self-supporting. Also in the Bonair Marine Park in Central America, management and operating costs of the MPA are covered by a US$10 per year ticket price, which divers must affix to their dive tank. Similarly, the Egyptian Environmental Affairs Agency is considering charging a small fee for each tourist visiting MPAs in Sinai, to assist with monitoring and protection work for coral reefs that can be damaged by tourism (EEAA 1997). There is now wide acceptance of 'park fees' for visitors in terrestrial national parks, and users of marine parks should similarly not object to a nominal fee charge to help protect the MPA (particularly if the users' impacts can be cumulative and damaging).

9 MANAGING THE WHOLE SYSTEM: NO MPA IS AN ISLAND

The decline from global to local scales of coastal marine water quality, benthic habitats, fisheries populations and biodiversity, shows that traditional marine management strategies are clearly inadequate. A more holistic, integrated approach is clearly required to ensure that the multiple values[13] of marine systems will not continue to be compromised by numerous, poorly regulated and uncoordinated activities and impacts. MPAs can be used as *one method* of protecting marine ecosystems, and improving the management and 'sustainability' of marine activities. However, successful marine conservation and management at local, regional and national scales obviously require concomitant, systems-based measures, as discussed below in Sections 9.1, 9.2 and 9.3.

Calls for a more systems-based approach to marine management began with the IUCN (1975, 1980), which stressed the need for nations to 'extend the principles of conservation and ecologically sustainable use beyond the bounds of protected areas (IUCN 1991, cited by Kenchington 1993). The need to couch MPAs in the context of whole system management has been considered in Australia since at least 1991, when the national Fenner Conference on the Environment discussed a strategic approach for MPA establishment in this country (Ivanovici *et al* 1993). The conference called for marine conservation efforts to move beyond sectorial (ie separate agency, separate activity) management, given the connectivity between marine systems (at all scales) and their biota, to an ecosystem-based approach (Kenchington 1993).

Whole system management requires the necessary links to be established between the legislation and management activities of different government agencies, industries, organisations, developers, and individuals that are connected with the marine environment. 'Connectivity' in this case refers not only to those who directly use marine resources or environments, but also to the 'upstream' activities that can affect marine ecosystems. The development of integrated coastal zone management in Australia during the past decade (eg Brown 1995) has seen a shift towards a more 'holistic', organised and cooperative approach to coastal management, and such development needs to be extended below the waterline to marine management.

In relation to MPAs, 'whole system' management of marine environments would obviously include management of potentially damaging activities outside MPAs, particularly those that will directly or indirectly affect MPA viability. Examples would include industrial, agricultural and domestic discharge of pollutants; 'unsustainable' fishing practices; pollutants from shipping and boating; inadequately managed aquaculture operations; land reclamation; construction; and mining, amongst others.

Three basic tenets of managing the 'whole system' are relevant to the establishment of MPAs, regardless of whether those MPAs are large, multiple-use marine parks, or small reserves for fisheries management, habitat protection or threatened species conservation. These tenets are

[13] Some of the multiple values of marine systems include (i) 'ecosystem services' that are usually taken for granted, and not economically valued (such as pollutant absorption, oceanic gas regulation, prevention of coastal and subtidal erosion, water quality maintenance, nutrient storage and recycling, and substrate production); (ii) provision of social and economic benefits, such as fisheries, recreation and tourism, and production of industrial and medical products; (iii) a storehouse of genetic diversity, which provides the material for maintaining future states of biodiversity and 'ecosystem health' at many levels; (iv) subjects for scientific research and marine education; and (v) 'wilderness' value, and the many other intrinsic values of undamaged marine environments and their component biota.

inter-related parts of a holistic approach to marine conservation and management, and are described below.

9.1 ECOSYSTEM MANAGEMENT

As early as the 1970s, when marine management commonly referred to little more than fishing effort controls, G. Carleton Ray impressed upon policy makers the need to manage marine environments in terms of whole ecosystems (Ray 1978). Ray considered that marine impacts such as effluent pollution, over-fishing and physical destruction of critical habitat, caused *alteration of process and function* in marine ecosystems. Obviously natural processes that structure marine systems and their component biota are not controllable by humans. It is the *human-induced changes to marine processes* that can and should be controlled. A few of numerous examples would include:

- the changes to system function that occur through the removal of large predatory fish from marine food webs by over-fishing
- the alteration of coastal marine physiographic processes due to near-shore constructions such as breakwaters, marinas and canal estates, which can affect sand deposition, water movement and larval supply to near-shore areas
- changes to benthic composition, habitat quality and water quality, due to channel dredging, stormwater or effluent discharge, trawling or intensive fish farming.

Whilst the protection of process is still, 20 years later, a difficult and unpopular concept, the result of ignoring the functional links within and between ecosystems is abundantly clear, in the form of degraded coastal marine habitats, declining coastal marine water quality, depleted fisheries, biodiversity reduction, and exotic species proliferation, to name a few examples.

Ray's (1978) plea for an ecosystems-based approach has been frequently echoed during the past decade, to improve the management of both terrestrial and marine environments and activities (Kenchington and Agardy 1989; Agardy 1994; Grumbine 1994, 1997; Edyvane 1996a; Wilder *et al* 1999). In Australia, the Commonwealth Government has adopted the ecosystem management paradigm, in its call for '*maintenance of ecological systems and protection of biodiversity*', one of the principles of ecologically sustainable development that is now an internationally-accepted goal of environmental management (ESD Fisheries Working Group 1992).

Ecosystem management could be described as spatially-integrated management of coastal marine ecosystems and activities, that crosses jurisdictional boundaries, and considers the functional connections between parts of marine ecosystems that have traditionally been managed separately. For example, river mouths/estuaries are functionally connected to the subtidal ecosystems (reefs, seagrass meadows, sand beds, etc), into which they deposit water, solid wastes, chemicals and biota. In the opposite direction, marine environments are functionally linked with estuaries—for example, by the movement of marine fish into estuaries to breed, or by the larvae or juveniles of marine species that use estuaries as nursery areas. Separate legislation and agencies have traditionally been involved in the management of the 'depositing' and 'receiving' environments in both directions, to the detriment of the entire system.

Approaches to ecosystem management are summarised below, in relation to two major impacts on marine ecosystems: fishing and pollution. A desired approach to ecosystems management would be to ensure that fisheries really are 'sustainable' in the true sense of the word, and to protect marine ecosystems and their biodiversity from the effects of pollution, habitat degradation and fishing impacts (Larkin 1996).

9.1.1 Ecosystem management of fisheries

Fish stocks and fisheries management benefit from an ecosystems-based approach. In an ecosystem framework, fishing is managed as an activity that affects entire marine ecosystems, not just the particular stocks that are being managed by ‘input’ and ‘output’ controls and regulations. Steele (1991) considered that an understanding of marine ecosystem structure and function is essential for long-term effective fisheries management. Amongst others, Ludwig *et al* (1993), Huntsman (1994), Dayton *et al* (1995), Hilborn *et al* (1995), Lauck *et al* (1998) and Murray *et al* (1999) all provided examples of overfished stocks and the associated ecosystem effects, and called for fisheries to be managed on an ecosystem basis. There are numerous examples of the ecosystem effects of fishing, such as the following:

- *Population depletion because ‘prey-switching’ does not occur*: In a natural system, many predators in the marine environment would ‘switch’ prey species when their usual food supply declines in abundance (eg due to an oceanographically-induced poor recruitment season). If we consider fishing in its ecological context, as another form of predation, economic and social structures often reduce the opportunity for ecologically-balanced ‘prey-switching’, and the system becomes ‘unbalanced’. Fishers may continue to fish for the species for which their gear is set up to catch, which they are legally permitted to catch, or which the public values highly, regardless of stock decline. There is high incentive to ‘fish down to the last’ and politically lobby to ‘keep their fishery going’.
- *Serial depletion*: Conversely, in fisheries where ‘prey-switching’ is an option, fishers may serially deplete different species, moving on to less valuable and / or smaller species as the highly prized, larger species are depleted. An example is the replacement of the yields from the crashed cod and haddock fisheries of the North America with species that were previously considered to be ‘trash fish’ such as dogfish and skates (Wilder *et al* 1999). Similar fishing patterns occur with advances in fishing technology: species which formerly had ‘refuge’ in deeper water or otherwise inaccessible areas, are targeted when the means to catch them becomes available, and often following the depletion of more accessible coastal species. Despite the change in species composition of the landed fish and the serial depletion of populations, the overall fishery yields often remain buoyant, and thus there is often little pressure from fishers and fisheries managers upon governments to exercise restrictions.
- *Species replacement*: This effect is related to both serial depletion from over-fishing (see above) and fishing-induced habitat damage. A common example is the increase in populations of small (often herbivorous) species, when large, carnivorous species are depleted. For example, ‘opportunistic’ or ‘industrial’ species such as Norway pout, sprats, and sand eels proliferated due to the decline of herring and mackerel fisheries in the North Sea[14]. An Australian example is the change in yields from the heavily trawled northwest shelf, where smaller, low-value fish such as butterfly bream and lizardfish replaced large high-value fish such as snappers and emperors. This replacement was considered to be a response to trawling-induced habitat damage (Sainsbury *et al* 1993, 1997). When species replacement occurs, fishers often have no option but to ‘fish down the food chain’.
- *Species decline with no apparent recovery:* This is perhaps the most widely publicised ecosystem effect of fishing: crashed fish stocks that fail to recover. Examples include some of the clupeoid fisheries in the northern hemisphere, where the combined effects of

[14] Up until the 1990s, approximately 20 million tonnes of Norwegian herring had been removed from the North Sea.

oceanographically-induced spawning failure and high fishing levels decimated the stocks: herrings, sardines, pilchards, and anchovy populations haven't fully recovered even after drastic cuts in the formerly high fishing effort levels. The Arcto-Norwegian Cod is another example, as is the Icelandic spring-spawning herring – considered to be a classic case of fisheries collapse, because the stock has failed to re-appear after 20 years of drastic effort reduction. The ecosystem effects caused by the removal of millions of tonnes of single species biomass are still poorly documented, but some of the more obvious effects include loss of productivity in marine food webs, declines in marine species populations that relied upon the fish as a food source, and changes in the structure and composition of entire marine ecosystems (see below).

- *Great loss of productivity that keeps marine food webs balanced*: This includes reduction in food supply for top marine predators and other fish-eating marine biota (sea birds, sea lions, seals, dolphins, etc), and the consequential change in population numbers of those species. For example, decreased numbers of harbour seals, sea lions and fish eating birds in the Bering Sea area have been related to the intense fishing of wall-eye pollock. The estimated global fisheries catch is in the hundreds of millions of tonnes, and although the ecosystem effects have only recently been the subjects of research, the impacts on total productivity are highly significant.
- '*Ecosystem shifts*', '*biomass flips*' and '*trophic cascades*': These effects refer to a change in ecosystem structure, function and species composition that can result from over-fishing the major functional groups (often large predatory species) that keep ecosystems 'balanced'. A startling example was reported by Schoning *et al* (1992, cited by Dayton *et al* 1995), in which a bycatch of over one million large sharks from a northwestern Atlantic swordfish fishery resulted in an order of magnitude increase in the sharks' prey species (grey seals). This 'seal explosion' in turn led to increased infection of cod from parasites harboured by the seals, as well as high increases in destruction of fishing gear by the seals. The seal population subsequently crashed, in part due to the stress of abnormally high population densities. There are also instances of fishing effects interacting with pollution effects, to cause significant ecosystem changes, such as the smothering of coral by a proliferation of 'nuisance' algae on reefs, due to the combined effects of fishing herbivorous fish and nutrient pollution (Botsford *et al* 1997). Dayton *et al* (1995) and Goni (1998) provided an international overview of 'ecosystem shifts', such as the proliferation of echinoderms and consequent benthic habitat damage, following the removal by fishing of predators from marine ecosystems. A similar Australian example was reported by Environment Australia (1998): damage to marine algae and associated benthos from the apparent proliferation of sea urchins resulted when their major predators, the grey nurse shark and the blouper grouper, were overfished.
- *Potential loss of genetic diversity*: The genetic diversity of heavily fished populations can be reduced (see Section 3), which affects the populations' chance of adapting to environmental change or to 'natural population disasters', such as disease outbreaks. The situation can be particularly adverse for spatially-separated populations, which are linked reproductively: for example, some parts of the population may rely upon the genetic transfer (larval supply) from a heavily fished portion of the population in an 'upstream' area. The selective removal of older, larger individuals through fishing can also affect genetic diversity (Bohnsack 1993), resulting in smaller or even 'stunted' animals in heavily fished populations (eg see Shepherd's 1995 example for abalone). Goni (1998) reviewed the paucity of data on reductions in genetic diversity from over-fishing, and noted visible effects of genetic change such as change in growth rates, maximum size and reproductive age / size of fish.

- *Bycatch impact on ecosystems:* At a global scale, dozens of millions of tonnes of non-target (bycatch) species are killed and discarded annually from otter trawling, purse seining, various types of set nets such as gill and drift nets, and long-lining. The impacts of fishing on bycatch species have received much publicity during the 1980s and 1990s. Dayton *et al* (1995) and Goni (1998) provided detailed reviews of the bycatch effects of fishing (particularly trawling and other forms of netting), including highly significant reductions in benthic species abundance, size structure and biodiversity. Some of the many millions of non-target marine vertebrates and invertebrates that are injured or die every year include dozens of species of marine mammals, sea birds, sharks and rays, and thousands of species of fish, crustaceans, echinoderms, molluscs, bryozoans, calcareous worms and other marine worms, coral, and sponges, amongst many other groups. In Australia, most reporting of bycatch effects has come from northern trawl fisheries (reviewed by Cappo *et al* 1998); however, bycatch is also an issue in some southern Australian fisheries that has received little attention until the late 1990s. 'Ghost fishing' also has a major ecosystem impact in many marine ecosystems, in which discarded crustacean pots, nets, fish traps, drift hooks and lines, and/or nylon strapping from bait boxes, are responsible for the death every year of countless thousands of marine mammals, reptiles, seabirds, fish and other biota. 'Ghost' fishing equipment can continue to kill marine biota for many years after it is abandoned.
- *Habitat destruction and associated impacts on fauna:* Trawling and 'fish dredging' over both soft and hard sea bottoms disturbs the benthic topography and can result in increased turbidity from benthic damage and sediment mobilisation (Dayton *et al* 1995; Collie *et al* 1997; Engel and Kvikek 1998; Prena *et al* 1999). One of the most significant effects of benthic habitat damage from fishing is the destruction of living benthic structures, such as corals, bryozoans, sponges, seagrass, macro-algal stands and calcareous algae. Such structures provide physical support for marine communities, and help to maintain biodiversity. Apart from reduction in abundance and spatial cover of these types of biota, their removal can adversely affect the survival of fish and invertebrates that used the benthic structures for recruitment, feeding, and/or shelter from predators. In New Zealand, for example, destruction of bryozoan beds by trawlers was correlated with a reduction in the number of juvenile fish of commercial importance, which relied upon the bryozoan habitat (Bradstock and Gordon 1983). Another impact on habitat occurs from the discarded offal and other waste products that result from fish processing at sea, which can cause benthic oxygen depletion and change in species composition. Dayton *et al* (1995) and Goni (1998) reviewed the reported instances where benthic habitat destruction affected both target and non-target species populations as well as biodiversity. Damage to the benthos also fragments available habitat, which can reduce the chance of successful recruitment and colonisation for a variety of species with limited dispersal capabilities (see references in Ruckelshaus and Hays 1998).

'Whole system' management of fisheries, including fisheries MPAs, requires an acceptance that fishing can have serious ecosystem effects such as those discussed above. Ideally, ecosystem-based fisheries management could draw upon the results of research on (i) the ecosystem linkages of fisheries stocks, and (ii) the distinction (and the interactions) between the effects of human-induced impacts and natural environmental variability on habitat quality, species composition, and fish stock abundance and distribution over space and time. Increasingly, it is being recognised that marine populations may vary unpredictably, or exhibit cyclic fluctuations (Steele 1985), and often change in distribution, abundance or population structure when oceanographic conditions change, or when habitats are altered, either naturally or by human impacts. There have been numerous pleas for fisheries management to consider the physical, chemical and biological processes and functions of marine ecosystems and the human-induced impacts upon those systems (Salm and Clark 1984; Kenchington 1990; Ray 1985, 1991; Steele 1991; ESD Fisheries Working Group 1992; Edyvane 1993; Botsford *et al* 1997). However obvious it is becoming that such knowledge should be considered in an ecosystem approach to fisheries, the knowledge base is still rudimentary compared with understanding of single species dynamics. Also, it is still very difficult to separate the effects of environmental variables from the effects of fishing on marine populations, particularly those in coastal ecosystems (Goni 1998).

Understanding ecosystem functioning is one of the most difficult goals for marine research and management, but significant developments have occurred internationally during the past decade. Examples include both mensurative and experimental work in trophic dynamics (eg food web analysis); mapping and GIS-based correlative analysis of physical processes with biological distributions; mass-balance trophic models such as ECOPATH (Pauly 1998) and its spatial simulation modelling successors, ECOSIM and ECOSPACE; and other models incorporating empirical relations between primary production and several levels of consumption in marine ecosystems (Brey 1999). Such models can include fishing as part of the modelled predator-prey interactions, and can be used with food web data, long-term catch and effort data from fisheries, historical records of marine environmental conditions, and even socio-cultural data, to:

- ‘reconstruct’ ecosystems and their components, as they would have existed prior to long-term impacts from fishing and pollution (University of British Columbia 1998a, 1998b), to provide informed policy choices for ecosystem management
- evaluate different fisheries management strategies in multi-gear, multi-species fisheries, considering the ecological interaction and impacts of those fisheries (Pauly 1998; Bundy 1998)
- evaluate the trophic roles of commercially and recreationally fished species, such as large predatory fin-fish (Arreguin-Sanchez and Manickchand-Heileman 1998), the over-fishing of which can result in significant ecosystem impacts
- build trophic models of the interactions between consumer groups in marine food chains/food webs (Rosado-Solorzano and Guzman del Proo 1998), to assist conservation and management of these groups, particularly if they are fished.

However, there has been some criticism of ecosystem models for fisheries management. Larkin (1996) suggested that more traditional methods such as optimisation of fishing fleet activity and interactions, and multi-species virtual population analysis (VPA) models[15] offered more practical guidance for fisheries management in an ecosystems context. Both ecosystem models and fisheries models require large amounts of data, and it is hoped that such applications for which data are available will at least provide more ecologically informed choices for management. Other developments that can assist ecosystems management include the work of Done and Reichelt (1998), who have developed an index-based approach to monitoring the ecosystem effects of fishing (amongst other activities that affect coastal marine ecosystem functioning).

More consideration of historical fisheries data and other marine data sets is also required to elucidate the long-term effects of fishing on marine stocks and the ecosystems of which they are part. Tegner and Dayton (1998) and Dayton *et al* (1998) have described the ignorance of

[15] VPA uses catch and effort statistics and current stock sizes to back-calculate the population sizes and dynamics of stocks in an unfished state.

data on former levels of stock abundance, species composition and habitat quality, as operating within a 'sliding baseline'. In other words, each generation of fisheries scientists and managers accepts the current stock sizes and ecosystem states as a 'baseline' for management, which masks long-term, incremental, cumulative impacts that result in ecosystem damage and decimation of fisheries stocks.

In Australia, a recent national report commissioned by the Fisheries Research and Development Corporation's Ecosystem Protection Program, reviews and synthesises Australian fisheries habitat research (Cappo *et al* 1998), and discusses the significant new developments towards ecosystems management in some Australian fisheries. This includes recent advances in fishing gear technology in Australia, such as inclined grids and mesh panels in trawls, that have helped to reduce two of the major ecosystem effects of fishing: bycatch mortality and benthic habitat damage.

In South Australia, there has been little progress during this decade in ecosystem management of fisheries, despite recommendations for its application (Edyvane 1993; Shepherd 1999). Appendices 4 and 5 of Cappo *et al* (1998) showed that very little applied marine ecosystem research has been undertaken in South Australia, particularly in relation to fisheries. That which has been undertaken has been mostly restricted to ecosystems at West Island and Barker Inlet, through the numerous long-term and short-term studies of Shepherd, his collaborators, and other marine researchers at the former site (rocky reef), and the various studies by Jones and collaborators (eg Jones 1984; Jones *et al* 1996; Jackson and Jones 1999) and by Connolly (1994a, 1994b, 1994c) at the latter (estuarine area). Although there is still lack of public knowledge in South Australia of fisheries ecosystem management and the research it requires, recent projects which are designed to investigate marine food web relationships, trawl bycatch composition and the effects of fishing gear on benthic habitats, indicate that the concept is finally gaining research and management interest.

Understanding ecosystem linkages and the combined effects of fishing and physical processes on marine ecosystems and their biota is a long-term research and management goal that may take decades to elucidate. Academic calls for more research to be undertaken before management acts to control ecosystem effects can no longer be justified. The implementation of adequate fishing controls and gear regulations outside MPAs (and inside multiple-use MPAs) can be attained over a much shorter time than the achievement of idealised research goals. Indeed, development of precautionary and adaptive management frameworks (see Sections 9.2 and 9.3) to reduce the fishing pressure that causes ecosystem effects is becoming imperative on a global scale, and waiting until ecosystems knowledge is vastly improved cannot be afforded (Botsford *et al* 1997).

9.1.2 Ecosystem management of coastal marine pollution

Protecting both MPA and non-MPA marine ecosystems / habitats from pollutants, particularly chemical pollutants, is obviously an enormous and onerous task. Most coastal rivers, estuaries, drains and stormwater outlets near inhabited areas coast carry domestic, agricultural and / or industrial water-borne effluents to the coast. Local oceanographic processes and coastal configurations (physiography) can transport and concentrate those pollutants in habitats within gulfs and bays, or transport them further offshore, where they can affect many different ecosystem types. Other major sources of marine pollution include ballast water from ship and boats, dumping of solid wastes and chemicals at sea, and hydrocarbon emissions from ships, boats and ports (ie apart from oil spills, there is a continuous stream of hydrocarbon pollutants from shipping and boating activity). When those pollutants are assimilated into the benthic environment and / or taken up by the biota, the amenity of an area as a MPA can be seriously compromised.

Control of marine pollutants is particularly important, because their effects are often not obvious, yet they are cumulative, often synergetic, and can be spread widely from the source, due to the connectivity between marine systems. Control of marine pollution includes *physical* (eg land-based sediments and marine sediments mobilised by human impacts), *chemical* (eg sewage, stormwater, agricultural chemicals, heavy metals, chlorinated hydrocarbons) and *biological* (eg marine pests, of which there are now more than 50 established in Australia) detriments to the marine ecosystems, and their *interactions*. 'Whole system' marine management includes 'upstream' areas away from the coast, if activities in those areas affect marine environments. An example would be the long-term, poorly-controlled runoff of tonnes of soil and agricultural chemicals into the Great Barrier Reef Marine Park, from land clearing and farming, which has increased coastal water turbidity and killed areas of coral, amongst other effects. Without control of upstream runoff, site management of impacts and activities will be fruitless in the long term, on parts of the reef that are affected by the runoff. Ignoring the systemic links between terrestrial and marine ecosystems has caused similar problems in the Gulf of Mexico – excessive release of land-based fertilisers, pesticides, sediments, street oils and other pollutants have caused algal blooms and consequent deoxygenation of marine ecosystems, resulting in mass fish kills and 'dead zones' of 'lifeless' water that are considered to be the largest in the western hemisphere (Wilder *et al* 1999). There are numerous other marine ecosystems around the world that have suffered a similar fate.

Protecting one part of the system is not enough. A simple example of systemic effects would be the mistaken belief that protecting an estuarine MPA that is subject to sewage or stormwater discharge from physical habitat destruction (eg dredging) or unsustainable fishing practices would sustain the fish and crustacean stocks in that MPA in the long term. Nutrient pollution of the system would likely result in fish stock decline, regardless of the level of protection of the stocks 'on paper', by causing:

- proliferation of microalgae and / or macroalgae, smothering of seagrass, decreased light penetration for seagrass growth and consequent death of seagrass
- increased turbidity and sedimentation from seagrass death and consequent benthic destabilisation
- habitat change due to proliferation of filter feeders and detritus feeders
- reduced availability of seagrass habitat for incoming larvae to shelter and grow in
- potential decline in fish and crustacean species, which relied upon the seagrass for shelter from storms / currents and predators, and as food sources (especially epiphytic food on seagrass blades).

Scientists have spent many years investigating ecosystem linkages such as these, and many would claim that increased funding and opportunities are warranted for further ecosystem investigations. Unfortunately, as discussed above in the examples above on fisheries impacts, pollutants are degrading ecosystems at a rate that far exceeds the capacity of scientists to study them. Management should not afford to wait, considering (i) the need to apply a precautionary approach to coastal marine management (see Section 9.2), and (ii) that the signs of ecosystem degradation (eg seagrass loss, sedimentation of reefs, reduced biodiversity) are obvious, even if the full functional web of ecosystem effects is not known. From a practical viewpoint, perhaps managers don't need to know all the ecosystem links in the process before controlling some types of ecosystem impacts, but simply the start point and observed end points – eg pollution goes in: habitat is degraded; species composition and abundance are changed. In other words, simple results can emerge from complex processes, and explicit details of the processes are not required before management action is taken to prevent ecosystem-level impacts.

Cappo *et al* (1998) reviewed the impacts of pollutants on near-shore marine ecosystems in Australia, and pollution issues that specifically affect Australia's current system of MPAs are summarised by Zann (1995a, 1995b, 1996). Controlling the ecosystem effects of pollutants will require concerted efforts at local, State, Commonwealth and even international levels. In South Australia, pollution issues that must be mitigated for marine ecosystems to be well managed (including those in a MPA system) are discussed in Section 5.3.5, and detailed in Edyvane (1996c) and Lewis *et al* (1998).

Important processes in ecosystem management were summarised by Edyvane (1996a), who stressed the need for a variety of ecosystem management strategies, including the following:

- marine managers to work with ecosystem boundaries, rather than administrative and political boundaries
- marine research and management to consider the hierarchically scaled nature of marine systems (and to therefore define a hierarchical, bioregional framework for management that considers the relationships between process and activities from large, biogeographic scales down to local scales)
- researchers to investigate human impacts on ecosystem structure and function
- ecosystem researchers and marine managers to better utilise spatial simulation models, geographical information systems and other decision support systems.

9.1.3 Ecosystem management and MPAs

As discussed above, the connectivity between physical, chemical, biotic components of marine systems means that factors affecting any one of these can have highly detrimental, yet often unobvious, effects upon other components of marine systems. Protecting the functioning of MPAs and other parts of the marine environment requires cooperation between the many agencies responsible for managing parts of the whole, and between the numerous user groups engaged in both extractive and non-extractive activities.

In the context of MPAs, Agardy (1994) considered that marine planning and management must be ecosystem-based, in order for MPAs to effectively conserve marine environments and resources, as well as accommodate the diverse uses of many different stakeholders. Ideally, ecosystem management entails knowledge of the functional links between marine ecosystems/habitats and their biota, including those inside and outside MPAs, to define 'functionally viable' management units (Kenchington and Agardy 1989, Agardy 1994). This is a tall order for both scientists and managers, considering that knowledge of coastal marine ecosystem function is still rudimentary in most countries (but see examples in Section 9.1.1), and the current sectorial management arrangements in most coastal marine areas do not consider the functional links between marine systems. In any case, ecosystem components often exhibit unpredictable behaviour, which often leads to confusion between 'natural' and 'unnatural' (ie human-induced, and thus hopefully manageable) variations in the state of marine ecosystems, such as habitat quality, species composition, and fish stock abundance, for example.

In practical terms, managers don't need to have *every* piece of the puzzle fitted together for them by scientists before taking action to conserve marine environments and their component biota. MPAs have been recommended as one vital component of an ecosystem management framework for protecting both specifically valued marine resources and biodiversity in general (Ray and McCormick-Ray 1992; Agardy 1994; Edyvane 1996a; Botsford *et al* 1997; Wilder *et al* 1999).

In a 'whole system' framework, in which impacts inside and outside the MPAs are controlled, the MPA portions of marine ecosystems remain 'intact', and can thus be used to improve ecosystem knowledge and to provide more educated management options. MPAs can also provide a 'testing ground' for elucidating the impacts of fishing and other activities on marine ecosystems, as discussed in Section 9.3, on adaptive management. MPAs can thus reduce the uncertainty about the effects of fishing on marine ecosystems (Botsford *et al* 1997). MPAs are considered to provide one of the only means of separating the effects upon marine populations of human-induced impacts compared with natural environmental stresses. They also serve as sites for biodiversity inventory, and for monitoring biodiversity changes. Another ecosystem research function of MPAs is their role in studies aimed at understanding the functional links between fished and non-fished parts of marine communities, and between biota inside and outside of the MPA (Ruckelshaus and Hays 1998).

Ecosystems models such as ECOSIM and ECOSPACE have recently been used to assess the utility of MPAs in the context of (i) the trophic interactions required to maintain exploited fish populations and (ii) the habitat preferences of fished species, and effects of degraded habitat quality (Walters *et al* 1998). The development of a set of ecosystems performance standards (Done and Reichelt 1998) also holds much promise for monitoring the effects of MPAs, including their function in fisheries sustainability and stock replenishment.

9.2 THE PRECAUTIONARY PRINCIPLE

One of principles of ecologically sustainable development in Australia is the need to 'deal cautiously with risk, uncertainty and irreversibility' (ESD Fisheries Working Group 1992).

The 'precautionary principle', often cited and rarely defined, is part of whole system management, and requires that managers must take a conservative, cautious approach to marine uses and impacts, in the face of scientific uncertainty about the response of marine ecosystems and resources to both human-induced impact and natural stresses. Under the precautionary principle, actions that are likely to produce irreversible changes to ecosystems must be avoided (Agardy 1994).

The precautionary principle calls for a shift in the 'burden of proof'—for example, there have been repeated requests for industries that catch fish, intensively farm marine species, or issue known pollutants into marine environment to demonstrate that their actions do not cause species population declines, loss of biodiversity or 'ecological harm' (Wilder *et al* 1999). (See Section 9.1 for examples of 'ecological harm'). This is the reverse of common and current management practice, which permits potentially damaging practices to continue to the level at which impact is obvious, upon which reactive management measures are applied.

The growing list of stock declines and seemingly irreversible damage to many marine ecosystems from both acute and chronic pollutants, indicates that marine managers and stakeholders still rarely consider the precautionary principle. Marine environments are still largely considered, if not in theory, then certainly in practice, to be both bottomless waste receptacles and endless providers of a bountiful harvest.

9.2.1 The precautionary principle and fisheries management

Fisheries management in both developed and developing countries has traditionally not adopted a precautionary approach. There are many reasons for this, including the following:

- Numerous social, economic and political reasons (eg see Ludwig *et al* 1993), such as the economic importance of fisheries for domestic and export markets; the perceived 'rights' of recreational and commercial fishers; ‘the tragedy of the commons’ (Hardin 1968), by which a large number of people use public resources and contribute to their over-exploitation, but expect someone else to pay the costs associated with that over-exploitation; the ‘gold rush’ mentality, in which politicians and governments ally themselves with lobby groups when large gains from resource exploitation are in prospect (Ludwig *et al* 1993); and the so-called ‘ratchet effect’, which causes managers to maintain or increase fishing levels, regardless of stock abundance, due to the constant political pressure associated with short-term benefits to society (jobs, profits, election wins, etc) (Botsford *et al* (1997).

- Poor understanding of the time lags in impacts, coupled with the fact that cumulative, incremental, synergistic, human-induced impacts may not be obvious in marine systems, due to the limitations posed by underwater sampling, and the general inaccessibility of the marine environment for many people, compared with terrestrial environments. Consequently, the effects of unsustainable fishing practices are not important to many people because they cannot easily be seen.
- A previously unquestioned faith in the single species population models and paradigms that have driven applied fisheries science for much of this century. One example of the ‘reckless’ approach to fisheries management is the ridiculous (but commonly accepted) notion that fish recruitment and consequent long-term abundance are driven entirely by density-independent oceanographic processes, and are thus independent of fishing levels. The collapse through over-fishing of even the ‘best’ researched and managed fisheries (Ludwig *et al* 1993; Roberts 1997; Botsford *et al* 1997; Goni 1998) has finally debunked this fisheries myth.
- Difficulty in determining the distribution and abundance of fished species over space and time, due to the highly mobile nature of many marine species, the spatial separation of the various life stages in many coastal marine species, and the often unpredictable nature of recruitment (Rothschild 1986; Kenchington 1990; Fairweather and McNeill 1993).
- Inadequate knowledge of (i) the integrated physical and biotic processes, such as tides, currents, water temperatures and plankton blooms, etc, that greatly influence the spatially-separated life stages of many marine species and (ii) the critical habitats upon which some life stages of fisheries species depend for their survival and reproduction.
- Scientific uncertainty and conjecture about absolute stock sizes and recruitment levels; unknown relationships between the fished stocks and the physical, chemical and biological variables that affect them, and lack of predictive power regarding the reaction of the stocks to fishing over space and time.
- The fact that the behaviour of both fishers and fish can mask the signs of over-fishing. Fishers’ behaviour that tends to result in underestimation of fishing effects on stocks and ecosystems, are discussed in other sections of this report. A summary would include:
  - improved gear, and increased knowledge and efficiency of fishers, due to satellite navigation, benthic plotters, and echo sounders, for example
  - accumulated knowledge over time of the spatial distributions of fish; changes in fishing patterns in response to local declines in stocks, such as fishing further from shore (see Keesing and Baker 1998 for a South Australian example)
  - increased fishing in areas that were once ‘natural refuges’ for fish due to their former inaccessibility, and ‘fishing down the food chain’ to maintain yields.
- Fish behaviour and responses to fishing that ‘compensate’ for over-fishing and can thus mask declines until critically low stock levels are reached include:

- faster growth rates and/or earlier maturity (eg Cockrum and Jones 1992, for South Australian whiting, and Goni 1998 for lobster, plaice, rays and other northern hemisphere examples)
- aggregation behaviour of many species, so catches initially remain high despite decline in total population numbers (eg South Australian examples include snapper, abalone, cuttlefish)
- delay in the visible effects of over-fishing the spawning stock biomass and size/age structure of fish populations, and in the fishing-induced decline in genetic diversity, until critically low population levels are reached
- the high dispersal capabilities of marine species, and the 'replenishment' of over-fished areas by migrating fish, so that catch rates initially remain high whilst the total fish population is declining
- periodic 'good' recruitments, often oceanographically-mediated, which encourage fishers to fish harder in the 'good years', without considering that fishing-induced reductions in stock levels will eventually weaken the resilience of the stock to the effects of 'bad recruitment' years.

⅄ 'Sliding baselines' (Tegner and Dayton 1998; Dayton *et al* 1998), in which scientific assessments and management decisions are made using data and knowledge that are limited to the present, and ignorant of historical states (eg of fish abundance, species composition, habitat quality, etc) (see Section 9.1.1).

Precautionary approaches have been particularly difficult to justify because of these reasons, and there is thus usually little incentive or management directive to reduce catch and effort in fisheries. The decline in fisheries stocks worldwide has been exacerbated by the forementioned factors. Fishers are now so efficient and persistent that many species populations are being caught much faster than they can be produced.

There is now heightened awareness of the fact that fish populations are part of complex ecosystems, and are thus affected over various space and time scales by the combined effects of biological interactions among species, environmental variability, and fishing pressure (Section 9.1.1). Despite this awareness, traditional notions of the public 'right' to keep fishing, the infallibility (rather than the uncertainty) of scientific data, and the unerring abundance of marine resources all still pervade many fisheries management decisions, despite a climate of 'sustainability' rhetoric.

Roberts (1997), Botsford *et al* (1997), Lauck *et al* (1998) and many others have called for the precautionary principle to be adopted in fisheries management. These authors all suggested that MPAs are one of the best methods of applying precautionary management principles, particularly for the benefit of fisheries management (see following section, 9.2.2). Hilborn (1996, cited by Cappo *et al* 1998) recommended adoption of the precautionary principle in Australian fisheries management.

9.2.2 The precautionary principle and MPAs

MPAs provide an opportunity for the precautionary principle to be applied, by protecting areas from a variety of impacts whose synergetic and cumulative impacts are unknown, and allowing for valuable comparisons with the areas outside the MPA.

Fisheries MPAs ('harvest refugia', 'fisheries reserves', 'closed areas', 'no fishing zones', 'fish sanctuaries', etc) are considered to be one of the most useful precautionary measures for managing fisheries (Roberts 1997; Botsford *et al* 1997; Lauck *et al* 1998).

MPAs permit the application of precautionary management in the following ways, by:

- reducing the scientific and managerial uncertainty about fish population numbers over space and time, and the response of those stocks to fishing pressure
- providing an insurance mechanism for the fishery, in light of non-conservative management decisions, few sound indicators regarding the long-term sustainability of fishing rates and yields, and periodic changes to stock abundance, distribution and/or size structure. (Note that even fisheries for which good quality long-term scientific data are available have been known to collapse—see references cited in Section 9.1)
- shedding light on the ecosystem effects of fishing (see Section 9.1), and on the effects of pollutants on marine ecosystem structure and functions (the latter example referring to MPAs in which pollution is controlled, and which can be compared with corresponding areas that have suffered impacts).

MPAs are an important, but not exclusive, part of precautionary management. The precautionary approach must extend beyond the boundaries of MPAs, into all fisheries management strategies and regulations, and into ecosystems-based pollution control strategies. For example, if MPAs contain populations which are connected to critical habitat or migration paths outside the MPAs for part of their life, and if the associated fishery is over-fishing (i) the young adults leaving the MPA, and/or (ii) an offshore adult spawning stock that supplies the MPA with larvae, then the MPA will likely fail in the goals to provide a refuge for juvenile fish growth and a source of 'export' for the fishery. Similarly, attempting to protect biota in a MPA that is subject to numerous polluting impacts (effluent discharge, sediment mobilisation from coastal structures, exotic species introduction, etc) cannot be considered as a precautionary management strategy.

9.3 ADAPTIVE MANAGEMENT

The other major requirement for effective whole system management, related to the precautionary principle, is adaptive management practices. Walters (1986) first suggested adaptive management as a means to counteract the failures in natural resources management that have occurred too often during this century. In the ensuing years since the development of that approach, numerous new examples of collapsed fisheries, destruction of benthic habitat, changes to ecosystem structure and function, and rapidly deteriorating coastal water quality, show that the mistakes of the past are still being repeated in the management of marine activities. In a call for the application of adaptive management, Ludwig *et al* (1993) and Parma *et al* (1998) considered that assessments and forecast models in fisheries often fail, due to (i) inadequate knowledge of the processes that drive (marine) systems, and (ii) underestimation of the inherent uncertainty in the responses of ecosystems and their biota to both environmental variables and human-induced impacts.

Adaptive management in marine systems encourages experimental manipulation of systems *in situ*, and permits the testing of hypotheses about the likely causes of change to fisheries productivity, ecosystem structure or function, and the interactions between them. In some cases, models can be used to supplement an experimental approach. For example, Bundy (1998) used the dynamic multi-species ECOSIM to explore interactions between fishing and ecosystem function, and to assess fishing impacts under alternative management scenarios.

9.3.1 Adaptive management using MPAs

MPAs can be used to test hypotheses about the likely changes to fisheries productivity and ecosystem structure and function, and to monitor the 'responses' of the system to conditions and activities inside and outside of the MPA. This permits more informed management decisions to be made, and a precautionary approach to be adopted. Temporary MPAs ('closed areas') can be used as good 'case studies' for application of adaptive management principles to a wider area, by providing areas in which the effects of experimental fishing levels can be tested (Elder 1991; Agardy 1994; Smith 1996; Botsford *et al* 1997).

Sainsbury (1991, cited by Smith 1996) and Caputi (1993) provided examples for tropical multi-species fisheries and prawn fisheries in Australia, where changes in the systems (altered species composition in the multi-species fishery, and decline in yields in the prawn fishery) were observed. Various hypotheses about these changes to the systems were tested using closed areas. Over time, comparison of populations inside and outside the closed areas showed that the impacts observed prior to the closures were both caused by fishing practices (trawl-induced benthic damage and over-harvesting, respectively), and not by environmental variables. In the latter case, closing an area helped the prawn population to recover abundance.

Adaptive management experiments have also been conducted on the Great Barrier Reef (Parma *et al* 1998), to investigate the effects of spear fishing and line fishing on reef fish populations, using a combination of areas open to fishing, temporary MPAs (which are periodically opened to fishing), and permanent 'no-take' zones. In South Australia, there is one experimental management area for investigating the effects of long-term harvesting on the population dynamics of an abalone stock, at Waterloo Bay (Shepherd 1995), where a 'temporary MPA' has been established.

Many of the recoveries of over-fished populations cited in Chapter 3 could not have been accurately predicted prior to MPA establishment. However, the closures have added weight to the hypotheses that these populations declined due to fishing, rather than for other reasons, and provide information for better management decisions to be made. Such knowledge could not have been gained without the use of MPAs to 'test' the response of these fished populations to release from fishing pressure.

In areas where experimental manipulation of fishing effort levels is infeasible, *de facto* adaptive management can occur, by altering management arrangements in response to changes in either fish stock abundance or habitat quality. Management should also adapt fishing regulations over time in response to improved knowledge of ecosystem function and the ecosystem effects of fishing (see Section 9.1.1).

10 WHY SOME MPAs FAIL

The examples listed in Chapters 3, 4 and 5 show that many MPAs have achieved their various aims of protecting and enhancing fisheries resources; protecting critical habitats / ecosystems; conserving threatened species; maintaining local biodiversity; and providing undamaged areas for recreation / tourism, education and research.

However, not all MPAs are successful in practice. Obviously, MPAs can fail to protect habitats and their associated biota if physical, chemical and / or biological agents (see Chapter 9) damage them. MPAs can also fail if habitat quality is seriously degraded from over-use of the MPA by visitors (see Chapter 8). MPAs for fisheries enhancement are also sometimes not successful, due to poor design (eg inappropriate size and / or location), degradation of habitat, or breach of management regulations. Regarding the latter reasons for MPA failure, there is evidence (see Chapter 3) that it may take several years for fisheries MPAs to replenish the size and densities of site-attached fish that have been depleted by over-fishing (Russ and Alcala 1996b). Fishers hoping for short term gains from MPAs can be disappointed with the required lag time, and start fishing in the MPA. Others take advantage of the fact that the MPAs eventually have a larger number of mature animals than surrounding heavily fished areas, and deliberately poach the populations inside the MPA. Fisheries MPAs can also fail if the protected areas are selected without scientific justification, or if the size and / or location of a scientifically valid MPA are compromised, to avoid upsetting interest groups.

Examples of the various reasons for failure of fisheries MPAs include those in the following table:

Table 6: Failure of fisheries MPAs

Reason for MPA Failure	Discussion
Pollution / inadequate protection of critical habitat	Some MPAs for fisheries management fail to protect populations despite the cessation of fishing, because habitat quality in the MPA is low. Samoilys (1988) provided an example of an African reef MPA that failed to enhance biomass and abundance of commercially important species. The failure was caused by (i) excessive siltation of the reefs due to poorly managed catchment activities, leading to high suspended silt loads and greatly reduced water quality; and (ii) dynamite destruction of much of the reef substrate prior to reserve designation, leading to reduced habitat amenity for the reef-dependent fish species. Tegner (1993) showed how inadequate habitat protection led to fisheries decline for Californian abalone species. Despite a 15-year closure of the fishery, there was no recovery of pink (*Haliotis corrugata*) and green (*H. fulgens*) abalone populations which had previously declined concomitantly with sewage-induced loss of the *Macrocystis* kelp canopy, because kelp loss led to food deprivation for the abalone.
Location in marginal habitat	MPAs that are designated in areas that are at the end of a species range may not be successful. For example, fisheries MPAs in Palau did not increase the densities of trochus shell over 20 years, perhaps because they were situated in marginal habitats (Heslinga *et al* 1984, cited by Dugan and Davis 1993). Similarly, MPAs in New Jersey, USA, did not increase hard clam populations, because they were positioned in marginal habitats, rather than in the main spawning area for clams (McCay 1988, cited by Dugan and Davis 1993).

Reason for MPA Failure	**Discussion**
Inappropriate design (size, location) for single species protection/fisheries enhancement	A MPA in Florida that was designated for protection of lobster spawning stock failed to protect the lobsters due to its small size (0.5 square km), coupled with the fact that lobsters in the MPA migrated from the MPA into the fishing grounds every night to feed, whence they were caught by divers and traps (Hunt 1991, cited by Rowley 1994). Armstrong *et al* (1993) reported that a Bering Sea MPA was ineffective in protecting a red king crab stock from the impacts of trawling, and did not increase relative abundance of crabs, because important breeding and hatching grounds and juvenile habitats were not protected by the MPA. Protection was complicated by the fact that (i) spatial centres for spawning varied considerably over the 13-year period of study, and (ii) there was much variation in the suitability of habitat within the area to which juveniles recruited. The most suitable parts of the bay for juvenile recruitment (and consequent fisheries replenishment with crabs) were not protected by the MPA, which did not augur well for the fishery.
Inability of a MPA designed for habitat or multi-species protection to be optimal for single species fisheries enhancement	In the example cited above (Armstrong *et al* 1993), the trawling refuge was originally designated to protect multi-species from becoming by-catch. Although such refuges may help to protect some species that have similar patterns of habitat utilisation, they cannot be optimal for others with different life history traits and habitat requirements. In the case cited above, the MPA could not provide adequate protection for the red king crabs because the oceanographic patterns affecting crab demography (distribution and dispersal patterns of life stages) were not considered when the location for the MPA was chosen. Murray *et al* (1999) also stated that MPAs which are successful in protecting habitat and enhancing some fisheries stocks may be unsuccessful in enhancing stocks of other species (eg abalone, sea urchins) if numbers of predators (eg sea otters, in that case) increase in the MPA.
Combined factors (oceanographic events; MPA design; fishing)	Shepherd (1991) and Shepherd and Brown (1993) discussed the failure of a small MPA established in 1971 at West Island in South Australia to protect the local spawning population of greenlip abalone (*Haliotis laevigata*). That population suffered long-term decline due to (i) an extended period of oceanographically-driven recruitment failure; (ii) exacerbation of the 'natural' causes of population decline due to fishing abalone adjacent to the reserve, where the majority of the local abalone population exist. These factors together reduced the critical numbers and densities of adult abalone for successful spawning and fertilisation. Fishing the population outside the MPA may also have reduced the replenishment of the MPA with recruits. Despite its great success for marine research, for over three decades (see Section 5.2.5) the MPA was considered too small to adequately protect the abalone population from decline, due to reproductive linkages between the abalone inside and outside of the MPA, and the fact that most of the local abalone population existed outside the MPA. However, the reserve was extended in 1992 towards the mainland, and is currently the only MPA of its type in South Australia to ostensibly provide a refuge for a regional 'source' of recruits to replenish the local population.
Socio-political compromise to MPA design (size, location, etc)	Four small 'no-take' reserves established under 'Proposition 132' in California were reportedly placed in areas away from fishing grounds, to avoid conflict with fishers. The MPAs were also too small to be effective, and were not monitored or managed (Wilder, unpublished comment to California Marine Protected Areas Network 1999). Such 'politically compromised MPAs' apparently benefit neither the fishers nor the fish stocks.
Poaching / breaches of management regulations	Goodridge (1996) provided an example of a St Lucian fishery for which 25% of the catch came from illegal fishing inside the local MPA. Recorded biomass of three commercially significant fish families was low in a Seychelles MPA due to poaching (Jennings *et al* 1996), and the structure of the fish community compared with protected areas was considered to be significantly altered by this activity. Lane *et al* (1991) reported that poaching was a problem at a MPA in the Cayman Islands, where many hundreds of large conch shells were removed. The poaching depleted the population of conch in the protected zone to the extent that no differences in abundance between inside and outside the MPA could be detected. Poaching of sea urchins has also been a problem recently in the Cape Rodney-Okakari Point Marine Reserve in New Zealand.

Reason for MPA Failure	Discussion
Transfer of fishing effort	McClanahan and Kaundra-Arana (1996) showed that the establishment of a previously heavily fished area as a 'no-take' MPA failed to replenish the fish populations. The MPA failed because fishers who previously fished the area prior to its closure transferred their effort to the MPA boundary, and consequently total fish biomass in the boundary area was reduced by 10 times. This extra effort may have prevented fish from moving into the MPA, which could thus not build up fish biomass or numbers (in addition to the fact that the MPA had been depleted of fish prior to its designation).

Much has been written recently about MPA design, management and monitoring, to guard against the types of failure described above, particularly for fisheries MPAs that are designed to protect and enhance specific populations of commercially and recreationally valuable fish. Section 6 discusses the many and varied design criteria for 'optimising' the performance of fisheries MPAs, other small no-take MPAs, and large multiple-use MPAs. Management and monitoring strategies to minimise the risks of both poaching and MPA degradation are discussed in Chapters 8 and 9.

11 CONCLUSIONS

During the past decade, MPAs have become increasingly valued as one of the principal means of protecting marine environments and managing marine resources. The benefits of MPAs have been well documented in more than 100 publications, particularly their roles in protecting marine ecosystems and habitats and their component biodiversity, sustaining and enhancing fisheries, managing multiple conflicting uses, and controlling impacts in coastal marine areas. Some indication of their wide acceptance is the fact that over 1300 MPAs of various types have now been established in coastal marine regions throughout the world, and many more are planned, under major national and international programs that have been instigated during the past five years. The recent resurgence in interest in MPAs has coincided with:

- growing awareness of the many environmental, social and economic values of protecting marine ecosystems / habitats and biodiversity
- the mounting list of damages and future threats to coastal marine ecosystems and resources
- the apparent failure of some alternative management measures, particularly traditional approaches to managing fisheries, and sectorial methods of controlling impacts on marine ecosystems
- an increasing public acceptance that marine resources are exhaustible, and also that marine ecosystems are not bottomless waste receptacles
- in some countries, the need to improve the current inadequate system of ‘paper parks’, in which extractive uses, activities and impacts are not limited or managed.

It is evident that MPA planning, design, selection and management strategies have developed during the past decade to meet those challenges, and few countries would now advocate the earlier *ad hoc* approaches to MPA placement. In many countries, there are now fewer ‘single-purpose’ MPAs being declared, such as the preservation of attractive dive sites, or the protection of single fish stocks, unless those single-purpose MPAs form part of a systematic, coordinated network. Although small, ‘single-species’, ‘un-networked’ MPAs still have an important role in protected area planning, particularly in fisheries management, there is now a preference, in most countries that advocate MPAs, for:

- large, multiple use, zoned MPAs, that include smaller, highly protected ‘core’ areas or
- regional networks of highly protected MPAs (often called reserves), without a zoning system for the other parts of the region that are not declared.

Current international trends in MPA design and placement emphasise:

- representation of biogeographic regions and ecosystems / habitats
- the need for network designs (a ‘systems approach’), that consider oceanographic and ecological linkages between regions, and therefore help to protect a mosaic of inter-connected ecosystem types / habitats, and their biota
- multiple purpose MPAs, which can assist the goals of biodiversity conservation and ecosystem / habitat protection over space and time, fisheries enhancement and management, habitat protection against periodic impacts, as well as providing numerous social and economic benefits
- the use of some types of MPA in scientific studies that assist marine conservation and management, such as ‘control sites’ for elucidating the impacts of fishing and other

activities on marine ecosystems; for biodiversity inventory and ecosystem function studies; and for monitoring changes over space and time in biodiversity and habitat quality

- where possible, provision to declare alternative ‘replicated’ sites in areas subject to high levels of human interference and / or natural environmental disturbance.

MPA declaration is no longer uncoordinated, or the request of a single-agency, lobby group or individual. As discussed in Chapter 8, increased public participation in the MPA planning and management process is now seen as critical to MPA success, and is considered particularly important in the management of large, multiple-use MPAs. This participatory approach to both planning and management of MPAs is now being used in North and Central America, and parts of Europe, and Africa. Participants in some of the MPA programs in these countries have included fisheries managers and other managers of economic development, resource user groups, coastal land use and water quality managers, conservation representatives, scientists, indigenous users, recreation / tourism representatives and other ‘stakeholder’ groups. Direct participation of user groups in monitoring multiple-use MPAs is also now considered to be an effective management approach. One example is the involvement of fishers and recreational divers in data collection, which has occurred in the Philippines and parts of North America. Another recent trend in MPA management is the use of fees in multiple-use MPAs, similar to developments in the terrestrial national parks system. The revenue accrued from nominal fees can be used for monitoring and control of impacts, particularly in MPAs with high levels of visitor use.

As part of the consultation and ongoing management processes, the promotion of MPAs through comprehensive public education programs is also becoming more prevalent in some countries that have current plans for new MPA establishment. In New Zealand, for example, this has included public meetings to discuss MPA proposals; discussion and promotion through media (television, radio, Internet and magazines); support for conservation groups; diver surveys; and MPA monitoring programs (Ballantine 1999).

In some aspects of MPA planning, design and establishment, Australia and New Zealand appear to have led the way for the past decade. As discussed in Chapter 4, the inadequacy of previous approaches to MPA planning and designation in Australia, has been recognised for more than 10 years. This resulted in both the Commonwealth’s Ocean Rescue 2000 Program, and its successor, the current National Representative System of Marine Protected Areas strategy and guidelines. The NRSMPA has specified that new MPAs should be developed according to the main principles of:

- comprehensive ecosystem / habitat representation
- adequacy of MPAs in terms of number, design and management regulations, to ensure that ecological processes are protected, and viable populations, species and communities and maintained
- declaring representative examples of biotic diversity and ecosystems / habitats, at defined scales within the MPA network
- protecting special areas and biota (such as species ‘hotspots’; centres of endemism and ‘refugia’; rare, threatened and depleted populations; unusual habitats and ecosystem types; and migratory species or those with complex life cycles).

Under the NRSMPA, maintaining commercial, social and cultural activities is considered to be an important part of Australia’s MPA system, and another major objective of the new systems approach to MPA planning seeks to optimise the multitudinous socio-economic benefits of MPAs. In some States of Australia, several methods are now being considered in the difficult task of simultaneously trying to:

- ensure that the primary goals of the NRSMPA, and the conservation and management goals of both existing and new MPAs, are met by the system that is eventually established
- maximise social and economic benefits of new MPAs
- minimise user group conflicts over space and time.

These methods include: (i) the employment of newer scientific tools that assist multi-objective MPA design (outlined in Chapter 6), in which important socio-economic data sets can also be considered in the design process, and (ii) the provision for greater public participation in MPA decision-making (including selection, placement and management).

Despite the recent developments in MPA planning, design, selection and management that are occurring throughout the world, there is still no consensus about the ‘best’ way of establishing a representative system of MPAs. An example of contrasting approaches is the two methods being advocated in New Zealand and Australia. Government agencies in New Zealand persistently promote the benefits of MPAs, and actively engage the public in decisions about tactical MPA design. The Australian approach that is advocated by the ANZECC Task Force on Marine Protected Areas is more prescriptive, and involves:

- a long-term national strategy (related to the NRSMPA and its Ocean Rescue 2000 precursor)
- major Commonwealth and State-level bioregional and ecosystems mapping and data collection programs, to identify candidate lists of MPAs for a comprehensive, adequate and representative network
- decisions about MPA design being led by government agencies, but with public participation in the final *selection* and exact placement of MPAs, from the government-driven *candidate list* of sites.

For example, in contrast to the MPA *identification* process (which is largely scientific, and reliant upon good quality physical, oceanographic, ecological and biological data), the MPA *selection* process in Queensland utilises ecological, economic, social, cultural and legal and practical criteria, gathered during detailed public consultation. One aim of the selection phase is to determine the ‘least cost’ of alternative MPA siting choices, with ‘cost’ being defined in a variety of social, economic and conservation terms.

It should be recognised that the ecological criteria used for identifying potential MPAs are not independent of those criteria used to finally selected areas from a candiate list. Therefore, the distinction between the identification and selection phases is artifical, in the development of a comprehensivce, adequate and representative network of MPAs.

Regardless of the choice of method used to designate new MPAs, their success in any country hinges upon a long-term ‘whole system’ management approach, as discussed in Chapter 9, to control potentially damaging activities within and outside MPAs, particularly those that will directly or indirectly affect MPA viability, and to minimise detrimental changes to marine ecosystem structure and function.

The declining quality of coastal marine regions throughout the world indicates that the 2000s will present even greater challenges to marine policy-makers, managers, commercial and recreational and cultural user groups, scientists, and conservationists. Those countries that are choosing to adopt representative systems of networked marine protected areas, as a key conservation and management method, are well prepared to meet those challenges.

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APPENDIX 1

MPAs in which benefits for fish stock sustainability, enhancement and management have been demonstrated.

Table 7: Examples of MPAs in which size, range and biomass of commercial and/or recreational fishing species increased.

Reference	Locality	Species	MPA Benefits; Comments
Alcala (1981, cited by Davis, 1989); Alcala (1988, cited by Roberts and Polunin 1993); Russ (1987)	Philippines	Various coral reef fish species, including predatory groupers.	Significantly increased biomass (six to 31 times increase in grouper biomass). Increased survivorship of young fish also noted in MPA.
Wantiez *et al* (1995)	New Caledonia	Serranidae (groupers), Lutjanidae (snappers), Lethrinidae (emperors), Mullidae (goatfishes), Labridae (wrasses), Scaridae(parrotfishes), Siganidae (rabbitfishes), Acanthuridae (surgeonfishes), Chaetodontidae (butterfly fishes).	246% increase in fish biomass in the MPA, compared with fished reference sites. Sampling was done before and after MPA establishment, and also included fished sites for reference.
Roberts and Polunin (1993); Roberts (1995)	Netherland Antilles	Parrotfish, grunt, snapper and grouper families. Four commercially fished families.	Significant increases in biomass on reefs at five metres. Greater biomass and size of fish in MPA, especially predatory snappers (220% increase). Overall biomass of fish from four families increased by 60% over two years of protection.
Roberts and Polunin (1993)	Belize	Various reef species, including schoolmaster snapper (*Lutjanus apodus*).	Substantially higher frequency of very large individuals; greater size range of schoolmaster snapper in MPA than that recorded in three non-MPA areas. MPA included snapper individuals between 30 to 45 cm total length; fish over 30 cm were virtually absent in the areas outside the MPA.
Roberts and Hawkins (1997)	St Lucia (Caribbean)	Snappers (*Lutjanus* spp.), grunts, and parrotfish.	Small MPA of only 2.6 ha protected large individuals of major commercial fish species, including species not recorded anywhere else along the heavily fished coast.
Rakitin and Kramer (1996)	Barbados	Various reef fish species.	Average size of 18 out of 24 fish species was greater in the MPA compared with fished areas, especially for highly ‘trappable’ species.
Sluka *et al* (1996)	Bahamas	Nassau grouper (*Epinephelus striatus*).	Larger size of grouper in the MPA compared with surrounding fished areas.
Bohnsack (1982)	Florida	Sports fish (snapper spp. and grunt spp.).	Larger sizes of sports fish (MPA was protected from spearfishing for two decades).
Davis (1989)	Florida	Spiny lobster.	Greatly increased growth of juvenile lobsters.
Wallace (1997)	British Columbia	Northern abalone (*Haliotis kamschatkana*).	Abalone in unfished areas are larger than those in adjacent fished areas.
Palsson and Pacunski (1995, cited by Kripke and Fujita 1999)	Washington	Various fish (coppers, quillbacks and lingcod).	Larger fish in MPA.
Roberts and Polunin (1993)	Egypt	Seven species (from the serranid, scarid and acanthurid families).	Significant increases in fish size; individuals of one grouper species on average three times heavier than grouper on unprotected reefs. Reserve has been closed to fishing for around 15 years.

Reference	Locality	Species	MPA Benefits; Comments
Bell (1983)	Mediterranean	Eighteen species of reef fish (including several sparids and serranids).	Increased mean size range for 18 species. Greater number of medium and large individuals of two commercial species (*Dilodus sargus* and *D. vulgaris*). Fish study based upon species that were vulnerable to commercial netting, recreational angling and spearfishing outside the reserve.
Francour (1992 and 1996)	Corsica (Italy)	Various fish species.	Increased numbers of mature adult fish. Increased densities and biomass of fish in rocky reef habitats. MPA studied between 1988 and 1992.
Vacchi *et al* (1998)	Ustica Island (Italy)	Large reef fish: *Epinephelus marginatus, Labrus merula, L. viridis.*	Increased size of species vulnerable to over-exploitation. Size was positively correlated with the degree of protection of the area, ranging from largest in integral reserve with complete protection, to smallest in general reserve and partial reserve.
Jennings *et al* (1995)	Seychelles	Large reef fish in the Lutjanidae, Lethrinidae families, and other reef fish in fish-eating, invertebrate-eating and plant-eating groups.	Significantly higher total biomass of all major groups of fish in the MPA compared with heavily fished areas.
Jennings *et al* (1996)	Seychelles	Sixteen families (115 species) of reef fish, including grunts, wrasses, snappers, surgeonfishes, coral breams, parrotfishes, groupers, rabbitfishes, goatfishes, and others.	Biomass of total fish community, and biomass of many families greater in the protected MPAs compared with fished sites and unprotected MPAs (in which poaching occurred). The two protected MPAs were closer to the fished area than to each other, which strengthened the conclusion that the protected MPAs were actually effective, and that the results were not simply due to spatial factors.
Letourneur (1996)	Mayotte Island, East Africa	Large carnivorous and semi-pelagic fish species.	Increased diversity, biomass and abundance of large carnivorous and semi-pelagic fish in the MPA compared with fished areas.
McClanahan *et al* (1999)	Kenya and Tanzania, Africa	Angelfish, butterflyfish, parrotfish, surgeonfish, triggerfish and scavenger fish groups.	Biomass of fish in protected MPAs was 3.5 times higher than in fished areas.
Buxton and Smale (1989)	South Africa	reef fish in the Sparidae family.	Fish within MPA were larger on average, and maximum size was greater.

Table 8: Examples of MPAs in which abundance and/or density of species increased, to rebuild depleted populations

Reference	Locality	Species	MPA Benefits
Alcala (1981, cited by Davis, 1989); Alcala (1988, cited by Roberts & Polunin, 1993) Russ (1987), Russ *et al* (1992) Rigney (1990)	Philippines	Various coral reef fish species, including groupers and caesionaids.	Significant increases in abundance and density of reef fish species. Density of caesionaids, the major fisheries species at Sumilon, was almost twice as high inside as outside the MPA during a 1983 survey (Russ *et al* 1992). Periodic 'export' of fish out of the longest-running Philippines MPA and into the adjacent fished area occurred (see Table 9). Three MPAs showed significant increase in abundance after only one year of protection. Increased number of large predatory fish species was recorded in one MPA, and their abundance rose by over 3.5 times during five years. Highest abundance was recorded for 'favourite' target species such as grouper (Rigney 1990).
Wantiez *et al* (1995)	New Caledonia	Serranidae (groupers), Lutjanidae (snappers), Lethrinidae (emperors), Mullidae (goatfishes), Labridae (wrasses), Scaridae(parrotfishes), Siganidae (rabbitfishes) Acanthuridae (surgeonfishes) Chaetodontidae (butterfly fishes).	160% increase in fish density in the MPA, compared with fished reference sites. Sampling was done before and after MPA establishment, and also included fished sites for reference.
Rakitin and Kramer (1996)	Barbados	Various reef fish species.	Abundance of all fish was greater in the MPA, especially 'trappable' species.
Polunin and Roberts (1992)	Caribbean	Commercial grunt, snapper and grouper spp.; other reef fishes.	20 to 33% increase in abundance of fishery species (grunts, snappers, groupers; 8 to 24% increase in non-target reef fish species.
Roberts and Polunin (1993)	Belize	Spiny lobster (*Panulirus argus*); conch (*Strombus* sp.).	Spiny lobster densities up to 25 times higher inside the MPA; substantially higher conch densities.
Sluka *et al* (1994 and 1996)	Bahamas	Nassau grouper (*Epinephelus striatus*); other grouper species.	Greater abundance of grouper in the MPA compared with fished areas. Densities of grouper were three times higher in the MPA compared with outside areas.
Roberts (1995)	Netherland Antilles	Three commercially fished families (snappers, parrotfish, surgeonfish).	Greater abundance of fish in MPA, especially large predatory species.
Paddack (1996, cited by Kripke and Fujita 1999)	California, USA	Rockfish species.	More abundant rockfish in two MPAs less than 1.5 square km.

Reference	Locality	Species	MPA Benefits
Haskell (2000)	Florida, USA	Spiny lobster (*Panulirus argus*); yellowtail snapper, hogfish, grouper species.	Significantly increased numbers of lobsters and reef fish after two years of protection in MPAs, compared with fished reference sites.
Paddack (1998)	Monterey Bay, USA	Ten species of reef fish (mostly in genus *Sebastes*).	Slightly higher numbers of all fish species noted in MPA, and significant increases in average length and length frequency of fish in two of the three MPAs.
Davis (1989)	Florida, USA	Spiny lobster (*Panulirus argus).*	Increase in lobster numbers adjacent to the MPA.
Bohnsack (1982, cited by Roberts and Polunin, 1993); Clark *et al* (1989)	Florida, USA	Snapper and grunt spp..	Increased densities of snappers and grunts. Abundance of snappers and grunts increased by 93% and 439% respectively on protected reefs.
Johnson *et al* (1999)	Florida, USA	'Sports fish'.	Greater abundance of fish in MPA compared with adjacent fished areas.
Palsson and Pacunski (1995, cited by Kripke and Fujita 1999)	Washington	Various fish (coppers, quillbacks and lingcod).	Almost double the number of species in the MPA, compared with fished areas outside; number of lingcod and lingcod 'nests' nearly three times higher in MPA.
Bell (1983)	Mediterranean	Sparidae family: (five spp. of *Diplodus*, *Oblada*, *Spondyliosoma* & *Sarpa* spp.), Labridae family: (two spp. of *Labrus*; *Symphodus* sp.), two serranids (*Dicentrachus* & *Serranus* spp.), two scorpaenids (*Scorpaena porcus* and *S. scrofa*), *Sciaena*, *Mullus*, & *Mugil* sp.; Non-commercial pelagic school fishes *Boops*, *Spicara* & *Chromis* sp..	Increased densities of 18 species of fish that were vulnerable to commercial netting, recreational angling and spearfishing. Higher densities of non-commercial species also recorded. (The census technique ensured that cryptic behaviour and 'diver-avoidance' of reef fish were accounted for, ie surveyors sampled every cave and overhang carefully, so as not to underestimate sizes, densities or abundances of fish due to increased wariness of fish at non-MPA sites.)
Francour (1997, cited by Kripke and Fujita 1999)	France	Fish species in seagrass beds.	Number of species nearly twice as high in MPA; abundance of fish five times greater in MPA compared with fished area.
Vacchi *et al* (1998)	Ustica Island (Italy)	*Epinephelus marginatus*, *Labrus merula*, *L. viridis.*	Increased abundance of species vulnerable to over-exploitation. Abundance was positively correlated with the degree of protection of the area.
Buxton and Smale 1989	South Africa	Reef fish in the Sparidae family.	Fish of two major species were significantly more abundant (from four to 13 times) within the MPA than in outside areas.

Table 9: Examples of MPAs which indirectly improved fisheries by protecting spawning stock biomass (SSB) and/or spawning events.

Reference	Locality	Species	MPA Benefits
McManus (1988)	Philippines	Various coral reef fish species.	One of the MPAs improved spawning stock biomass over 10 years of operation (see Table 9 for evidence).
Davis (1995)	Chile	Edible ascidian (*Pyura chilensis*).	37% of ascidians reached sexual maturity in the MPA, compared with 6% outside the MPA. Low densities of adults outside the MPA were considered to affect reproductive success. Densities of three orders of magnitude higher were also noted in the MPA, as well as significantly increased sizes of ascidians.
Lane, Kitt, Ebanks-Petrie and Bush (1991)	Cayman Islands	Shallow water conch (*Strombus gigas*).	MPAs for spawning stock replenishment contained greater numbers of spawning-sized conch compared with the adjacent fished area. The study was undertaken five years after reserve establishment.
Barnes, Ward and Burnett-Herkes (1991)	Bermuda	Red hind fish (*Epinephelus guttatus.*	Two discrete spawning populations of red hind have been protected since 1972 through the use of seasonal closures (temporary MPAs). Bermudan refuges are believed to be of ‘significant value’ to the management of the commercial red hind species.
Sluka *et al* (1996)	Bahamas	Nassau grouper (*Epinephelus striatus).*	Reproductive output of grouper is protected by the MPA.
Chiappone and Sullivan Sealey (1998)	Central Bahamas	Reef fish species.	MPA maintained high spawning stock biomass relative to fished areas.
Stoner *et al* (1994)	Central Bahamas	Queen conch (*Strombus gigas*).	Greater larval densities of conch in the MPA (highest ever recorded in the wider Caribbean), due to greater densities of conch in the MPA compared with outside areas.
Martell (1998, cited by Kripke and Fujita 1999)	Vancouver, USA	Lingcod.	Greater spawning of lingcod in the MPA compared with fished areas outside.
Davis and Dodrill (1980, 1989)	Florida, USA	Spiny lobster (*Panulirus argus*).	Researchers consider that the small protected area of lobster had a greater reproductive output than the larger, fished area, because the lobsters in the MPA included a high proportion of large, reproductively active females, compared with fewer, smaller ones in the fished areas.
Harmelin *et al* (1995)	France	16 species of large carnivorous fish; two species of serranid and labrid fish.	MPA acted as a reservoir for large spawning adults, and a refuge for a labrid fish species that was induced by fishing to change sex earlier in life.
Zabola *et al* (1997)	Spain	Dusky grouper (*Epinephelus marginatus*).	MPA enables critical densities of undisturbed grouper to survive, so that adults could engage in the complex socio-behavioural ritual that leads to spawning success.

Table 10: MPAs which are either known or assumed to have 'exported' larvae, juveniles or adult fish to adjacent fished areas, or fisheries in the region.

Reference	Locality	Species	MPA Benefits
McManus 1988; Russ and Alcala (1989); Alcala and Russ (1990); Russ and Alcala (1996b)	Philippines	Various commercial coral reef fish species.	Strong evidence for role of MPAs in exporting biomass to fishery: densities of snapper and emperors declined by 94% outside reserve after MPA protection broke down (10 years after establishment) and catches declined by 50%, (compared with situation during 10 years of MPA operation, when catches and densities significantly increased, ie catches near MPA were approx. 50% higher than in unprotected areas when MPA was intact. One of the MPAs improved spawning stock biomass over 10 years of operation, evidenced by (i) provision of reef fish recruits to surrounding areas, and sustained fisheries during the life of the MPA; and (ii) decline in catches and catch rates when protection of the MPA broke down. Within two years of MPA establishment, mean harvest rate in adjacent fishing area had tripled, and catch per unit area over a several year period was one of the highest reported for any coral reef in the world (Davis, 1989). Densities of large predatory fish decreased significantly twice when the MPA was opened to fishing, and increased again during all periods of closure.
Russ *et al* (1992)	Philippines	Various coral reef fish species.	MPAs for reef fish can protect residual spawning stocks, thus potentially supplying recruits to a broad fished area, because larvae of many reef fishes disperse over tens to hundreds of kilometres (Doherty and Williams, 1988, cited by Russ *et al*, 1992).
Holland *et al* (1993)	Hawaii	White goatfish (*Mulloides flavolineatus*).	Provision of eggs & larvae to reefs areas adjacent to MPA, via regular spawning migrations of goatfish away from the MPA, and subsequent return to protected home range.
Yamasaki and Kuwahara (1989, cited by Kripke and Fujita 1999)	Japan	Zuwai crab.	46% increase in catch rates in areas adjacent to MPAs, after five years of protection.
Tegner (1992)	California, USA	Green abalone (*Haliotis fulgens*).	A 'defacto' MPA (closed area) into which brood stock were transplanted, was assumed to have provided recruits to the surrounding depleted areas, based upon surveys before and four years after the transplant. Recruitment of abalone in the fished areas was greater after the MPA was established.
Schlining (1999) cited by Kripke and Fujita (1999)	California, USA	Shrimp (prawn).	Median catch-per-unit-effort (CPUE) close to MPA significantly greater than median CPUE far from MPA.

Reference	Locality	Species	MPA Benefits
Davis and Dodrill (1980), cited by Rowley (1994); Davis and Dodrill (1989), cited by Roberts and Polunin, (1993); Davis (1989)	Florida, USA	Spiny lobster (*Panulirus argus*).	Almost all adult spiny lobsters are removed from Florida reefs every year, and local MPAs appear to have sustained the fishery despite such intense harvesting, by protecting a portion of the SSB. Tagging studies have demonstrated movement of spiny lobster out of a Florida MPA and into the fishery. MPA is assumed to provide larvae and recruits to both adjacent and distant fishing zones.
Bohnsack (1995 and pers. comm. to Californian Marine Protected Areas Network)	Florida, USA	Stone crab, spiny lobster.	Closed areas in two national parks and one sanctuary have helped to improve fisheries yields of stone crabs and lobsters, and yields were highest recorded after the closed areas were established.
Johnson *et al* (1999)	Florida, USA	Sports fish.	Tagging studies showed that ‘important sports fish’ species moved out of the MPA into the adjacent fishery.
Beets and Frielander (1992 and 1999)	Virgin Islands, USA	Red hind fish (*Epinephelus guttatus*).	Spawning aggregations of red hind were replenished by the MPA. Spawning aggregations had been seriously depleted prior to the MPA during the 1980s and large fish were overfished. MPA helped to re-establish age structure and spawning potential in the population, and revived the fishery.
Sluka *et al* (1996)	Bahamas	Nassau grouper (*Epinephelus striatus).*	Some groupers migrated out of the MPA into the fished area.
Bennett and Atwood (1991) Atwood and Bennett (1994)	South Africa	Surf zone fish spp. such as *Coracinus capensis*, *Diplodus* spp., and *Lithognathus lithognathus*).	Increased catch rates of six species of rock- and surf- zone fish adjacent to MPA. Catch rates of two species near the MPA were four to five times higher than catches in non-MPA areas. Catch rates of four species were 30-60% of unexploited levels after two to four years of MPA establishment. Study of catch rates around reserve ran for six years. Increased catch rates attributed to MPA protection, and were observed a few years after the MPA was declared. Fish move out of the MPA into the fishery, replenishing yields.
Buxton (1993b)	South Africa	Two species of slow-growing, long-lived, sex-changing reef fish.	Growth rates of one species outside MPA were significantly slower compared with those inside the MPA. Sex ratio was less skewed inside the MPA. MPAs considered to protect fish from recruitment failure, by maintaining population size structure and sex ratio.
Tilney *et al* (1996)	South Africa	Larvae of commercially and recreationally significant Sparidae family fishes.	Field sampling studies of fish larval movement suggest that Sparid fishes are ‘exported’ of the MPA into the fished area.

APPENDIX 2

Examples of coastal marine ecosystem types that benefit from protection by MPAs, and summary of their ecosystem functions

Examples of coastal marine ecosystem types that benefit from protection by MPAs, and summary of their ecosystem functions

At a national level, ecologically important estuarine saltmarsh-mangrove complexes have been better represented in the existing network of Australian MPAs than some other significant nearshore marine ecosystem types. Examples include the aquatic reserves of the upper gulfs in South Australia, and the numerous fish sanctuaries that have been declared along the Queensland coast. Young and Glaister (1992) and Edyvane (1994) summarised the literature discussing the ecological importance of saltmarsh and mangrove ecosystems. Such ecosystems have a major role in protecting juveniles of many coastal species, including economically significant fish and crustacean species. Saltmarshes provide food sources, such as marsh crustaceans, and shelter from predation. Mudflats and saltmarsh are periodically covered with waters which are too shallow for larger predatory fish to survive in, thus providing a refuge for juvenile fish. Morton *et al* (1987, cited by Fairweather, 1991) have demonstrated that saltmarsh-mangrove creeks aid fish recruitment by supporting large numbers of juvenile fish. Robertson and Duke (1987, cited by Young and Glaister, 1992) showed that densities of fish and prawns in Queensland mangrove habitats are an order of magnitude greater than in the adjacent nearshore habitats.

Compared with mangrove ecosystems and tropical coral reefs, seagrass and macroalgal-dominated coastal ecosystems have been poorly represented in Australian MPAs, particularly in the southern States where those ecosystems have major physical, chemical and ecological functions. The following summarises the major functions of seagrass beds and macroalgal-dominated coastal reefs.

Nutrient recycling: Macroalgae and seagrasses that are dislodged by wave action, and deposited as large accumulations of 'beach wrack' along southern Australian surf zones and beaches, become an important source of nitrates, phosphates, silicates and other dissolved nutrients essential for continued growth of biota in nearshore marine ecosystems. When broken down by wave action, invertebrates and microbes, the beach wrack is a major recycling route for carbon, essential for most marine life forms. Researchers in Western Australia have estimated that as much as a quarter of the yearly energy production in coastal ecosystems passes through the surf and beach zones as beach wrack, and is recycled back into the system (Robertson and Hansen 1982). Seagrass and algae (along with phytoplankton) oxygenate marine waters through photosynthesis, thus providing optimum living conditions for fauna. Even uprooted macroalgae, such as *Ecklonia*, either in the surf zone or drifting at sea, can photosynthesise and oxygenate the water for several weeks, whilst it is slowly broken down and recycled (Robertson and Hansen 1982).

Coastal stability and erosion prevention: The rhizomes and roots of seagrasses help to stabilise coastal sand and other sediments, thus helping to prevent erosion due to scouring. Another major protective role of seagrasses and macroalgae is absorption and deflection of wave energy, which also protects beaches from erosion. By trapping sediments and also taking in nutrients for growth, seagrasses and macroalgae act as 'bio-filters', helping to keep coastal waters clean. If seagrass dies (from the effects of pollution-induced eutrophication, for example) or is physically removed, sediments are suspended, clouding coastal waters, which affects plant photosynthesis and disturbs the physiological functions of filter feeders and fish. Some algae are even involved in sand production, such as the corallines *Metagoniolithon, Amphiroa, Haliptilon*, which are significant sources of calcium carbonate, thus contributing to the production of coastal sea floor sediments when old and worn plants break down.

Biodiversity promotion: The presence of either seagrass or macroalgae can maintain and promote benthic faunal diversity and abundance, by providing space for fish and invertebrates to settle, feed, seek shelter from currents and wave action, and escape predation.

Macroalgal beds also provide microhabitats for a variety of fauna through layering of the vegetation, often comprising both an understorey and overstorey of algae of different size, shape and texture. When the topographic complexity of a habitat is increased by 'seaweed strata', different-sized fish and invertebrates with spatially-separate feeding preferences can co-exist in the same habitat. Russell (1977) showed that the type, height and density of algal cover were some of the important features affecting the distribution and abundance of 'resident' reef fishes in a New Zealand reef ecosystem. Areas of extensive algal cover supported 'standing crop' estimates of reef fish that were as high as those recorded for tropical coral reefs. Other studies since that time have also emphasised the role of seagrass and macroalgal 'architecture' in the distribution, abundance and diversity of fish and benthic invertebrates.

Large brown algae (such as *Scytothalia, Ecklonia, Phyllospora, Cystophora,* and *Sargassum*) and thicker seagrasses such as *Amphibolis* and *Posidonia*, provide 'microhabitats' for bryozoans, hydroids, sponges, tubeworms, and many other species, which live on the blades, around the base of the plants, or even under the holdfasts or rhizomes. It would be harder for the larvae of many of these invertebrate species to find a settling place if the seagrass or macroalgae weren't available to them. Faunal diversity can be higher in the presence of marine plants compared with unvegetated habitats. In Victorian waters, for example, Edgar *et al* (1993) collected significantly more species from seagrass in Westernport Bay than from bare sand, and calculated more than twice the total production of small fishes from seagrass compared with unvegetated substrate. Biodiversity of fish can be adversely affected by the loss of seagrass and macroalgal stands. For example, loss of kelp beds can reduce fish abundance, including that of ocean-going pelagic species, because juveniles of many fish species use seaweed to shelter and feed in (Lehtinen *et al* 1988, cited by Fairweather 1991). Recently, Jenkins *et al* (1996) showed that species richness and abundance of fish, including that of commercial species, were much higher in macroalgal reef habitat than on bare sand. Macroalgal reef beds have been described as 'keystone' habitats, which provide conditions for continued recruitment of fish and invertebrates (Fairweather 1991). Fish species typical of subtidal rocky reefs, outcrops and headlands in South Australia include the magpie perch (*Cheilodactylus nigripes*), beardie (*Locella rhacinus*), drummer (*Kyphosus sydneyanus*), roughy (*Trachichthys australis*), dusky morwong (*Dactylophora forsteri*) and gurnard perch (*Neosebastes pandus*) (Glover 1979; Johnson 1988a; Hutchins and Swainston 1986). Similarly, it is now commonly known that loss of seagrass beds can adversely affect fisheries, as shown by Jenkins *et al* (1993), who associated the declining catches of whiting and leatherjacket fish in Victoria with seagrass loss.

Even dead seagrass and algae contribute to coastal marine biodiversity, by providing micro-habitats for coastal and marine invertebrate species (such as micro-crustaceans, worms and insects, among other fauna), that live in beach-washed piles of seaweed and seagrass. Such beachwrack-dwelling fauna assists in maintaining other levels of biodiversity, through their role as food sources for other coastal species, such as wading birds, sea birds and crabs.

Direct food sources: Seagrass beds and macroalgal forests provide both direct food sources (for herbivorous juvenile and adult fish, mollusks and other fauna) and indirect food, by providing a home for numerous small crustaceans other invertebrates, many of which are consumed by fish. The diversity of sizes, shapes and densities of marine algae and seagrass provide a variety of foraging spaces**.** Some marine animals feed directly on green, brown or red algae, such as southern damselfish, southern cale and sweep species, blackfish, zebrafish and seacarp, various mollusks and sea urchins (Jones 1988; Jones and Andrew 1990). Juvenile abalone eat coralline algae, and adults often consume drifting red macroalgae and seagrasses with epiphytes (Shepherd 1973, 1988; Shepherd and Cannon 1988). Some species of drummer, bream, garfish, mullet, weedy whiting and leatherjacket families eat, among other dietary items, eelgrass and other seagrasses, algal epiphytes on seagrasses, green algae and other macroalgae (Robertson and Klumpp 1983; Klumpp *et al* 1989). There are also numerous crustacean herbivores, such as amphipods, many of which eat filamentous algae; some isopod species, which eat both seagrass and large branched algae; the crab *Pugettia* sp., which grazes on *Zostera* (eelgrass), and the rock crab *Nectocarcinus*, which eats *Posidonia* (tapeweed) as part of a mixed diet, (the crab in turn being a major food item for a flathead species). The diet of juvenile rock lobsters in Western Australia comprises around 10% seagrass, and adult rock lobsters eat coralline algae to supplement their diet of seagrass epifauna (Klumpp *et al* 1989).

Indirect food sources: A number of reef and seagrass-dwelling fauna rely upon marine plants for their epibiotic food sources, feeding on the small crustaceans, hydroids, mollusks and worms which live on, under or around algae and seagrass. The blades of seagrasses and the fronds, stems and rhizomes of macroalgae provide additional attachment space for epifauna and small epiphytes, which are utilised as major food sources for many animals. Some fish species eat the amphipods which cling to foliose red algae, such as *Gelidium* (Holbrook *et al* 1990), *Plocamium* and *Asparagopsis*, or the blades and holdfasts of brown macroalgae (*Ecklonia, Cystophora, Sargassum,* for example). Micro-crustaceans (amphipods, copepods, isopods) living in macroalgal stands and on coralline algal turf flats are the primary food source for juveniles of many reef species, such as snapper, goatfish, southern wrasses, leatherjackets, kelpfish and numerous species of leatherjackets.

By attenuating currents and creating physical barriers to particle transport, seagrass and macroalgal stands can also trap mobile food sources (including larvae) for animals living in the stands of vegetation. High correlations have been found between small fish production and marine plant epifaunal production (particularly small crustaceans). The trophic role of seagrasses is further supported by the death, or emigration away from seagrasses, of small fishes in the season when epifaunal abundance in seagrass is lowest (Edgar *et al* 1993). It has been estimated that decline in seagrass abundance in Westernport Bay has an associated loss of around 2500 tonnes of epifaunal food sources which would have been attached to the seagrass in unpolluted conditions (Edgar *et al* 1993).

Lower down the food chain, shrimps, micro-crustaceans and small marine snails often feed on the ‘periphyton’ mixture of diatoms, encrusting microalgae and coralline algae, bryozoans, bacteria and fungi that is often found growing on seagrass blades (Klumpp *et al* 1989). The micro-crustaceans and epifauna associated with coastal seagrass beds are utilised by many different coastal marine animals, including decapod crustaceans such as lobsters, and economically important fish species, such as juvenile King George whiting, yellow-fin whiting, and garfish in South Australia (Jones 1984; Jones *et al* 1990), and blue rock whiting and six-spined leather-jacket in Victoria (Bell and Pollard 1989), the latter two species of which were considered to be adversely affected by loss of seagrass beds in Westernport Bay (Jenkins *et al* 1993). Thin seagrasses such as *Zostera* and *Heterozostera* (eelgrass) are utilised by garfish, young King George whiting (*Sillaginodes punctata*) and other commercial species (Jones 1984). Connolly (1994a) showed that in a South Australian seagrass estuary, more species and more individuals of small fish, including juvenile King George whiting, gathered over eelgrass than in bare sandy areas, and his removal experiments (Connolly 1994b) supported the notion that young fish primarily use seagrass estuaries for epifaunal feeding.

Different marine plant associations attract different species, which highlights the importance of maintaining macroalgal and seagrass diversity in nearshore marine systems. A variety of habitats within any one reef ecosystem serve the various life stages of numerous fish species, which have different foraging and sheltering requirements served by specific habitat types within the one reef ecosystem. The presence of both eelgrass (*Zostera*) and tapeweed (*Posidonia*) in an estuary, for example, is considered to increase the availability of habitat for species which prefer one type of seagrass over another for feeding or shelter (Bell and Pollard 1989; Ferrell *et al* 1993). Different life stages of the same fish species can require specific types of macroalgal habitat during their development. For example, the flat, coralline algal beds in northern New Zealand are an important feeding area for early juvenile snapper, whilst older juvenile snapper feed in *Ecklonia* forest rather than in the coralline turf (Kingett and Choat 1981). Closely related fish species may each have a special algal habitat feeding preference—work in North America, for example, has shown that three species of surfperch each feed separately on the invertebrates found in kelp beds, red algal beds, and in coralline turf (Holbrook *et al* 1990). Work in New Zealand showed strong positive correlations between the density and extent of macroalgal cover and abundance of juvenile snapper and wrasse (Kingett and Choat 1981; Jones 1984), and when the preferred macroalgae was removed from the habitats, fish abundances declined. In southern Australia and New Zealand, differences in fish species compositions between various southern reefs have been attributed to differences in food availability on the reefs, directly correlated with the kinds of macroalgae growing on the substrate. Choat and Ayling (1987) for example, showed that small labrid fish species that eat crustaceans do not venture away from dense macroalgal stands in which their preferred food sources are available.

Even the beachwrack deposits of uprooted macroalgae and seagrass detritus are important food sources, providing habitat for large densities of shrimp and other small crustaceans, such as amphipods, isopods and copepods. Some of these micro-crustacea live on filamentous red algae beachwrack, and feed directly on brown macroalgal beachwrack blades (such as *Ecklonia radiata*), helping to break the algae down for recycling (Robertson and Hansen 1982). Bacteria also break down the plant detritus, and the detritus provides a major food source for various mysids, shrimp species, polychaete worms and crabs (Klumpp *et al* 1989), which are in turn eaten by nearshore fish, seabirds and wading birds (GBRMPA 1995). Detritus-dwelling crustaceans can form a large part (up to 75%) of the diet of nearshore fish species which are associated with macroalgae and seagrass accumulations, such as yellow-eye mullet (*Aldrichetta forsteri*), school whiting (*Sillago bassensis*), Australian herring (*Arripis georgiana*) and cobbler (*Cnidoglanis* sp.) (Lenanton 1982; Robertson and Hansen 1982). Seagrass detritus in sandy areas has been shown to increase the abundance of prey species for coastal benthic feeders such as flounder, thus resulting in higher feeding rates and growth rates for those fish compared with areas which have no seagrass detritus (Jenkins *et al* 1993), and similar ‘flow on’ effects may also occur in the case of King George whiting growth (Jenkins *et al* 1992). Some fish, such as species of juvenile mullet, pipefish and gobies even eat the detritus itself, as a dietary supplement (Klumpp *et al* 1989).

Habitat for reproduction: For many nearshore species, macroalgal forests and seagrass beds are the centres for courtship behaviours, mating and egg laying and development. Male scalyfin (*Parma victoriae*), for example, maintain and ‘weed’ and fiercely defend their own patch of algae, for courting, mating and nest building (Jones and Andrew 1990; Kuiter 1996). In New South Wales, herring cale (*Odax cyanomelas*) aggregate prior to spawning every year in the same territorial patches of *Ecklonia* kelp, their preferred food species (Andrew and Jones 1990). Squids and many other mollusks lay their eggs in marine vegetation patches, often attaching them to the seagrass or macroalgal blades. By attenuating currents and creating local eddies, seagrasses and macroalgal stands act as ‘vertical traps’, assisting in localised recruitment of larvae and preventing some species of benthic larvae from being transported away from suitable habitat. Alcala (1988, cited by Roberts and Polunin, 1993) reported that the Philippines MPAs protected reef habitat which was suitable or obligatory for recruitment of various reef fish species.

Nursery and sheltering functions: The protective ‘nursery’ function of seagrass beds has been widely reported during the past decade. Pollard (1984), Jones (1984) and Bell and Pollard

(1989) summarised the importance of seagrass as a ‘nursery’ habitat for a wide range of commercial and recreationally significant fish and macro-invertebrates. Seagrass beds, especially those in estuarine areas, may provide shelter for young fish that rest on the seagrass leaves, under the canopy, or on the sediment under the grasses. Apart from protecting coastal marine biota from predation, shallow-water seagrasses can provide refuge for small vulnerable fish fry against currents, storms and other rough conditions.

Seagrass beds in estuaries are particularly important for juvenile fish newly settled out of the plankton, because (i) other suitable forms of shelter are usually lacking in the vicinity of estuaries; (ii) the newly settled larvae that reach the estuary are usually too young to move out and seek alternative shelter, and (iii) many species which are not cryptic on sand require the cover of seagrasses so they can escape predation (Bell and Pollard 1989). Jenkins *et al* (1993) showed that three commercial species of leatherjacket, as well as rock flathead, were largely restricted to eelgrass habit, and that seagrass beds were directly utilised as a habitat by other juvenile commercial fish.

The size, shape and spatial arrangement of seagrass beds can influence their nursery function. Characteristics such as seagrass blade length, shoot density, biomass and ‘patchiness’ of seagrass habitats, have been shown to affect survival of seagrass-associated fauna, as well as influence species composition (Worthington *et al* 1992; Irlandi *et al* 1995). Seagrass workers in the US have shown that predation rate upon crabs and large mollusks are higher in bare and sparsely vegetated areas, compared with denser seagrass beds (Heck *et al* 1987; Irlandi *et al* 1995).

In seagrass beds that act as ‘nurseries’, it is difficult to disentangle the role of seagrass as predator refuge compared with food source, and it remains a popular topic for marine experimental scientists.

Less well known is the similar ‘nursery’ function played by shallow water macroalgal beds. Jenkins *et al* (1996) showed that in Victoria, shallow reef algae was as important as seagrass for providing larval and juvenile fish habitat, for both commercial and non-commercial species. The type, height and density of large canopy-forming algae (such *as Macrocystis, Ecklonia, Cystophora, Sargassum, Seirococcus and Scytothalia*) have a modifying effect on the reef environment, protecting fish from excessive wave action, surge and strong currents; and creating hiding space for fish to escape predation. Some juvenile fish will only reproduce on their ‘home’ reefs, and others will not move away from the shelter of dense seagrass and macroalgal stands, perhaps due to the higher likelihood of being preyed upon if they traverse bare sandy areas in which they are not camouflaged.

Apart from juvenile fish and crustaceans, cryptically-coloured animals and slow moving species are also provided with protection from predators in seagrass and macroalgal beds.

Some of the southern fish species which use seagrass and macroalgal reef beds for protection include:

- juveniles of King George whiting, yellow-fin whiting, weedy whiting, yellow-eye mullet, blue swimmer crabs and western King prawns (Jones 1984; Connolly 1994a), which are more commonly observed in estuaries containing eelgrass and other seagrasses than in unvegetated habitats
- juveniles of the eastern and western blue groper, weed whitings, weedfish, rainbow cale and blennies, all of which have camouflage ability to suit their seagrass or macroalgal habitats
- juvenile sea trumpeters and striped trumpeters, which live in seagrass estuaries and often shelter under floating vegetation (Kuiter 1996)

- the unusual red velvet fish and pretty polly wrasse, which shelter in macroalgal beds
- the grass flathead, which sleeps under macroalgae and seagrass, and the bright green Brownfield's wrasse, which schools over seagrass beds (Kuiter 1996)
- the pygmy leatherjacket, which often anchors itself by biting into macroalgae (Kuiter 1996)
- short-headed and big-bellied seahorses, which often shelter among *Sargassum* plants and kelps (Kuiter 1996), and seadragons, which sometimes camouflage themselves next to macroalgal and seagrass beds
- various pipefish species, some of which live in seagrass beds (and are camouflaged to resemble a blade of seagrass) and other pipefishes whose thin tortuous bodies are camouflaged as algae or seagrass 'roots'.

Other southern Australian fish that commonly shelter in macroalgae include leatherjackets, juvenile strongfish (dusky morwongs), southern wrasse species, and juveniles of 'old wife' fish (*Enoplosus armatus*) which, unlike the black and white adults, are well camouflaged in *Cystophora* and *Sargassum* stands.

Ecological function of surface drift plants: Predator refuges are especially important for juveniles, and even surface drift macroalgae and seagrass can help to protect post-larval and early juveniles from predators and wave-battering, and provide indirect food sources. Lenanton *et al* (1982, cited by Kailola 1993) reported that western school whiting (*Sillago bassensis*) juveniles are often associated with accumulated beachwrack seaweeds in the surf zone, because the seaweed provides shelter and a suitable habitat for prey species. Numerous other reef fish species have near-surface dwelling juveniles that utilise drift macroalgae as 'transport' and shelter prior to settling on reefs. Juvenile trumpeters, wrasses, leatherjackets, drummers and scads are some of the species that seek shelter in floating seaweeds (Kuiter 1996).

Socio-economic function in supporting fisheries: A large number of studies have demonstrated the importance of estuarine and marine seagrass function, structure, location and type in providing food and shelter for numerous commercially and recreationally valuable fish species. The critical link between seagrass habitats and commercial fisheries in Southern Australia have been well demonstrated by Jenkins *et al* 1992, 1993; Edgar *et al* (1993); and Connolly 1994a). In South Australia, Jones (1984) has demonstrated the importance of the seagrass estuary at Barker Inlet as a nursery ground for major economic species in this State such as King George whiting (*Sillaginodes punctata*), western king prawn (*Penaeus latisulcatus*), and yellow-fin whiting (*Sillago schomburgkii*). More recently, Connolly (1994a) showed that more species and more individuals, including juvenile King George whiting, reside over eelgrass (*Zostera muelleri*) beds in seagrass estuaries, compared with bare areas. A distinction should be made between estuarine and non-estuarine seagrass beds in terms of their ecological function for fish populations. In parts of southern Australia, seagrass estuaries (eg *Zostera*, *Heterozostera*) are mainly utilised by the juvenile stage of many coastal fish species, whilst deeper marine seagrass beds (eg *Posidonia*, *Amphibolis*), which are often interspersed with bare sandy areas, are mainly utilised by the adults of various species, such as garfish, flathead and yellowfin whiting (Robertson and Klumpp 1983; Jones *et al* 1990). Seagrass detritus in sandy areas has been demonstrated to increase the abundance of prey species for coastal benthic feeders such as flounder, thus resulting in elevated feeding and growth rates of flounder compared to those in bare sandy areas with no seagrass detritus (Jenkins *et al* 1993). Edgar *et al* (1993) found that coastal seagrass beds did not have a substantive nursery function compared with estuarine seagrass beds; however, they were considered to play an important role in enhancing adult fish production, through supporting epifaunal crustaceans upon which seagrass-utilising fish feed. For instance, the abundance of two commercial species, blue rock whiting (*Haletta semifasciata*) and six-spined leatherjacket (*Meuschenia freycineti*), was considered to be adversely affected by the loss of coastal seagrass beds in Western Port Bay in Victoria. The protection of seagrass beds from nutrient pollution has increasingly attracted the attention of marine ecologists, fisheries scientists and marine managers during the past two decades, particularly due to the important role of seagrass in supporting fisheries. However, little consideration has been given to protecting the ecologically significant macroalgal-dominated reefs in southern Australian waters. Phillips (1998) provides a partial overview of the problem, in the context of MPA designation. Many commercially and recreationally significant species utilise reefs, including the migratory adults of tommy ruff (*Arripis georgianus*) and Australian salmon (*Arripis trutaceus*), snapper (*Pagrus auratus*) and rock lobster (*Jasus edwardsii*) (Johnson 1988a; Jones *et al* 1990), as do blacklip abalone (*Haliotis rubra*), which live in caves, fissures and crevices (Shepherd 1973).