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1 Introduction

“The art is in saying only as much as the experiment has shown”


This thesis aims to demonstrate how systematic assessments of current knowledge can improve planning for marine protected areas (MPAs) and promote research and adaptive management for marine ecosystems. Using case studies from the Great Barrier Reef and NSW in Australia, I show how complex marine management objectives can be assessed using information, modelling tools and expertise from scientists, managers and the community.

The diverse goals of marine protected areas and other marine conservation programs represent a significant challenge. However, explicitly modelling these goals provides a way to systematically assess a range of independent information sources. By providing a comprehensive source of information, systematic assessments can help to differentiate among alternatives and make decision making more flexible, transparent, accountable and repeatable. In this chapter, I discuss the objectives, background and justification for the thesis and introduce the topics addressed in subsequent chapters.

1.1 Objectives of the thesis

The primary objective of this thesis is to identify a comprehensive, adequate and representative network of MPAs for NSW. My aim is to summarise current knowledge of broad scale patterns in marine biodiversity, environments and processes in a systematic way that can be used to assist managers in making decisions for the conservation and sustainable use of this region.

I first review marine protected area management, ecosystem science, and the systematic methods available to define, map and assess conservation values for MPAs. These techniques are then used to assess MPA goals, criteria and performance measures using spatial planning units in a Geographic Information System (GIS) with a variety of reserve selection modelling tools.

The thesis also aims to demonstrate the capabilities of information systems and analytical tools and to promote the integration of data from many sources as one way to address ecosystem scale problems in marine ecology and management. This is an important way to advance scientific research and to complement the way that knowledge is compiled and tested verbally in the scientific literature. Sharing and presenting data to a wider forum through some of the techniques presented in this thesis is also a means of promoting ongoing research. Figure 1.1 summarises the primary goals and criteria of the thesis as a conceptual multiple criteria model.
1.2 Human impacts on marine ecosystems

It has long been recognised that the earth’s ecosystems are being substantially altered by the increasing scale and impact of human activities (Ray 1975, Holt and Talbot 1978, Sobel and Dahlgren 2004). Many of these changes adversely affect the environment and the survival of many species. While the effects are more readily observed on land (Beatley 1991, Upton 1992, Milewski 1995), impacts in marine and coastal environments have become increasingly apparent and are the subject of large body of literature and research (Hatcher et al. 1989, Norse 1991, Smith and Buddemeier 1992, Gray 1997).

Reviews of changes in different habitats range from cautiously optimistic (Hall 2002, Thompson et al. 2002) to dire (McClanahan 2002), but all predict that marine ecological conditions will deteriorate, particularly in less developed regions. The most optimistic forecasts are based on hope for the increasing use of conservation measures like the establishment of MPAs and all reviews propose that the keys to success are research and the education of environmental managers and the community.

The changes described range from numerous direct impacts at local scales, to indirect, but far-reaching effects operating at regional (Wilkinson 1999, Pogonoski et al. 2000, Wolanski and De’ Ath 2005) and global scales (Wilkinson 1992). These effects have been especially well documented for tropical regions and coral reefs in particular (Aiello 1996, Berkelmans and Oliver 1999, McClanahan 2002, Sweatman et al. 2002), but also for seagrass meadows (Duarte 2002), mangroves (Alongi 2002),...
Introduction

Saltmarshes (Saintlan and Williams 2000, Adam 2002), beaches (Brown and McLachlan 2002), rocky intertidal shores (Thompson et al. 2002), kelp forests (Steneck et al. 2002), estuaries (Kennish 2002), continental shelves (Hall 2002), pelagic ecosystems (Verity et al. 2002) and the deep ocean (Hyrenbach et al. 2000).


1.3 Marine Protected Areas

MPAs are a pragmatic solution to reduce some of these impacts given “that humans can impact environments and ecosystems faster than they become aware of their effects” (Verity et al. 2002). There are various interpretations of what MPAs are, and this can unfortunately confuse their role and the way they are applied. However, it is generally accepted that MPAs include not only highly protected ‘no-take’ marine reserves but also other areas managed effectively for the conservation of biodiversity, ecological processes and cultural resources.

The World Commission on Protected Areas (IUCN 1994) describes a protected area as:

“An area of land and/or sea especially dedicated to the protection and maintenance of biological diversity and of natural and associated cultural resources, and managed through legal or other effective means.”

By definition, this general description of all protected areas specifically includes MPAs and is adopted by the Australian Department of Environment and Heritage (www.deh.gov.au). A special version for marine areas adopted by the IUCN and other international and national bodies defines MPAs as:

“Any area of intertidal or subtidal terrain, together with its overlying water and associated flora, fauna, historical and cultural features, which has been reserved by law or other effective means to protect part or all of the enclosed environment” (Kelleher and Kenchington, 1992).
This marine version of the definition differs slightly in that it is defined by the act of reservation rather than by the objectives of the area although objectives are listed in the IUCN’s “Guidelines for Establishing Marine Protected Areas” (Kelleher and Kenchington, 1992) as:

- maintain essential ecological and life support systems
- ensure the sustainable utilization of species and ecosystems and
- to preserve biotic diversity.

The definition is altered further in the draft National System of MPAs policy for the United States as:

“Any area of the marine environment that has been reserved by Federal, State, territorial, tribal or local laws or regulations to provide lasting protection for part or all of the natural and cultural resources therein.”

In this definition, marine ecosystems and biodiversity are regarded as resources, and the policy admits the potential for confusion when it cautions that:

“Without further clarification, the key terms of ‘area,’ ‘marine environment,’ ‘reserved,’ ‘lasting,’ and ‘protection’ found in the MPA definition are subject to a range of interpretations and lead to an uncertain scope for the National System. Without clear definitions for these five key terms, identifying the sites that should be considered MPAs for the purposes of participating in the National System would be unclear and efforts to fully implement the Order would be fragmented, diffused, and ultimately unsuccessful.” (www.mpa.gov)

The New Zealand Department of Conservation and the New Zealand Ministry of Fisheries (2006) have agreed on another definition for their “Marine protected areas policy and implementation plan” which defines an MPA as:

“An area of the marine environment especially dedicated to, or achieving, through adequate protection, the maintenance and/or recovery of biological diversity at the habitat and ecosystem level in a healthy functioning state”

The New Zealand version allows for the inclusion of areas that while not specifically dedicated to the protection of biodiversity, achieve some level of protection in the course of managing for some other perhaps related objective, such as fisheries management. It includes areas under management for recovery, presumably because impacts have already occurred. The definition also specifically restricts protection, maintenance and recovery to “the habitat and ecosystem level” but omits the need to protect biological communities, species or finer scales of biodiversity.

Furthermore, the definitions of ‘habitats’ and ‘ecosystems’ and ‘healthy functioning state’ are open to interpretation, difficult to quantify and potentially restricted in their scope. In the context of the classification being drafted for this process (Department of Conservation, Ministry of Fisheries 2007a), “habitats and ecosystems” are defined as specific surrogate categories of substratum and depth and used to specify the minimum level of representation required in the MPA system. While additional
finer scale information on assemblages, species, processes, and other criteria may be considered during this process, the policy states that these do not necessarily require representation in MPAs.

An accompanying “Protection Standard” (New Zealand Department of Conservation, Ministry of Fisheries 2007b) aims to operationally define activities that might disqualify an area from being regarded as an MPA and thereby promote other areas to MPA status, which may incidentally have some level of protection, even though they do not explicitly have the conservation of biodiversity as a management objective. The latter have been taken to include areas with some level of protection or at least restraint including areas managed under fisheries, mining, transport, resource planning and biosecurity legislation. The danger is that small changes in the definitions of MPAs, habitat classifications and “protection standards” may be used politically to promote an existing system of “incidental MPAs” as sufficient for biodiversity conservation and then subsequently use this as an argument against establishing additional MPAs.

A major difference among definitions is that those that specify biodiversity conservation as the objective for the area provide a permanent and universal guideline against which all subsequent definitions, classifications, legislation, decisions, management and outcomes can be evaluated. There is considerable value in having management based on defined objectives (Jones 1994, Slocombe 1998, Barber and Taylor 2004, Edvardsson 2004) rather than on hypothetical outcomes that may or may not be realised or proven or on arbitrary classifications or rules that are subject to revision, differences of opinion or varying circumstances.

For example, the IUCN protected area categories (IUCN 1994, 2000) provide a goal based, internationally recognised alternative to schemes such as the New Zealand MPA “protection standard”. Each of the IUCN categories are defined by their primary goal, with accompanying definitions and guidelines for selection and governance. The different categories include various objectives and corresponding levels of protection ranging from strict nature and wilderness reserves, to areas managed mainly for recreation and sustainable use (Appendix 7). These categories are universal enough in their range and generality of goals to embrace many management options and they provide clear direction for the development of specific measures, protection standards and assessments.

Because of the potential for confusion and the misuse of verbal definitions, Chapter Two describes an explicit model of MPA goals and criteria which provides a way to operationally define conceptual goals in terms of specific objectives, criteria and performance measures. This is particularly important in the context of this thesis which addresses the selection and zonation of multiple use MPAs in Queensland and New South Wales. Within these large MPAs, a mosaic of different areas or zones are managed according to objectives aligned with several of the IUCN categories.

Many reviews have emphasised that the most immediate and demonstrable benefits from MPAs are those arising from fully protected “no-take” marine reserves (Halpern 2000, Roberts and Hawkins 2000, Roberts et al. 2001, Warner and Halpern 2002) but there are also legitimate reasons to establish MPAs with varying levels of protection. One, is that fully protected areas on their own, may not be
Introduction

sufficient to address all impacts (Allison et al. 1998). Because of their impact on existing activities, single highly protected areas are often limited in size and therefore may be unable to address the range of biodiversity, ecological processes and human activities operating at much broader ecosystem scales. Large multiple use MPAs, which include systems of highly protected areas and other zones, have however the potential to conserve a wide range of conservation values within areas large enough to include whole ecosystems, regions and a diverse range of human activities. The benefits of this approach include greater coordination in reserve design, planning, finance, compliance, monitoring and education and a more integrated approach to threat management.

The Great Barrier Reef Marine Park is probably the best known example of a large, multiple use MPA (Craik 1996) and marine parks established in NSW under the Marine Parks Act 1997 are based on this model (Brunckhorst 1998). In NSW, for example, legislation for marine parks is based on the primary objectives of the Marine Parks Act to “conserve marine biological diversity and marine habitats” and “maintain ecological processes.” These objectives are specifically aimed at conserving whole ecosystems. The Act also includes human activities as an integral component of this ecosystem approach. Where consistent with the primary aims, the Act aims to “provide for the ecologically sustainable use of fish (including recreational and commercial fishing) and marine vegetation” and “provide opportunities for public appreciation, understanding and enjoyment.”

Multiple use MPAs in NSW and Queensland include many, ‘no-take’ sanctuary areas which protect fish, invertebrates, plants and seabed from fishing and other extractive activities. However, these highly protected areas are buffered within other zones assigned different levels of protection and supported by integrated management of whole ecosystems throughout the entire multiple use MPA. The latter zones include those that allow limited recreational angling and commercial fishing and integrated management includes tools, legislation and advocacy to control impacts from within and outside the MPA including the effects of adjacent coastal development, pollution from catchments (Johnson et al. 1999, Done 1998, Koop et al. 2001), fisheries practices (Gribble and Robertson 1998, Russ et al. 1998, Fox and Knuckey 2001), dredging (Smith and Rule 2001) mining (Prideaux 1999), shipping (Roberts 2006), aircraft (Brown 1990) anchoring (Harriot and Fisk 1990), crowding (Inglis et al. 1997), off-road vehicles, trampling and diving (Davis and Harriot 1996, Harriot et al. 1997, Rouphael and Inglis 1997).

There is a consensus among most marine ecologists that MPAs, and especially ‘no-take’ reserves, can provide a variety of ecological, economic and social benefits (AAAS 2001). These benefits are the subject of many studies and reviews (Halpern 2000, Roberts and Hawkins 2000, Roberts et al. 2001, Warner and Halpern 2002) and there is a growing body of evidence that for many species, even small reserves can result in a greater abundance of larger individuals inside marine reserves (Jones et al. 1992, MacDiarmid and Breen 1992, Jennings et al. 1996, Edgar and Barrett 1999, Gladstone 2001, Kelly et al. 2000, Schroeter et al. 2001, Willis et al. 2001, Langlois and Ballantine 2005). There is also evidence these animals can ‘spill-over’ into surrounding areas (Russ and Alcala 1996, Roberts et al. 2001, Kelly et al. 2002). Other benefits include effects on sex ratios, fecundity and other life
Introduction

history characteristics (Harmelin et al. 1995, Pillans et al. 2005), as well as protection of habitat and indirect benefits for other species and habitats through trophic cascades and other interactions (Pinnegar et al. 2000, Shears and Babcock 2003, Langlois et al. 2006).


Many MPAs are established primarily for the protection of biodiversity, but MPAs are increasingly used as tools in sustainable fisheries management. These can provide security for both biodiversity and fisheries (Bohnsack et al. 1999, Ward and Hegerl 2003) and protect against the effects of fishing gear, over-fishing and the by-catch of non-target species (Ballantine 1997, Ward et al. 2001). Although there are many examples of effects within the boundaries of MPAs, there are fewer documented cases of benefits extending to fisheries outside of MPAs (Russ and Alcala 1996, Roberts et al. 2001, Russ 2002, Butcher 1999, Galal et al. 2002, Pillans et al. 2005). This may, at least partly be due, to the more subtle nature of many effects and the lack of studies targeting these more ambitious projects. This is one application of MPAs that has received little attention in NSW. MPAs in this region, may by default, have these effects, but there has been no comprehensive use of MPAs as part of fisheries management plans and few studies (Butcher 1999) have been designed to assess benefits to fisheries or other effects beyond the most easily predicted outcomes.

MPAs have a vital role in marine research and education as reference areas where scientists, managers and communities can study and understand how marine ecosystems behave in the absence of fishing and other activities. Without these areas, it is impossible to understand how marine ecosystems behave naturally and how human activities have altered the ocean.

While MPAs are already the focus of much marine research, there are many aspects of MPA performance that require investigation. These topics include the effectiveness of compliance (Gribble and Robertson 1998, Davis et al. 2004, Kritzer 2004), displacement of fishing effort (Russell 1997), effects on larval supply and recruitment (Planes et al. 2000, Warner et al. 2000, Manriquez and Castilla 2001, Sale et al. 2005) and variations in benefits for different species, locations and human activities (Rogers and Beets 2001, Tupper and Rudd 2002, Shipp 2003, Nardi et al. 2004, Langlois and Ballantine 2005).

Most importantly, managers and scientists need to understand how to sustain marine ecosystems within and outside of the boundaries of MPAs. All of these questions warrant using a systematic approach to MPA selection and scientifically assisted, long term programmes of adaptive management to extract the most knowledge from each new MPA.
1.4 Science and management

In dealing with a range of human induced impacts, managers constantly need to make long-term decisions using the best available information. However, this information is rarely accessible in formats that can be readily applied to decision making. As a result, planning is often done on an ad hoc basis.

In a survey of 22 managers at the Great Barrier Reef Marine Park Authority, most agreed on the types of information they would like to use but disagreed on how easily they could access this information (Bollard-Breen 2006). Most planners said that ecological and social data were seldom used, as it was difficult to access and interpret. They often relied on precedents from past decisions, previous policy, the requirements of existing commercial and social activities and input from internal and external consultation. However, most respondents wanted to access all relevant information in an easy to interpret format; share information between State and Federal agencies; obtain detailed site information about the values and use of reef areas; look at relationships in the data; and identify where conflicts could occur between use and conservation.

In a survey of 38 conservation plans in the United Kingdom (Pullin et al. 2004), 71% of the plans were justified as a continuation of traditional management practices, 29% by habitat management handbooks and 16% by secondary reviews of literature. Only 11% made use of primary scientific literature. When 141 managers were asked about the most frequently used sources of information used to support their decision making, 60% replied ‘existing management plans’, 49% replied ‘expert opinion from outside the group’, 47% replied ‘handbooks, books or reviews’, and 46% replied ‘personal accounts of traditional management practices’. The least used sources were web based materials, popular articles (4%) and published scientific papers (23%).

When asked why they did not access primary scientific literature more frequently, most said it was too time consuming to access (65%) or read (60%), and 25% said it was ‘too technical and difficult to interpret in the context of their decision making.’ When asked to scale the relative inputs of ‘experience based information’ versus ‘evidence based information’, 75% rated experience-based information as more important while 5% thought evidence-based information was more influential. The authors suggested that the managers “were not making full or systematic use of information available to support decision making” and “frequently rely on the status quo of continuing with an established but unevaluated practice.”

Guikema and Milke (1999) surveyed 22 government and non-government organisations involved in biological conservation, natural resource management, recreation in natural areas and strategic planning and decision support. Only five agencies indicated using some form of quantitative tool for conservation planning. The rest reported relying on managerial experience and political process. For the five agencies reporting use of quantitative methods, almost all reported using basic scoring and ranking methods (Table 1.1).
Table 1.1. Summary of conservation planning procedures: current practice (Guikema and Milke 1999).

<table>
<thead>
<tr>
<th>Conservation agency</th>
<th>Summary of process used</th>
</tr>
</thead>
<tbody>
<tr>
<td>NZ Dept of Conservation – Conservancy</td>
<td>Negotiation and interpretation of guidance documents. Some project scoring</td>
</tr>
<tr>
<td>NZ Dept of Conservation – Regional</td>
<td>Project scoring with cut-off scores</td>
</tr>
<tr>
<td>US National Parks Service - conservation planning</td>
<td>Project scoring with weighted additive aggregation model</td>
</tr>
<tr>
<td>US National Parks Service - staff planning</td>
<td>Gap analysis based on current and required staffing level</td>
</tr>
<tr>
<td>Australian Nature Conservation Service – selecting protected areas</td>
<td>Scoring of areas</td>
</tr>
<tr>
<td>Hong Kong</td>
<td>Ranking of projects by criteria</td>
</tr>
<tr>
<td>World Conservation Union</td>
<td>Criteria scoring, conversion of scores to values and an additive aggregation model</td>
</tr>
</tbody>
</table>

The importance of these difficulties in linking science with conservation management is well recognized and the gap between science and management is often significant, even where substantial resources are devoted to both (Done 1998). Part of this dilemma may have arisen from the contrasting perspectives of scientists and policy makers. In Table 1.2, Crosby et al. (2000) highlight differences in these priorities and some of the problems in applying ecological research to immediate management problems.

Woodley and Ottesen (1992) identified four factors limiting the use of science in decision making at the Great Barrier Reef Marine Park Authority. Firstly, research results were not comprehensive or conclusive and variables could not be controlled to establish precise cause and effect relationships without uncertainty in variable, complex ecosystems. Secondly, scientists often gave results that were highly qualified, which although rigorous, did not provide quick and simple solutions and there were also issues with intellectual property. Thirdly, consensus among scientists was not always possible, with heated disputes within and among disciplines, requiring managers “to develop conflict resolution skills when dealing with scientists, in the same way as dealing with conservationists and developers.” Fourthly, decisions were made ‘within a legal and administrative framework involving public participation and statutory time limits…and scientists unfamiliar with this process … expressed frustration and concern with the manner in which research results (were) used – or not used’ (Woodley and Ottesen 1992).
**Table 1.2. Contrasting perspectives between scientists and policy makers in how they view and deal with similar parameters (Crosby et al. 2000).**

<table>
<thead>
<tr>
<th>Ends and Means</th>
<th>Scientists</th>
<th>Policy makers</th>
</tr>
</thead>
<tbody>
<tr>
<td>Goal, purpose:</td>
<td>Seek truth</td>
<td>Public welfare, represent constituents</td>
</tr>
<tr>
<td>Basic orientation:</td>
<td>Understand, explain</td>
<td>Act, decide</td>
</tr>
<tr>
<td>Mechanisms:</td>
<td>Unbiased methods, impersonal</td>
<td>Adversarial, opposing interests, highly personal</td>
</tr>
<tr>
<td>Real World Affairs:</td>
<td>Problems to be solved</td>
<td>Problems to be resolved</td>
</tr>
<tr>
<td>Currency:</td>
<td>Knowledge, expertise</td>
<td>Power, influence</td>
</tr>
<tr>
<td>View of personal judgement:</td>
<td>Mistrustful stick to data</td>
<td>Essential, need to act before being certain</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Time and Attention Spans</th>
<th>Scientists</th>
<th>Policy makers</th>
</tr>
</thead>
<tbody>
<tr>
<td>For System:</td>
<td>Long, incremental</td>
<td>Short – must act now</td>
</tr>
<tr>
<td>For individual:</td>
<td>Next grant, tenure</td>
<td>Next election</td>
</tr>
<tr>
<td>Cognitive demands:</td>
<td>Depth, detail, little range</td>
<td>Huge range, little depth</td>
</tr>
<tr>
<td>Attention span:</td>
<td>Long</td>
<td>Short – situational press</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Accountability and Rewards</th>
<th>Scientists</th>
<th>Policy makers</th>
</tr>
</thead>
<tbody>
<tr>
<td>Responsible to:</td>
<td>Standards, peers</td>
<td>Constituency</td>
</tr>
<tr>
<td>Real-world accountability:</td>
<td>Low</td>
<td>High</td>
</tr>
<tr>
<td>Rewarded for:</td>
<td>Experimenting</td>
<td>Being right</td>
</tr>
<tr>
<td>Career Incentives:</td>
<td>Successful research, publishing</td>
<td>Power, orchestrating outcomes</td>
</tr>
<tr>
<td>Idealized mode of action:</td>
<td>Autonomy</td>
<td>Being a team player</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Communicate and Interact</th>
<th>Scientists</th>
<th>Policy makers</th>
</tr>
</thead>
<tbody>
<tr>
<td>Primary means:</td>
<td>Written</td>
<td>Personal, face to face</td>
</tr>
<tr>
<td>Role of interpersonal skills:</td>
<td>Low relevance to work quality</td>
<td>Heart of work effectiveness</td>
</tr>
<tr>
<td>Value characteristics in colleagues:</td>
<td>Knowledge, analytical power, creativity</td>
<td>Loyalty, judgement, knowledge</td>
</tr>
<tr>
<td>Language imagery:</td>
<td>Precise, impersonal, technical</td>
<td>Understandable, inclusive, personal</td>
</tr>
</tbody>
</table>
While these problems are relevant, managers and scientists must still work with the complexity entailed in trying to conserve whole ecosystems, and the many species, habitats, processes and threats they include. The inability of science to provide simple answers to these ‘wicked’ or ‘trans-science’ problems (Miller 1993), has led to what have become known as ecosystem approaches to management and ecological research.

1.5 Ecosystem management

Research and management for individual species and sites has a major role in conservation research and management. However, it cannot hope to address all ecosystem components at the scales required by many environmental problems. Similarly, general ecological theories and paradigms have a crucial role in research and management strategies. However, for these to be applied at the scales demanded by applied management, they often require basic information on the distribution, abundance and diversity of organisms, and the environments and the processes that effect their survival. In response to this complexity, there has been a shift in ecological management towards strategic broad scale, ecosystem and landscape approaches to conservation management and planning (Ray 1975, Franklin 1993, Noss 1996, Sherman and Duda 1999, Pitcher 2000, Trombulak et al. 2004). This approach is now an integral part of government policies, agreements and legislation and recognised internationally among many scientists, institutions and conservation groups.

Global marine conservation initiatives include the International Coral Reef Initiative, the Global Coral Reef Monitoring Network and the IUCN program for a Global System of Marine Protected Areas (Kelleher et al. 1995). These and other programs have worked with scientists, managers and communities to develop definitions, guidelines and reviews for marine protected areas throughout the world. Planning at these scales encompasses whole nations, regions and even the large intervening areas of deep ocean basin (Hyrenbach et al. 2000).

Australia has commitments to protect marine biodiversity under such agreements as the Convention on Biological Diversity (UNEP 19994), the World Heritage Convention (Lucas et al. 1997), the Intergovernmental Agreement on the Environment (Commonwealth of Australia, 1992a), the National Strategy for Ecologically Sustainable Development (Commonwealth of Australia 1992b), the National Strategy for the Conservation of Australia’s Biological Diversity (Commonwealth of Australia 1996) and a commitment to establish a National Representative System of Marine Protected Areas (ANZECC 1998ab).

In response, Australian marine managers and scientists have led an ongoing programme to establish and manage representative systems of MPAs within each of 65 marine bioregions and provinces throughout Australian waters. The Federal government has helped to fund and coordinate programs to establish MPAs in inshore waters through each State government, expanded the network of highly protected areas in the Great Barrier Reef Marine Park and established MPAs in Commonwealth waters of the Great Australian Bight, Ningaloo, Tasmanian sea mounts, Elizabeth and Middleton Reefs,
Introduction

Ashmore Reef, Cartier Island, the Cod Grounds, and offshore of NSW Marine Parks at the Solitary Islands and Lord Howe Island. The Commonwealth is now developing regional marine plans for the vast areas of the Australian economic exclusion zone that will include processes to identify and establish representative marine protected areas.

Decision making for these MPAs requires information ranging from spatial scales of metres to thousands of kilometres for a diverse range of physical, biological, social, economic and cultural criteria. However detailed information at such scales is rarely available, even for well known species. Marine surveys are often difficult and expensive and many habitats are hidden underwater in areas exposed to harsh weather conditions. The detailed studies that exist for some species, communities and habitats are usually restricted to particular locations which represent a small fraction of the ocean’s area. Extrapolating these results to other areas is fraught with uncertainty. Similarly, applying general paradigms and theories to specific sites may be unreliable for all but the simplest of predictions. There is therefore an urgent need to develop methods to apply the knowledge we have of relatively simple, small scale phenomena to complex large scale systems, while still exercising good judgement.

1.6 Ecosystem science

The increasing demand for ecological science in managing human interactions with large, complex ecosystems has led to ‘integrated’, ‘ecosystem’ approaches to environmental research and management (Ray 1975, Agardy 1994, Christensen et al. 1996, Sherman and Duda 1999, Smith and Smith 2001, Ward and Hegerl 2003) and an increasing advocacy for area based strategies like MPAs (Caddy 2000, AAAS 2001, Ward et al. 2001). However there are legitimate concerns over the necessarily approximate nature of ecosystem concepts and definitions (Inglois 1992, Simberloff 1998, Goldstein 1999, O’Neill 2001). These are at least partly due to the complexity itself and our current limits in easily understanding and describing ecological relationships and processes (Table 1.4). While interactions in physics and chemistry are often deterministic and highly predictable, more complex systems can be more stochastic, chaotic and difficult to predict using general theories. The difficulties in developing precise theories in ecology have been referred to as “a constipating accumulation of untested models, most of which are un-testable” and ecology has been described as being in a state of “paradigms lost” (Schoener 1972, Woodwell 1978, Schrader-Frechette and McCoy 1994ab).

The spatial and temporal scales of whole ecosystems are difficult, if not impossible, to fully control or replicate in a classical experimental setting (Schindler 1998). These challenges have led some researchers to describe ecology as diverging into ‘two cultures’ (Holling 1998). A so-called applied ‘integrative’ or ‘functional’ ecosystem oriented, inductive, conservation ecology and a more traditional deductive ‘pure’, ‘deep’ or ‘compositional’ ecology (Miller 1993, Jacob 1994, Schrader-Frechette and McCoy 1994ab, Wells 1995, Callcott et al. 1998). Differences between these extremes are summarized by Holling (1998) in Table 1.4.
Analytical ecology has been termed a ‘science of the parts’ and aims to reduce uncertainty to a point where acceptance is unanimous, however it may do so at the cost of being small in scale. The science of the ‘integration of the parts’ differs from the ‘essentially experimental, reductionist’ analytical ecology in that it is “fundamentally interdisciplinary and combines historical, comparative and experimental approaches at scales appropriate to the issues.” It “has emerged regionally in new forms of resource and environmental management where uncertainty and surprises become an integral part.”

It is a science “concerned with integrative modes of inquiry and multiple sources of evidence” and one that “has the most natural connection to…social sciences” and “provides a bridge between analytical science, policy and politics” (Holling 1998). It therefore lends itself in many ways to strategies in ecosystem management such as establishing networks of MPAs. While there have been attempts to resolve confusion in the terminology and principles of ecosystem ecology (Grumbine 1997, ANZECC 1999, Callicott et al. 1999, Trombulak et al. 2004), this is still a science very much in its infancy. However, the most compelling reason to pursue its development is that it aims to directly confront issues dealing with the very systems that humans and other organisms depend on.
Table 1.4. Comparing the two cultures of biological ecology (Holling 1998).

<table>
<thead>
<tr>
<th>ATTRIBUTE</th>
<th>ANALYTICAL</th>
<th>INTEGRATIVE</th>
</tr>
</thead>
<tbody>
<tr>
<td>Philosophy</td>
<td>• narrow and targeted&lt;br&gt; • disproof by experiment&lt;br&gt; • parsimony the rule</td>
<td>• broad and exploratory&lt;br&gt; • multiple lines of converging evidence&lt;br&gt; • requisite simplicity the goal</td>
</tr>
<tr>
<td>Perceived organisation</td>
<td>• biotic interactions&lt;br&gt; • fixed environment&lt;br&gt; • single scale</td>
<td>• biophysical interactions&lt;br&gt; • self-organisation&lt;br&gt; • multiple scales with cross scale interactions</td>
</tr>
<tr>
<td>Causation</td>
<td>• single and separable</td>
<td>• multiple and only partially separable</td>
</tr>
<tr>
<td>Hypotheses</td>
<td>• single hypotheses and null&lt;br&gt; • rejection of false hypotheses</td>
<td>• multiple competing hypotheses&lt;br&gt; • separation among competing hypotheses</td>
</tr>
<tr>
<td>Uncertainty</td>
<td>• eliminate uncertainty</td>
<td>• incorporate uncertainty</td>
</tr>
<tr>
<td>Statistics</td>
<td>• standard statistics&lt;br&gt; • experimental&lt;br&gt; • concern with Type I error</td>
<td>• non-standard statistics&lt;br&gt; • concern with Type II error</td>
</tr>
<tr>
<td>Evaluation goal</td>
<td>• peer assessment to reach ultimate unanimous agreement</td>
<td>• peer assessment, judgement to reach a partial consensus</td>
</tr>
<tr>
<td>The danger</td>
<td>• exactly right answer for wrong question</td>
<td>• exactly right question but useless answer</td>
</tr>
</tbody>
</table>
Ecosystem approaches are realistic given the widespread and cumulative nature of ecological problems (Peters et al. 1997) and because they acknowledge that ecosystems and impacts:

- operate at a wide range of spatial and temporal scales (Ray 1991, Sherman and Duda 1999)
- include species and habitats beyond the commercially valuable, charismatic, threatened or well studied areas that receive most attention (Franklin 1993, Jones and Kaly 1998)
- are dynamic, non-linear, non-equilibrium and stochastic (Brown and MacLeod 1996, Levin 1998)
- include humans and their social, economic and cultural values and behaviours (Kenchington et al. 1992, Bohnsack and Ault 1999, Agardy 2000, Ward and Hegerl 2003)
- may require pragmatic and innovative approaches to research and management (Holling 1998)

A perceived danger in relying on ecosystem ecology is in directing effort and resources away from research targeted at more specific goals and principles. However, ecosystem science should aim to incorporate the scientific principles and predictions of experimental research, modelling and other disciplines within a broader systematic framework that tests our ability to manage and adapt to changes in our environment.

Techniques that aim to apply the rigour of classical experiments to marine ecosystems include the use of mesocosms (Schindler 1998), exclusions and other field experiments (Underwood 1981, Hurlbert 1984, Jones et al. 1988, Thrush et al. 1995, Chapman and Underwood 1996, Ellis et al. 2002), before and after, control and impact monitoring designs (Underwood 1993, 1996, Underwood et al. 2003) and an overall framework of adaptive management. Adaptive management aims to apply a scientific approach to identifying and testing alternative management hypotheses through deliberate, experimentally designed ecosystem interventions (Walters 1986). Walters and Holling (1990) describe three levels of adaptive management as:

- evolutionary ‘trial and error’, in which initial choices are haphazard, but later choices may give better results
- ‘passive adaptive’ where historical data at each time are used to build a single best model to determine a management response and
- ‘active adaptive’ where data at each time are used to structure a range of alternate models and experimental designs to distinguish clearly among effects by making the best possible use of opportunities for replication and comparison.
Without reasonable information on the marine ecosystems involved, MPA selection processes are likely to be ‘trial and error’ experiments at best and their results likely to provide only marginal improvements. With at least a systematic approach to inform decisions, a ‘passive adaptive’ process may help promote intelligent choice. Various scenario modelling tools may also assist in recognising a range of alternative hypotheses and exposing uncertainties using what information is available. The following assessments in this thesis aim to apply MPA selection processes to at least this level.

However, an active adaptive strategy requires that MPA networks are experimentally designed to distinguish among competing hypotheses. MPAs provide an ideal opportunity for large scale experimental manipulations but the involvement of scientists during planning is often minimal. The design of monitoring programs to test the effectiveness of MPAs is usually only considered after the MPAs have been established. They are usually (but not always e.g. Langlois and Ballantyne 2005) based on a single unreplicated reserve, with limited data collected before establishment and a minimum number of control sites used for comparison.

Research is also usually restricted to simple hypotheses on whether or not there has been an effect. Opportunities to test different reserve sizes, shapes or configurations are usually disregarded as are investigations into more complex effects on surrounding areas and fisheries. This area remains a wasted opportunity for research and management to improve MPAs at a time when information for planning and advocacy is most critical. The systematic assessments in this thesis are a step towards a more rigorous approach to ecosystem research and MPA planning, but represent only a small part of the solution.

1.7 Systematic assessments to identify MPAs

Systematic assessments for protected areas aim to identify strategies that logically meet explicit conservation goals using available scientific data, theories and models. Because they involve many goals and stakeholder interests, these assessments often focus on ways to combine information from many sources to predict outcomes for different management scenarios.

The methods are explicitly goal driven, but often approximate. From a classical science perspective, they are essentially exploratory and usually based on the joint analysis, display and interpretation of many data sets. They are however, directly targeted at informing management in a field where certain predictions are unlikely and where priorities differ among managers and different sectors of the community. The assessments are, to large extent, information resources that allow scientists, managers, and communities to explore the potential effects of alternative management plans. This approach has evolved with an increasing emphasis on ecosystem wide planning, the coordinated establishment of protected areas and the development of computer modelling and Geographic Information Systems (GIS) (Lourie and Vincent 2004, Leslie 2005). Improvements in information technology have greatly assisted this approach, but technology is not necessarily an essential component of systematic assessments.

Vanderklift and Ward (2000) summarise the collection of biological information for this process in Figure 1.2. To be comprehensive, this should also include the consideration of social, economic and cultural information and include some form of performance assessment and feedback mechanism (Ward et al. 1998). The biggest challenges in this process are in assembling enough information of relevance to the questions asked and in integrating, interpolating and interpreting information at scales appropriate for management. While there are many gaps, even the existing knowledge for many marine ecosystems can represent a lot of information to be managed and interpreted.
An understanding of this knowledge is necessary if management is to have a comprehensive and up to date ‘whole ecosystem’ view of marine conservation issues. However, this information is often fragmented among many different studies for specific sets of sites, times, species and habitats with little coordinated planning to direct research effort. Many of these studies address specific hypotheses or taxonomic questions which introduce sampling biases that can make inferences about spatial distribution and temporal trends unreliable.

Literature reviews aim to define general patterns and trends and meta-analyses are sometimes used to quantify overall patterns in research results from many studies. These integrate the conclusions of research, but do not often present the individual results of studies or make the underlying data available to assess or develop alternative hypotheses or interpretations.

However, a growing number of researchers and institutions now electronically store, document, and share large databases of spatially explicit data for whole regions, groups of species, habitats and ecosystems (Hamilton et al. 1995ab, Blake 1996, Grassle 2000). Many research programs also set out to systematically survey regions, habitats and taxa. Finally, there are also regional assessments, like the studies in this thesis, that aim to collate and use this data to provide decision support for ecosystem scale decisions in environmental management.

Dealing with large amounts of data is challenging and there are limitations in using data that is collected for different purposes. However, systems and processes have evolved to address such problems. GIS techniques to map and integrate data from different sources have been available for over 30 years (Cocks and Baird 1991). Aerial and marine remote sensing techniques and biological survey methods to sample large areas have advanced in many different ways and have become increasingly efficient (Kennelly and Underwood 1984, Stoms and Estes 1993, Burrage et al. 1996, Pitcher et al. 1999, Rieg and Piller 2000, Ducrotoy and Simpson 2001, Harvey et al. 2001, Kostylev et al. 2001, Cappo et al. 2003, Ekebom and Erkkila 2003, Fitzpatrick 2003, Francis et al. 2003, Hewitt et al. 2004, Parsons et al. 2004, Spencer et al. 2005, Wright et al. 2006, Morrison and Carbin 2006).


These tools, models and underlying and data are usually used with GIS to link many data sets through shared locations and to display and analyse results in a spatially realistic setting. These systems are powerful tools for data visualisation and analysis and are particularly useful when dealing with area based strategies like MPAs (Bushing 1997, Kraak and MacCeachren 1999, Lewis et al. 2003).

GIS were previously considered as specialist technologies for trained operators, usually in geography. However, they have become increasingly accessible to a wider class of users and are now an important tool in many fields of ecology, modelling and community based management. Their realistic, visual capabilities and flexibility make them particularly useful for workshops and other participatory decision making processes that can include scientists, managers and the community (Craig and Elwood 1998, Elwood and Leitner 1998, Calamia 1999, Kerrigan et al. 1999, Harris and Weiner 1998, Talen 2000, Scholz et al. 2004, Leslie 2005, Bruce and Eliot 2006, Close and Hall 2006).

These techniques can make research and management more transparent and accountable as the rationale and information behind decisions can be shared and made available for ongoing review and future research. The current challenges however, lie not with the technology, but in the availability of suitable data and in incorporating these methods into mainstream planning, consultation and science. Answers to these challenges lie in making use of these tools in ‘real world’ applications to create an awareness of how effective they can be in applied environmental research and management.

1.8 Structure of the thesis

The structure of this thesis attempts to follow a logical path describing how to identify and select MPAs. It initially reviews goals and objectives for MPAs and develops detailed criteria to help determine the information, methods and processes required. The thesis then describes the relevance and limitations of the information and methods used and demonstrates how these techniques can be used to identify systems of MPAs using several case studies as examples. In the following chapter, I review goals and criteria for marine protected areas and develop a conceptual multiple criteria model for marine protected areas in Australia and New Zealand. These criteria are used to determine the types of information necessary for decision-making, which in this case, includes a range of different data sets describing many aspects of marine ecosystems for many locations.
Chapter 3 then reviews methods used to describe marine ecosystems and ways to integrate, store and analyse the resulting information. The chapter first considers the benefits and limitations of using physical and biological surrogates to estimate pattern in the distributions of environments and organisms. This includes the development of bioregionalisations and classifications using a range of approaches. It then describes the use of Geographic Information Systems to map and integrate this information and how modelling tools can be used to develop and evaluate different scenarios for MPA networks.

Chapter 4 outlines the primary data sources and methods used to map and describe marine ecosystems in NSW and the methods used to assess locations for their suitability as MPAs. In Chapter 5, I summarise the types of MPAs established in NSW, and their aims, limitations and capabilities. I also briefly describe the existing MPA system and how it was established.

In Chapters 6 to 8, I present case studies used to identify options for MPAs in four marine bioregions in NSW. These bioregional assessments were conducted across broad spatial scales using coarse surrogates for biodiversity and approximate indicators of condition and vulnerability. The chapter describes the systematic evaluation of alternative locations for MPAs using graphs, maps and two decision support tools, C-Plan (NPWS 2001) and Criterion Decision Plus (InfoHarvest 2000). Recommendations from these studies are summarised in these chapters but presented in more detail in Appendices 2-4. The results of these studies are now being used to implement a representative system of marine protected areas in NSW and have so far, resulted in the declaration of large, multiple use marine parks near Port Stephens and Batemans Bay.

In Chapter 8, I combine information from the above assessments and a previous study (Avery 2001) within databases and models for ongoing marine conservation planning in NSW. These include statewide models built using C-Plan and in Marxan, the reserve selection software adapted for the Great Barrier Reef Marine Park Authority. I also develop a multiple criteria model in Criterion Decision Plus to assess options for large marine parks in the Hawkesbury and Batemans Shelf marine bioregions.

Chapter 10 uses the Cape Byron Marine Park as a case study to show how finer scale ecological, social and economic data, public consultation and local knowledge can be used to help develop detailed plans for MPAs. Once large, multiple use marine parks are declared in NSW, the government is required to develop a zone plan for each park and assign different locations to zones with varying levels of protection and restrictions on extractive human activities. ArcView GIS, C-Plan and the Marxan simulated annealing algorithms are used to analyse social and ecological data and develop zoning options according to a range of criteria. The data and models are then used in an interactive ‘participatory GIS’ with managers and community representatives to develop draft plans and provide recommendations to the State government. The zone plan for the park has now been implemented and incorporates many of these recommendations.
The final chapter of the thesis discusses the benefits and limitations of these approaches and suggests how they can be improved. Despite using only approximate data, the assessments provided convincing support for environmental decisions based directly on recognised conservation goals. The assessments also establish an organised framework to methodically plan new research and management and the information systems and tools to store, explore, and incorporate new information.

Many of the techniques have existed in some form for over 20 years, but marine managers and scientists have been slow to exploit this potential. These methods have however, become more readily available and cooperation among scientists and institutions has provided greater opportunities for their use. The techniques aim to improve conservation planning and research in the following ways.

1. Explicitly linking management objectives and information with problems and decisions.
2. Increasing access to information for managers, scientists, stakeholders and communities.
3. Promoting transparent, accountable and repeatable decision making.
4. Preserving and integrating knowledge from many digital, paper and anecdotal sources.
5. Establishing information systems and processes to build on existing research.
6. Identifying information gaps for future research.
7. Providing products for data visualisation, education and awareness.
8. Assessing complex ecosystem scale interactions and emergent properties.
9. Sharing ideas, priorities and decisions among scientists, managers and communities.
10. Generating realistic models to prioritise, test and plan conservation research and management.

There are substantial benefits in systematically integrating information for ecological planning and research but still many barriers to success and large gaps in our knowledge of marine ecosystems. I present the spatial representation, integration, preservation and use of ecological knowledge as a fundamental responsibility for ecological science. It is a field still in its infancy, but one with the potential to change the way we manage and understand ecosystems and our place in them.

Statement of the contribution of others in Chapter 2.

Ron Avery and I together adapted the main conceptual criteria and their interpretation for MPAs in NSW from national guidelines, policy, literature and consultation. Ron also researched and summarised most of the ecological reserve design guidelines included in Appendix 1. I developed the conceptual multiple criteria models to represent and interpret these goals and criteria.
2 Goals and criteria for marine protected areas

The number of specific objectives for MPAs, and especially multiple use MPAs, is potentially very large. While some MPAs are established to protect particular values, most attempt to conserve a range of habitats, species and processes and manage a wide array of human activities. As part of a network, there is also the potential to select MPAs that can collectively meet many objectives.

To do this requires clearly articulated goals and ways to consistently apply criteria. For MPAs, this step is particularly important, as it is unlikely that a site will actually be removed from a network after it has been established. The cost of an ineffective MPA is at least, the potential loss of a successful MPA located at an another, more effective location. A lack of clearly defined goals is likely to undermine support for a reserve and lead to an ad hoc selection of areas of potentially low value. It will also fail to provide direction for the data collection, survey designs, site assessments, regulations and performance assessment (Day 2002, Day et al. 2003). Objectives help to direct management planning, strategies and activities. They also ensure accountability and help to avoid ‘goal displacement’ (Barber and Taylor 2004). The more vague that goals are, the more likely the reserve will be managed according to external criteria that may have little to do with conservation or the benefits that an MPA can provide.

The paradox in defining ecosystem based goals, such as those for selecting and managing a system of marine protected areas, is that goals should be universal enough to include all of the potential benefits and problems likely to occur, yet be specific enough to be measurable for individual locations and situations (Edvardsson 2004). It can be difficult to impose a consistent structure or priority for goals and criteria or relate how these should link to the actual information and selection methods used or, to the decisions that are made.

Many authors have therefore, compiled lists of criteria, often summarised under separate headings (Margules and Usher 1981, Smith and Theberge 1986, Theberge 1989, Jones 1994, Salm and Price 1995, Jones 2001). Some authors separate criteria into those aiming to conserve ecosystems and those aiming to sustainably manage and provide for human activities. Jones (1994) however, lists conservation goals under headings for ‘Scientific’, ‘Economic’ and ‘Cultural,’ as does Roberts et al. (2003), although they also include a category for ‘Feasibility/Practicality’. Leslie (2005) groups objectives under biodiversity conservation, sustainable fisheries and scientific research. Others like Done and Reicheldt (1998) are more specific in advocating for selection and assessment based on explicit and measurable ecological outcomes.

Agardy (1997) organises ‘myriad’ objectives under seven broad goals, including some that are not overtly recognised elsewhere. The first, is to assign a sense of place of place to an area to encourage ownership, a function of MPAs that is often overlooked. The second is to provide a testing ground for management. The third is to provide social benefits. The fourth, to regulate
levels of natural resource harvest. The fifth and sixth include the protection of ‘sensitive or ecologically valuable’ areas and ‘species of special concern’. The seventh is to buffer against unforeseen management mistakes. In a previous paper, she lists 10 similar, but somewhat different objectives for MPAs (Table 2.1).

### Table 2.1. Objectives of marine protected areas (Agardy 1993)

<p>| | |</p>
<table>
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<tbody>
<tr>
<td>1</td>
<td>To safeguard traditional sustainable uses.</td>
</tr>
<tr>
<td>2</td>
<td>To serve as centres for public education and schooling.</td>
</tr>
<tr>
<td>3</td>
<td>To act as models for training programs in coastal zone management.</td>
</tr>
<tr>
<td>4</td>
<td>To serve as research stations for monitoring and ecological research.</td>
</tr>
<tr>
<td>5</td>
<td>To provide controlled habitats for ecological restoration.</td>
</tr>
<tr>
<td>6</td>
<td>To guarantee public access to shorelines.</td>
</tr>
<tr>
<td>7</td>
<td>To institute a means to limit entry to an area or to a particular group of users.</td>
</tr>
<tr>
<td>8</td>
<td>To facilitate the political empowerment of local users who might not otherwise be represented.</td>
</tr>
<tr>
<td>9</td>
<td>To allow coordination of existing management facilities.</td>
</tr>
<tr>
<td>10</td>
<td>To provide a salient example of how to achieve sustainable use of coastal marine resources.</td>
</tr>
</tbody>
</table>

Pollard (1977, 1980) summarises the objectives of MPAs as areas for: 1. conservation of ecosystems, genetic diversity and fauna and flora to repopulate surrounding areas; 2. provide scientific undisturbed representative areas for applied ecology on the effects of human activities and pure ecological and biological studies; 3. education for students and instilling conservation values in the general public; and 4. passive recreational use. However, he points out that there is considerable overlap among the categories and that arbitrary distinctions can be drawn within categories.

Cleary there are problems in defining how general and more specific criteria for protected areas are defined within a similar, consistent framework. Slocombe (1998) recommends several desirable characteristics for goals and objectives based on the theory and practice of ecosystem management (Table 2.2). He concludes that “ecosystem management needs a linked set of criteria and goals that vary by place, scale and time that are pursued in an ongoing, adaptive process.” These characteristics are strongly reflected in the objectives and recommendations of this thesis and the way in which goals and criteria are modelled and analysed in the following sections.
Table 2.2. Desirable characteristics of ecosystem management goals and objectives based on the theory and practice of ecosystem approaches and management (Slocombe 1998).

1. Imply and reflect specific values and limits (normative).
2. Reflect ‘higher’ values and ethical principles and rules (principled).
3. Reflect the wide range of interests, goals and objectives that exist (integrative).
4. Work with, not artificially reduce, complexity (complex).
5. Accept and recognise the inevitability of change (dynamic).
6. Synthesize a wide range of information and knowledge (transdisciplinary).
7. Be applicable to a wide range of ecosystem types and conditions.
8. Involve actors, stakeholders, public (participatory).
9. Be inherently tentative and evolving as conditions and knowledge change (adaptive).

2.1 National and NSW goals and criteria for MPAs
The following assessments are based primarily on the Australian National MPA goals and identification and selection criteria (ANZECC 1998ab, 1999). These criteria, adopted by the NSW Marine Parks Authority (NSW Marine Parks Authority MPA Strategy Working Group 2001), were developed to promote consistency throughout Australian jurisdictions. They reflect over 30 years of international and national discussion, published research and practical management experience in protected areas (Ray 1975, Kelleher and Kenchington 1992, Ray and McCormick-Ray 1992ab, Thackway 1996, Davey 1998, Kelleher 1999).

Table 2.3 lists national goals and Table 2.4 lists national criteria recommended for the identification of marine protected area (MPA) options on ecological grounds. Table 2.5 lists national selection criteria recommended for the selection of MPAs from among the ecological options identified. These goals and criteria are the basis for the conceptual multiple criteria models described in the following section. These models also aim to include criteria from other overseas studies, MPA policies developed for NSW (NSW Marine Parks Authority MPA Strategic Working Group 2001), the objects of the NSW Marine Parks Act 1997, the NSW Fisheries Management Act 1994 and the National Parks and Wildlife Act 1974 and an environmental classification developed with Ron Avery and the NSW Marine Parks Scientific Committee.
Table 2.3. National goals for Australian marine protected areas (ANZECC 1998ab).

The **primary goal** of the National Representative System of MPAs (NRSMPA) is to establish and manage a comprehensive, adequate and representative system of MPAs to contribute to the long term ecological viability of marine and estuarine systems, to maintain ecological processes and systems, and to protect Australia’s biological diversity at all levels.

The **secondary** goals are to:

- promote development of MPAs within the framework of integrated ecosystem management
- provide a formal management framework for a broad spectrum of human activities, including recreation, tourism, shipping and the use and extraction of resources
- provide scientific reference sites
- provide for the special needs of rare threatened or depleted species and threatened ecological communities
- provide for the conservation of special groups of organisms – for example, species with complex habitat requirements or mobile or migratory species or species vulnerable to disturbance and which may depend on reservation for their conservation
- protect areas of high conservation value including those containing high species diversity, natural refugia for flora and fauna and centres of endemism
- provide for recreational, aesthetic and cultural needs of indigenous and non indigenous people.
Table 2.4. National identification criteria for marine protected areas.

<table>
<thead>
<tr>
<th><strong>1. Representativeness (Figure 2.3)</strong></th>
<th>Will the area:</th>
</tr>
</thead>
<tbody>
<tr>
<td>• represent one or more ecosystems within an IMCRA bioregion, and to what degree</td>
<td></td>
</tr>
<tr>
<td>• add to the representativeness of the NRSMPA, and to what degree</td>
<td></td>
</tr>
<tr>
<td>• reasonably reflect the biotic diversity of the marine ecosystems from which they derive?</td>
<td></td>
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</table>

<table>
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<tr>
<th><strong>2. Comprehensiveness (Figure 2.2)</strong></th>
<th>Does the area:</th>
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<tbody>
<tr>
<td>• add to the coverage of the full range of ecosystems recognised at an appropriate scale within and across each bioregion</td>
<td></td>
</tr>
<tr>
<td>• add to the comprehensiveness of the NRSMPA?</td>
<td></td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th><strong>3. Ecological importance (Figure 2.3)</strong></th>
<th>Does the area:</th>
</tr>
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<tbody>
<tr>
<td>• contribute to the maintenance of essential ecological processes or life-support systems</td>
<td></td>
</tr>
<tr>
<td>• contain habitat for rare or endangered species</td>
<td></td>
</tr>
<tr>
<td>• preserve genetic diversity</td>
<td></td>
</tr>
<tr>
<td>• contain areas on which species or other systems are dependant e.g. contains nursery or juvenile areas or feeding, breeding or resting areas for migratory species</td>
<td></td>
</tr>
<tr>
<td>• contain one or more areas that are a biologically functional, self-sustaining ecological unit?</td>
<td></td>
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</table>

| **4. International or national importance (Figure 2.3)** | Is the area rated, or have the potential to be listed on the world or a national heritage list, declared a Biosphere Reserve or subject to an international or national conservation agreement? |

<table>
<thead>
<tr>
<th><strong>5. Uniqueness (Figure 2.3)</strong></th>
<th>Does the area:</th>
</tr>
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<tbody>
<tr>
<td>• contain unique species, populations, communities or ecosystems</td>
<td></td>
</tr>
<tr>
<td>• contain unique or unusual geographic features?</td>
<td></td>
</tr>
</tbody>
</table>

| **6. Productivity (Figure 2.3)** | Do the species, populations or communities of the area have a high natural biological productivity? |

| **7. Vulnerability assessment (Figure 2.4)** | Are the ecosystems and/or communities vulnerable to natural processes? |

| **8. Biogeographic importance (Figure 2.3)** | Does the area capture important biogeographic qualities? |

| **9. Naturalness (Figure 2.4)** | To what extent has the area been protected from, or not been subjected to, human induced change? |
Table 2.5. National selection criteria for marine protected areas.

1. Economic interests (Figure 2.5)

Does the site:
- make an existing or potential contribution to economic value by virtue of its protection, e.g. for recreation or tourism, or as a refuge or nursery area or source of supply for economically important species
- have current or potential use for the extraction of, or exploration for, resources
- have importance for shipping and/or trade
- have importance to traditional users including commercial fishers
- contribute to local or regional employment and economic development?

2. Indigenous interests (Figure 2.5)

Does the site:
- have traditional usage and/or current economic value
- contain indigenous cultural values
- have native title considerations
- have importance for maintaining indigenous ecological knowledge?

3. Social Interests (Figure 2.5)

Does the site have existing or potential value to the local, national or international communities because of its heritage, cultural, traditional, aesthetic, educational, recreational or economic values?

4. Scientific Interests (Fig. 2.5)

Does the site have existing or potential value for research or monitoring?

5. Practicality/feasibility (Figure 2.4)

Does the site:
- have a degree of insulation from external destructive influences
- have social and political acceptability, and a degree of community support
- have access for recreation, tourism, education
- have compatibility between an MPA declaration generally and its uses
- have relative ease of management, and compatibility with existing management regimes?

6 Vulnerability assessment (Figure 2.4)

Is the site vulnerable and susceptible to human-induced changes and threatening processes?

7. Replication (Figure 2.4)

Will the site provide replication of ecosystems within the bioregion?
2.2 Conceptual multiple criteria goals for MPAs

MPAs should be selected and managed according to the reasons for which they are established. To implement general goals requires an interpretation of what the goals aim to achieve and these goals should also form the basis of subsequent performance assessments. The practical application of such guidelines requires specific criteria that should also relate directly to more universal goals. A hierarchical model can assist in describing these relationships.

The following multiple criteria model (Figures 2.1 to 2.5) was built using the ‘brainstorm’ function in Criterium Decision Plus (InfoHarvest 2000). The model expresses general goals as a functional hierarchy of successively more detailed criteria. The model is object oriented in that each criterion inherits the general qualities of the parent criteria higher in the hierarchy. Modeling relationships between goals, criteria and available information explicitly maps the chain of logic between what is ideally required, what may be possible and ultimately what is actually achieved.

For the assessments in Chapters 6-9, these models were used to evaluate goals and criteria in terms of specific measures derived from available ecological data (Figures 2.1-2.5). In this chapter, the model is used to systematically describe some of the goals and criteria that would ideally be considered to identify and select MPAs.

Figure 2.1 to Figure 2.5 represent one possible hierarchy of specific criteria nested within more general criteria and an overall goal. Annotations in Table 2.4 and Table 2.5 link all national goals and criteria to the model hierarchy in Figure 2.1 to Figure 2.5. The model is not meant to be exhaustive and in the interest of simplicity, an attempt was made to limit criteria to the most relevant. Structural relationships in the hierarchy were simplified, where possible, to produce a parsimonious model that might be easily understood and applied. Alternative models are possible and more extensive explorations of these are considered part of the approach.

The model may appear to include more criteria than can be practically applied, but most of these criteria will at some time, require consideration in decisions for MPAs. To exclude criteria, risks having unrecognised criteria influence decisions without being formally recognised. Where detailed data is unavailable, heuristic judgements can be made. Comparisons between the conceptual and applied models also provide a systematic way to assess data limitations and identify where more information is needed.
Goals and criteria for Marine Protected Areas

Figure 2.1. Primary and secondary goals for a system of marine protected areas.

Figure 2.2. Criteria for comprehensiveness.

Figure 2.3. Criteria for representativeness.
Figure 2.4  Criteria for adequacy.

Figure 2.5  Criteria for human activities.
2.3 Protection of biodiversity and ecosystem viability

The multiple criteria models group the identification and selection criteria for MPAs into two main branches (Figure 2.1): primary goals to protect biodiversity and ecosystem viability and secondary goals to provide for human use. Ecological criteria for the broad scale assessments are organised under three main branches: comprehensiveness, representativeness and adequacy. The broad scale assessments for the Manning, Hawkesbury, Batemans and Twofold Shelf bioregions were constrained to identify options for MPAs using only these ecological criteria. In Chapters 6-9, this conceptual model is used to apply the Simple Multiple Attribute Rating Technique (SMART) to assess general MPA goals as a function of standardised criteria scores for alternative MPA sites.

However, more detailed site assessments of fine scale ecological data and social, economic and cultural values were planned once broad options were identified. These criteria are, for example, incorporated in the assessment for the Cape Byron Marine Park Zone Plan described in Chapter 10. In the following sections, I describe the rationale behind the main ecological criteria and also the criteria addressing social, economic and cultural values.

2.3.1 Comprehensiveness

Comprehensiveness is defined as including ‘the full range of marine ecosystems and habitats’ within MPAs (ANZECC 1998ab). Strictly speaking, ecosystems and habitats are too complex and dynamic to define and map accurately. However, ‘surrogate’ measures can be used to approximately map generally recognised broad scale patterns in physical and biotic environments as a proxy, or indicator to represent coarse scale patterns in biodiversity. The ‘ecosystem’ surrogates are defined as types of estuary classified by Roy et al. (2001) and offshore depth zones (0-20 m, 20-60 m, 60-200 m and >200 m). The ‘habitat’ surrogates are defined as the environments provided by seagrass, saltmarsh, mangrove, rocky shore, reef, beach, islands and sediments. These ‘ecosystems’ and ‘habitats’ (Figure 2.2) were defined in an environmental classification based on broad scale differences in geomorphology, depth, substratum and exposure. The classification is essentially two-dimensional in definition and application. However it does seek to include the influence of three-dimensional features and processes throughout the water column, particularly in the effects of islands, different estuary types, offshore depth zones and the influence of features such as the East Australian Current. The classification only applies to the relatively shallow waters of the continental shelf, in deeper waters a vertically stratified classification might be appropriate as has been applied elsewhere.

The classification was developed with Ron Avery and discussions with NSW Marine Parks Scientific Committee. The largely physical differences in these environments are assumed to reflect a corresponding diversity in different habitats, species, and ecological processes. These assumptions are based on scientifically documented patterns observed for species studied in
NSW waters or similar locations and are summarised in Chapter 4 and elsewhere (Avery 2001, Breen et al. 2003, 2004, 2005, 2006). The aim of the following assessments is to represent all of these features in MPAs for each marine bioregion defined in NSW by Pollard et al. (1997) for the Interim Marine and Coastal Regionalisation of Australia (IMCRA 1998).

2.3.2 Representativeness
Representativeness is taken to mean that areas included in MPAs should ‘reasonably reflect the biotic diversity of the marine ecosystems from which they derive’ (ANZECC 1998ab). That is, while comprehensively sampling the range of biotic variation, MPAs should also include a reasonably unbiased and sufficiently large, representative proportion of the variation within this range.

The intention of this approach is to protect typical species, processes and areas as well as well known, charismatic, rare, threatened, scenic, recreational or convenient elements of biodiversity (Pressey 1995). A representative system of MPAs should protect both typical and ‘special’ components of biodiversity (Inglis 1992, Jones et al. 1992, Jones and Kaly 1998).

Figure 2.3 describes representativeness as a function of typical and special communities and species. In practice, typical communities and species may be represented through finer scale physical and biological surrogates, species assemblages, broad scale species surveys, incidental sightings and descriptive records of communities and populations.

Special species and communities in Figure 2.3 include rare, endemic, threatened, ecologically important, unique, productive and biogeographically, internationally and nationally important communities and species. In NSW, threatened communities and species include those communities, populations and species listed as endangered and vulnerable under the NSW Fisheries Management Act 1994 and the NSW Threatened Species Conservation Act 1995.

The life history characteristics of ‘potentially threatened’ species (Jones and Kaly 1998) should also receive consideration in locating and designing MPAs. These include species:

- with unusually restricted geographic ranges
- with unusually restricted breeding sites
- which are very large, long lived or have low fecundity
- subject to large-scale, mass mortality
- subject to prolonged periods of recruitment failure
- which are highly susceptible to stress
- which are extreme habitat specialists
- which are obligate supra-tidal, intertidal, estuarine and coastal embayment species
- species which are, or have been subject to over exploitation.
Noss (1990) also nominates species worthy of ‘special’ conservation status that serve as:

- ecological early warning indicators
- keystone species on which other diversity depends
- umbrella species whose large area requirements protect other species and
- flagship species that provide support for MPAs because of their public popularity.

One aspect of representativeness and adequacy not explicitly included in the conceptual model is genetic variation. The potential role of MPAs in maintaining genetic variation within and between marine species, populations and sub populations of fish and invertebrates is significant (Polunin 1983). Populations depend on this variation to survive disturbance, disease, competition and changes in their environment. For many endangered terrestrial animals and plants, the maintenance of genetic variation is the most critical factor affecting their survival.

A number of phylogenetic measures have been proposed that aim to implicitly include inter and intraspecific genetic variation in estimates of representativeness, but few have been implemented in the selection of marine protected areas (Vane-Wright et al. 1991, Faith 1992, 1994, Moritz 1994). While there is a tendency for marine protected area programs to represent biodiversity as coarse scale surrogates, consideration should also be given to the finer scale genetic diversity that maintains populations and drives the evolution of marine organisms.

### 2.3.3 Adequacy

Adequacy is defined as ‘the required level of reservation to ensure the ecological viability and integrity of populations, species and communities’ (ANZECC 1998ab). Adequacy includes criteria that affect the ability of MPAs to sustain the biodiversity they aim to conserve. It involves consideration of vulnerability, condition, reserve design, connectivity and practical MPA management (Figure 2.4).

#### Vulnerability

Vulnerability may be interpreted in two ways. Where there is a range of options available for the protection of a feature, it may be preferable to include areas that are least threatened and select locations where survival is more likely (Jamieson and Levings 1998). This approach may apply when threats originate from outside the MPA and are beyond the immediate control of MPA management. An example might be in selecting marine areas at locations less prone to pollution (Edgar and Barrett 2000) or less likely to be impacted by development.

However, where there are only a few examples of a habitat or species, there may be urgent reasons for protecting the areas most threatened, particularly where threats operate inside the MPA and are under some control of MPA management. This urgency would include habitats or species that might otherwise be lost without the protection of a MPA.
Condition

Condition or ‘naturalness’ reflects whether an area has already undergone some degree of impact. If an area has been affected by pollution, disturbance, pests, disease, habitat loss, or over-exploitation, the ecological viability of the area, as well as the diversity of organisms present may be affected.

Ecological reserve design

Ecological viability requires consideration of reserve design including the size, shape, replication and configuration of reserves within a network. Reserve design criteria aim to ensure that individual MPAs and the overall reserve system remain ecologically viable. There are many general recommendations for reserve design in the scientific literature (Ballantine 1991, 1997, Roberts 1998, Chiappone and Sealey 2000, Allison et al. 2003) and workshops where specific principles have been developed for a project (Day et al. 2002, Fernandes et al. 2005). These recommendations were reviewed for the bioregional assessments and used to help identify potential MPA sites (Appendix 1 summarised from Breen et al. 2004.)


Practical applications of this kind of information or theoretical models to MPA design and management are however rare. A combination of these approaches with the types of spatial models developed in this thesis may, in the future, provide a more rigorous approach to designing MPA networks.
**Management practicalities**

Management practicalities also affect the ability of MPAs to adequately conserve biodiversity. Criteria that need to be considered in identifying MPAs include:

- education (recognition of values, regulations and boundaries)
- cooperation (best practices, consultation, voluntary compliance, volunteer work)
- planning, regulation and enforcement considerations
- prospects for research and monitoring designs to aid adaptive management
- benefits from integrated ecosystem management of surrounding areas
- ease of administration, planning, permitting, impact assessment and finance, and
- political and community support to establish and make the MPA system work.

**Education**

For a system of MPAs to be effective, community support is essential. Support can only be gained if people are properly informed and educated about the value of MPAs. For management processes to be seen as transparent, people need to be made aware of the reasons for MPAs and how decisions are made. The complexities of MPA management can also lead to misinterpretation of management strategies. Education can help avoid confusion, create support and allay unjustified fears in the community (Alder 1996).

Some locations are particularly suited to educational activities and may already have programmes in place. Areas recognised for their high natural values are often good subjects for documentaries and printed articles that can be entertaining, informative and promote marine conservation to audiences internationally.

Providing information for local displays, tours, businesses, schools and other agencies provides tangible benefits to the community and opportunities for community input. In these instances, education can involve all age and community groups including children and the broader community, as well as those stakeholders most directly affected by MPAs. Some of the best education programs involve bringing people and marine ecosystems together and many MPAs are well known for this (Ballantine 1997).

**Planning, regulation and enforcement**

In an integrated system of MPAs, there needs to be coordination of planning and compliance among MPAs, and among management jurisdictions. In NSW, there are three agencies responsible for MPA management, and several other agencies with marine responsibilities. There is therefore, much scope for cooperation as well as potential for confusion over jurisdiction. In addition, responsibilities for marine bioregions in NSW are shared with the Federal Government (e.g. for part of Jervis Bay and waters more than 3 nm offshore) and with neighbouring state governments (for sections of the Tweed-Moreton and Twofold Shelf bioregions).
MPA design also needs to take into account strategies and restrictions already in place. In this way, they may take advantage of existing regulations, programs and facilities, avoid legal complications, and minimise additional impacts on existing use. Care should also be taken to ensure that ecological objectives are not compromised by differences in jurisdiction and that management includes negotiation among all agencies involved. For example, where MPAs are declared to the high water mark, measures should be taken to ensure that the mangrove and saltmarsh habitats inland of this boundary are also conserved. These considerations apply to a wide range of issues including catchment management, agriculture, development, fisheries, national parks, pollution, shipping, waste management and law enforcement. Opportunities for integrated management exist across all these areas in surveillance, research, monitoring, education, consultation, best practices, pest control, risk assessment and rehabilitation.

**Research**

MPAs have a crucial role as reference sites in understanding changing marine environments and the impact of human activities. Without reference sites where impacts are controlled, there are no baselines for distinguishing natural from human disturbances or for differentiating the causes of impacts from sources as diverse as fishing, land use, pollution, pests, development or climate change. Without this knowledge, our ability to detect problems, and develop and test effective solutions, is severely limited.

In particular, without consideration of experimental design in the identification, selection and design of MPAs, it may be very difficult to assess whether the reserves are even effective in achieving their objectives. Important considerations here are the replication of MPAs within a range of habitats and levels of protection, an interspersed allocation of these ‘treatments’ (Hurlbert 1984) and procedures to assess compliance (Davis et al. 2004) and ecological responses.

As the design of reserve networks and research programs share similar guidelines, even small alterations at the MPA design stage can have significant implications for future research and assessment (Kingsford 1999). The partnership between research and management should be regarded as an ongoing and iterative process of adaptive management to gradually improve the design and management of MPAs.

As research in marine environments is often difficult and costly there are significant advantages in cooperative research among MPA agencies and other agencies, universities, industries, organisations and individuals involved with marine environments. Consideration of existing research programs, infrastructure and expert knowledge can have important benefits for research, monitoring and conservation.
2.4 Managing and providing for human activities

Table 2.5 and Figure 2.1 and Figure 2.5 list criteria under the secondary goal to: ‘manage and provide for human activities’. Criteria for human activities are scheduled by national guidelines into a separate site ‘selection’ process. Where consistent with ecological goals, the selection process aims to minimise restrictions on human activities, and even enhance cultural, social and economic values. Often the ecological options for MPAs are flexible enough to allow for a variety of human uses.

Figure 2.5 lists just some of the interests potentially affected by MPAs. It is evident, even in this simplified view, that there is potential for conflict between conservation values and competing interests. Careful consideration of human activities is therefore required if MPAs are to be implemented. From a more positive point of view, the economic, social and cultural benefits from MPAs are often substantial. Nature based tourism and the associated services provide significant income and employment for many coastal regions (De Groot 1991, Agardy 1993, 2000, Dixon 1993, Driml 1994, 1999, Driml and Common 1995, Constanza et al. 1998, Cocklin et al. 1998, Access Economics 2005). The recognition that MPAs create for the conservation values of an area tends to promote MPAs as tourism destinations and also generates support for marine conservation and research generally. Where compatible with conservation goals, the selection and management of MPAs should therefore, seek to exploit this potential.

MPAs can also provide cultural benefits for local communities through economic opportunities and by protecting traditional use and spiritual values. However, for some communities, MPAs may also threaten these values. In most regions, there are now legal requirements for cultural involvement in considering locations for MPAs and careful planning is required to engage with traditional owners and custodians (Kenchington and Bleakly 1994, Smyth 1995, Schnierer and Woods 1998). Subject to intellectual property rights, indigenous knowledge should be included in MPA selection processes. In many areas local customs, cooperation and compliance are the main factors in establishing and maintaining MPAs (Kenchington and Bleakly 1994, Wolfenden et al. 1994, Gilman 1997) but careful consideration needs to be given to the manner and settings in which consultation occurs (Fiske 1992, Beaumont 1997, Helvey 2004, Crawford et al. 2006).

Information on human activities is also necessary to understand where impacts are likely to occur. Existing databases for permits (Alder 1993), charges and vessel logbooks (Valentine et al. 1997, Davis and Tisdall 1999, GBRMPA 1999), surveillance reports, surveys (Fleming 1991, Steffe et al. 1996, Queensland Transport 2001) and censuses can provide useful information to help anticipate potential impacts. Input from public consultation, interviews and other forms of social research can also provide important data on the distribution of human activities and on the aspects of marine ecosystems that people value most (Schafer and Inglis 1999, Williams et al. 2000, Breen 2006, Innes et al. 2006). As most decisions in selecting MPAs have a spatial component, it makes sense to map, or ask survey respondents to map, the
Goals and criteria for Marine Protected Areas

geographic distribution of these activities and the values they consider important. This information provides a direct quantitative input from individuals into planning the locations of MPA boundaries. It also enables social, cultural and economic values to be directly compared with ecological values measured at the same locations.

Stakeholders often spend much time observing marine ecosystems and can contribute valuable information on species distributions, habitats, vulnerability, condition and threats. When used cautiously, such information can provide important knowledge on local conditions, habitats and organisms (Johannes 1982, Calamia 1999, Berkes et al. 2000, Calheiros 2000, Johannes et al. 2000, Huntington et al. 2002). Williams and Bax (2002) for example, used data from commercial fishers’ logbooks and GPS plotters to map large areas of the continental shelf off southern NSW and Victoria. The fishers in this study provided boundaries of different habitats and descriptions of benthic invertebrates and fishes that were verified in subsequent scientific surveys using sonar and underwater video.

It is likely that, if designed accordingly, MPAs can benefit surrounding fisheries. However, the goal of enhancing effects such as ‘spill over’ was not specifically addressed in the identification or selection of MPAs in this thesis. This goal is currently not a priority in policy or research for MPAs in NSW, although it could create substantial support for MPAs. The contribution of MPA planning to economic (Dixon et al. 1993), social (Breen 2006) and cultural values is one of the least developed areas of MPA research and management and requires receive greater attention.

Davey (1998) lists eleven reasons why plans for MPAs fail, six of which involve stakeholder input:

- they do not address key issues
- they fail to involve stakeholders
- they rely too much on external experts and fail to involve local people
- they are weak on implementation
- they fail to raise political support for protected areas as a worthwhile concern
- they are poorly publicised.

There are many ways in which consultation can be enhanced through advisory committees (Vasseur and Renaud 1997), community meetings, information sessions, displays, the media and through the availability of staff for public communication (Innes et al. 2006). Effective consultation encourages public confidence and a sense of ownership and contributes to the effectiveness of MPAs in adequately conserving marine biodiversity. In Chapter 10, I describe some techniques to integrate social, economic and cultural information with ecological data during consultation for the Cape Byron Marine Park.
Marine research is constrained by factors such as weather, depth and the sheer scale of oceanographic systems. Even the best resourced research program can only hope to estimate some aspects of the ecology and biology of a few of the many marine species, habitats and processes. Most marine research has occurred at a limited number of sites sampled at select points in time. However, the extent of human influence, the potential for cumulative and synergistic impacts, and the need to provide convincing evidence to support management requires consistent information across large areas for a variety of ecosystem characteristics. This work also requires reliable systems, tools and effort to integrate and interpret data for scientists, managers and communities. This chapter describes how systematic methods in information collection, analysis and presentation can be used to help identify and select systems of MPAs.

3.1 Ad hoc and systematic selection of reserves

The selection of many protected areas has tended to be opportunistic or based on values for recreation, scenic beauty or other special characteristics. These so called *ad hoc* approaches have, in many cases, led to a relatively biased selection of areas that potentially neglects the conservation of less accessible, inconspicuous or less charismatic habitats and species (Pressey 1994a). If marine protected areas are to conserve a comprehensive range of ecosystems and species, a more organised approach to identification and selection is required.

The need for more transparent and accountable decisions also favours a systematic approach. Establishing and managing MPAs is often a difficult and costly process and the decisions made will affect the welfare of marine ecosystems and the human communities that depend on them. In many cases, there will be no second opportunity to correct decisions. It is therefore right to allow for sufficient resources, expertise and time to plan.

Many early developments in systematic protected area assessments occurred in terrestrial conservation planning in Australia. Typical problems included choices about which areas should be selected to protect habitats and species from clearing for timber and agriculture. The scale and urgency of these problems provided the impetus for many advances in applied conservation ecology, reserve design, and computer assisted decision support systems.

One of the most important principles to come out of this work is the need to protect representative areas of biodiversity. Pressey (1995) refers to protecting the ‘crown jewels’ of near pristine, scenic, highland wilderness habitats which are usually spared from land clearing and development but ignoring lowland habitats that may include a larger representation of biodiversity. He describes many of these flat, arable habitats as the ‘walking wounded’ and ‘irretrievably stuffed’.

"If the only tool you have is a hammer you tend to see every problem as a nail”

Abraham Maslow
This need to protect a representative range of biodiversity is a concept that has driven much of
the research to identify and map ecological ‘surrogates’ for biodiversity and the develop reserve
selection algorithms to efficiently represent this diversity within protected areas for the least
cost. The use of computer based methods is not essential to the process, but can make the
interpretation of large amounts of data less laborious and more accessible. The use of these
methods in planning for marine protected areas has not been widespread until recently.
However, an early application was initially tested alongside the planning process for the Cairns
section of the Great Barrier Reef Marine Park (Cocks et al. 1983) and Bakus (1982) explored
the use of multiple criteria analysis to select marine reserves. Other early assessments have used
a systematic approach to assess potential MPAs, albeit without the kinds of tools now available.

Ward et al. (1998) provide a general description and ‘tool kit’ for systematic reserve selection
which they term ‘Marine BioRap’, “a methodology and set of analytical tools for identifying
and assessing, in less than 18 months, priority areas of marine biodiversity.” The process is a
marine version of a terrestrial ‘BioRap’ process (Margules and Redhead 1995) which can be
summarised as follows.

1. Define objectives.
2. Review, choose and collate data
3. Select physical and/or biological variables to predict diversity.
4. Model the spatial distribution of this diversity.
5. Target minimum amounts of diversity to be represented.
6. Use algorithms and other tools to optimise representation and minimise cost.
7. Incorporate other ecological, social, economic, and management considerations.
8. Combine these techniques with consultation.

3.2 Representing spatial patterns in biodiversity

One of the more difficult problems in systematic protected area assessments is choosing a set of
variables that can be applied consistently across large regions to summarise the complex
diversity in marine ecosystems. For almost all situations these variables, often termed surrogates
or proxies for biodiversity, will represent only a small proportion of the full detail of species,
habitats and physical and biological processes. They are often based on predictions from a
limited data set of sparse observations. Their validity therefore relies on assuming that only the
most conspicuous and well known patterns will be adequately described. While surveys can
provide reliable information about some species at some places and times and perhaps
information on some related species, they are unlikely to be informative about other groups of
organisms (Faith and Walker 1994).
Figure 3.1 shows a conceptual representation of relationships among the environmental domains of a sample of fishes, other unsampled fishes and organisms, physical environments and other unknown factors. The implication is that while surrogates can infer distributions about some features, few will be comprehensive in capturing all variation.

Despite this, surrogates can be effective at describing gross spatial patterns in biodiversity providing that the assumptions linking the surrogate with the target biodiversity can be reliably demonstrated in some way. Surrogates for biodiversity can be grouped into environmental predictors, biological predictors, models that combine and interpolate physical and biological data and ‘delphic’ or expert consensus methods. These approaches are described in the following sections.

![Conceptual diagram of potential correlations and relationships among physical environments, biodiversity, unknown or unmeasured variables and one survey of fishes.](image)

**3.2.1 Mapping of physical and biogenic features**

This approach assumes, on the basis of ecological theory and research, that the physical or biogenic features mapped, will correlate with variations in ecological processes, habitats and the distributions of organisms. For broad scale patterns in biodiversity, general relationships of this kind are often well documented (Day and Roff 2000, Roff and Taylor 2000, Roff et al. 2003). It is reasonable, for example, to expect that the types of organisms in deep sea trenches will differ from those on continental shelves or in estuaries, or that assemblages of rocky shore species are distinct from the types of organisms found in seagrass beds or on coral reefs. Otway (1999), for example provides an example of how species richness increases as additional types of intertidal habitat are sampled.
A major advantage of this approach is that extensive physical surveys often already exist and can provide rapid, cost-effective predictors of major patterns in biodiversity. Sources of information include bathymetric models of the seabed, oceanographic surveys and geomorphological classifications of coastlines, estuaries and substrata. The increasing availability of technologies in sonar, aerial photography and satellite imagery mean that almost continuous coverages for some attributes can be obtained. This approach has the advantage of potentially capturing undescribed and perhaps unknown components of biodiversity (Faith and Walker 1996a, Vanderklift et al. 1998, Ward et al. 1999). It also has advantages in including a range of physical habitats and processes that may be important in maintaining biodiversity and evolution. For many areas, physical predictors will also provide a more stable predictor of long term patterns in biodiversity.

The disadvantage of this approach is that at finer scales, the relationships among physical and biological factors become more complex and include interactions with local biological effects such as competition, herbivory (Randall 1974, Potts 1977, Harmelin-Vivien et al. 1992), predation (Fairweather and Underwood 1991), recruitment (Williams 1983, Meekan et al. 1993) and biogenic modifications in habitat (Bologna 2000, Monteiro 2002, Hewitt et al. 2005). At local scales, physical predictors may only explain a small component of biological variation (Stevens and Connolly 2004). Ecological relationships may also involve indirect higher order interactions, synergisms and alternate states that may be dependent on a history of disturbance and other processes (Done 1992, Done and Potts 1992, Hughes 1994, Connell 1997).

Geographic overlays of several major environmental factors like temperature, depth and substratum may be useful in defining habitat differences among broad regions but simply intersecting an increasing number of physical variables to predict species assemblages at smaller scales may not be so successful. The key here may be to only rely on physical categories that can be clearly supported by documented differences in biodiversity and ecological processes. These categories are often well defined, and may also be easily recognised and accepted by the broader community. However, while they may describe conspicuous differences in biological assemblages, they may still fail to represent subtler differences in the distributions of many species and ecological processes. Examples of these include the differences in sediment infauna and epifauna associated with small structural isolates on the seabed (Thrush et al. 2001, Hewitt et al. 2005) or in areas adjacent to reefs (Barros et al. 2001) or around seagrass (Tanner 2005).

3.2.2 Biological surveys sampling organism distribution and abundance.

Direct observations on the location, abundance and other characteristics of organisms range from incidental sightings, museum collections (McCarthy 1998) and commercial harvest data (Pease, 1999), to dedicated statistically designed surveys (Otway 1999, De’Ath 2000, De’Ath and Fabricius 2000, Ninio et al. 2000, Otway and Parker 2000, Gladstone and Davis 2003, Shears et al. 2006, Morrison and Carbines 2006).
Where available, data that is collected systematically can provide reliable information on the distributions of the organisms sampled, and these distributions may also indirectly include other organisms associated with the sampled biota. However, for surveys and particularly museum collections and incidental sightings, there are often biases in the taxa, locations and habitats sampled. Faith and Walker (1994) and others (Kati et al. 2004) have also highlighted how different groups of organisms with different distributions may not provide good indicators for each other.

Observed biological distributions may also not persist with time, and in this respect, physical classifications can provide a more enduring surrogate. The biggest problem with species data is that it can be difficult and expensive to collect and identify and that observations are usually more sparse than for physical surveys. Both physical and biological data will however, usually require some level of interpolation and remote sensing for biological data is improving and being integrated with biological field survey techniques (Davis et al. 1990, Stoms and Estes 1993).

3.2.3 Modelling from biological, physical and spatial predictors.

Where data is sparse, it may be useful to interpolate patterns across the region of interest. Both physical and biological data sets are composed of individual sample points, although the distributions of biological data points are often more sparse and more irregular. Both however, usually require interpolation with, preferably, an estimate of the uncertainty of the predictions. The danger is that interpolations can give the impression that more is known than is the case.

Interpolation can be done subjectively, but there are a range of mathematical interpolation and smoothing methods that can be used. Interpolation may be based on correlations among physical data or among species data alone. However, where relationships can be established between physical predictors and species distributions, these correlations can be used to interpolate species occurrences at sites where only physical data is available. Where the prediction of the spatial distribution of organisms is the main goal, regression against spatial coordinates alone can be very effective (Legendre 1994, Fernandes et al. 2005). However, modelling the functional relationships between physical predictors and species assemblages can also provide information on the underlying environmental causes of species distributions (Nicholls 1989, 1991, Swartzman et al. 1994, Leathwick et al. 2006a).

Methods to identify MPAs

Fortin, Dale et al. 2002, Fortin and Dale 2005). Several of these techniques are able to fit non-linear and higher order interactions. Non-parametric permutation tests (Anderson 2000), likelihood estimates and Bayesian analyses (Hobbs and Hillborn 2006) can also assist in dealing with some of the other problems associated with ecological data.

The prediction methods chosen depend on the nature of the data and on what purpose is intended. For example, reducing the dimensions of multivariate data into groups of species or regions of similar habitat or fauna can clarify the dominant patterns in complex data sets. These summaries are useful for providing readily interpreted, graphic representations of patterns in biodiversity for planning, management, education and awareness. The resulting bioregionalisations are often used as the foundation for conservation planning but can become embedded in management policy as standard universal measures of biodiversity for conservation targets. It therefore needs to be recognised that they necessarily impose a reduction in the information available for subsequent analyses and decisions and may mask patterns in individual variables and neglect unsampled species.

On the other hand, most reserve selection algorithms, for example, implicitly aim to maximize representation for many individual habitats or species. A prior reduction in the dimensionality of the data for these methods may therefore be unnecessary and can result in subtle differences being ignored. In selecting a bioregion or other category, it may also be difficult to determine which particular species or other features are being targeted. Leathwick et al. (2006b) found for example, that using the predicted distributions of individual fish species as data in reserve selection algorithms generated a more representative system of MPAs for the New Zealand Exclusive Economic Zone than using a single compiled biological or physical regionalisation.

Algorithms may also give undue emphasis to the boundaries between classified bioregions which are often approximate and open to interpretation. The boundaries of the Interim Marine and Coastal Regionalisation for NSW (IMCRA 1998) for example, define five bioregions in the state. However, analyses for the initial derivation of these bioregions (Pollard et al. 1997) and subsequent studies (Pease 1999), define several alternative regions depending on the taxa and physical features analysed and for many years incorrect boundary locations were commonly reported before significant errors in coordinates were even detected.

Faith and Walker (1996a), Belbin (1995) and Pillar (1999) also note that categorical summaries lose information on the differences that exist between and within categories or regions. Different clusters or regions are sometimes assumed to be homogenous and equally distinct from each other when this is unlikely to be true. Faith and Walker suggest that this can be avoided by modelling variation as a continuous gradient and using a selection approach that minimizes the sum of the environmental dissimilarities between points and areas within the reserve system.
There are also methods that make use of indicator (MacNally and Fleishman 2004), keystone (Hurlbert 1997, Edinger and Risk 2000), umbrella and flagship species (Mouillot et al. 2002). These and other biodiversity indices (Foggo et al. 2003) may be useful in the right circumstances, but unless verified, may not provide reliable predictors for other components of biodiversity (Hurlbert 1971, Kershaw et al. 1995, Gibbons et al. 1997).

3.2.4 Delphic consensus of experts
This technique, in various forms, involves providing people with the relevant expertise, a set of objectives and having them reach a consensus in opinion. According to McArdle (1995), the method has its origins with the

“...Pythia, the prophetess at Delphi (who) produced uttered incomprehensible gibberish while stoned out of her mind on burning laurel leaves (cyanide poisoning), which obliging and politically astute priests of Apollo interpreted for the supplicant so that whatever actually happened, no blame could come on them, the original teflon bureaucrats. The thought of groups of marine scientists getting stoned out of their minds at meetings, and the hovering bureaucrats interpreting the resulting advice to their ministers so that none of the blame would attach to them is too far fetched...”

Despite this somewhat flippant interpretation, McArdle goes on to say that the experience and knowledge in such forums can complement other data and analyses which in turn can provide a core for discussions to focus on. In practice therefore, it may be useful to combine elements of all of the above approaches, providing that data and expertise are available. In most cases, there will be range of experience, information and classifications that can all provide meaningful input to planning. Management programs often attempt to seek the one ideal data set, classification or method that will explain everything. While this may seem more efficient and provide results that are easier to interpret, it may sometimes be better to work with complexity, rather than ignore information on the grounds that there is only one correct approach.

3.3 Decision support tools for MPA planning
Decision support is about providing and organising information to assist in decision making. In its broadest sense it can range from literature reviews, database reports and GIS displays, to computer assisted models analysing multiple criteria and providing scenarios for alternative decisions. Decision support is meant only to facilitate human decision making but is useful where large amounts of information and complex problems are involved. Identifying options for MPAs from a potentially large range of possibilities, criteria and information sources is one management area likely to benefit from decision support.

The following sections review some of the decision support tools that have been useful in conservation planning. These include GIS, simple scoring techniques, reserve selection algorithms and multiple criteria analysis. Other potentially valuable techniques include the use

### 3.3.1 Geographical Information Systems and databases

The primary question in identifying where MPAs are to be located is defined spatially. The aim is to identify the best areas to locate MPAs to conserve biodiversity given that protection will come at some cost and opposition from competing forms of use. Another consideration is how to synthesise within one framework, a range of information derived from many different disciplines, data sources and formats. A third basic problem is how to communicate this knowledge to managers, stakeholders, communities and politicians for many different locations, situations and scales of observation.

One development in the last 20 years has done much to address all three of these challenges. Geographic Information Systems are unique in that they mathematically map points, lines, shapes, regions, grids, processes and three dimensional objects in geographic space and can link each of these features (topology) to many different quantitative and qualitative data sets of descriptive attributes. This GIS environment is therefore capable of representing, in realistic detail, the landscapes, habitats, species, processes and human values that comprise ecosystems and link these directly with the mathematical and descriptive tools of modern science.

The raster grid cell capabilities of GIS also enable remotely sensed data from satellite, aerial photography, radar, sonar and video to be directly imported and geographically aligned with other spatial data sets. Directional and temporal attributes in line features can be used to represent flows and processes and the movements of organisms (Klimley et al. 2001, Willis et al. 2001) and propagules. When coupled with remote sensing systems, GIS can be used to quickly and accurately map large areas of coastal and submerged habitats and species using high resolution photography (Ekebom and Erkkila 2003), radar (Mason et al. 2001), LIDAR (Francis et al. 2005), multi-spectral camera, single beam (Pitcher 1999), sidescan sonar (Bickers 2004), multi-beam sonar (Wright et al. 2002) and video (Kostylev et al. 2001, Cappo et al. 2003, Fitzpatrick 2003, Jordan and Barrett 2003, Hewitt et al. 2004, Parsons et al. 2004, Morrison and Carbines 2006).

Grid data and vector (shape) data can be easily coupled to most other databases or modelling tools. Both are compatible with a range of mathematical modelling techniques including fuzzy set methods (Jenkins 1999, Zeng and Zhou 2001), expert systems, gap analyses (Bushing 1997, Powell 2000), individual based models, statistical models (Skidmore and Gauld 1996, Stoner et
Methods to identify MPAs


Measurements from non-GIS data sets that have at least some locational information can be readily linked to sites or approximate planning units to create new multivariate datasets derived from many different data sets. In this way, a wide range of information for different species, habitats, social, economic and management values can be integrated within the same model (Mallawaarachchi et al. 1994, 1996, 2001). GIS provides an ideal spatial environment to view the results of multivariate statistical analyses (Bollard-Breen 2006), predict and interpolate spatial distributions for species and environmental values (Kerrigan et al. 1999, Leathwick et al. 2006a) and to design research and monitoring programs (Nicholls 1989, Belbin and Austen 1991).

GIS can also quickly generate large ‘site by attribute’ datasets for use in reserve selection algorithms and to display and manipulate results for different scenarios. GIS are now also being used in ‘participatory’ and exploratory analyses that enable scientists (Kerrigan et al. 1999, Lewis et al. 2003), managers, stakeholders (Pressey 1998) and communities (Bruce and Eliot 2006) to work together with large, ecosystem-scale data sets. The powerful visualisation capabilities of GIS in two and three dimensional spatial formats is realistic and easily interpreted (Goodchild et al. 2000). For most people, GIS provides a more intuitive way to view information on natural systems than more abstract representations like summary statistics, graphs, ordinations and streams of numeric values in database tables.

However, the acceptance of GIS into mainstream ecology and marine science has been slow. This is surprising given the spatial context of many ecological questions, the evolution of fields like landscape and seascape ecology (Ray 1991, Fairweather and Quinn 1992, Jones and Andrew 1992), and the development of methods in spatial statistics (Dale et al. 2002, Liebhold and Gurevitch 2002, Fortin and Dale 2005) and modelling (Guenette and Pitcher 1999, Walters 2000, Walters et al. 2000).

GIS has been most frequently applied in geography, geology and planning to map landforms, vegetation and human infrastructure with a tendency towards descriptive rather than quantitative analysis. In ecology, and particularly marine ecology, there has been a strong emphasis on experimental techniques. However, in experimental approaches space, as a variable, is often ignored or randomised to avoid confounding with the immediate variables of interest. This approach has extended beyond laboratories and standard treatment plots to field experiments and regional monitoring programs which, until recently, have tended to ignore their obvious spatial context.

Data from many regional programs can provide an important input to spatial models and broad scale conservation planning. For this to be done however, requires at least approximate spatial coordinates, and the systematic sampling of whole regions, not just isolated sites. It also
requires that data be made available to integrated programs that extend beyond the immediate focus of individual research projects and that these opportunities are recognised when initially planning and designing field studies.

Finally, GIS are the preferred tools to accurately map, edit, document and communicate the legal boundaries that define MPAs. The following chapters will demonstrate how GIS can be used to integrate data on ecosystems, develop models to assist in conservation planning and eventually establish MPAs to help conserve marine ecosystems and manage their use.

### 3.3.2 Planning units.

Planning units are predefined geographic areas used to integrate different data sets, compare values among regions and assess alternative plans. They are usually contiguous non-overlapping polygon cells or raster grids that partition the area of interest into a network of units that can be selected, compared, assessed and included or excluded from hypothetical reserve systems.

Each cell is assigned a unique numerical code, descriptive attributes and values, and spatial attributes of size, shape and location. Manually or automatically selecting a cell or group of cells also selects the values associated with the planning units for display or as inputs into analyses. Planning units may be any shape but can be:

- square, hexagonal or other regularly shaped cells
- irregular polygons based on natural or man made boundaries or
- combinations of both regular and irregular cells.

The size and shape of the planning units should reflect the scale and accuracy of the information used, the scale at which MPA options need to be identified and the computational limitations of the software and computers employed. The use of regular cells of equal area may mean that initially, most units have an unbiased opportunity to include conservation features. Smaller irregular cells are less likely to include large areas of different features, however their shape can be used to closely follow natural landforms, jurisdictions, or other boundaries (e.g. high tide mark). This can be important, when tailoring specific, realistic reserves for some audiences.

Empirical trials have shown that planning unit size can significantly affect the outcome of analyses. Where a selection method is required to reach a critical threshold, larger planning units are more likely to ‘overshoot’ area targets (Pressey and Logan 1994, 1998). Smaller planning units, may however, not be large enough to include more than one conservation feature. In the latter situation, the selection methods may be unable to discriminate among the relative value of alternate units and selection can become arbitrary. It is therefore worthwhile to trial a range of planning unit sizes and it may even be worth considering different units for different purposes within the same project.
3.3.3 Simple scoring and graphic techniques.

If goals and criteria have been defined, and information or expert opinion is available, locations can be assessed using quantitative scores, ranks or qualitative values. Tables of individual values can be used to help identify suitable areas or aggregate totals of weighted or unweighted scores can be used as a measure of suitability (Rabe and Savage 1979, Purdie 1987). In NSW, this approach has been used to assess estuarine and intertidal aquatic reserves (Otway 1999, Frances 2000), coastal lagoons (Healthy Rivers Commission 2002) and estuaries (Bell and Edwards 1980, Digby et al. 1998). Conservation values can also be displayed on charts but graphical methods can be limited in situations where there are large numbers of sites, variables or categories.

Displays of these measures for planning units in a GIS allow spatial patterns to be rapidly identified at a range of scales. Pantus (1998a) for example, developed a prototype GIS based scoring model (MARES) as a stand-alone ESRI MapObject application for the Great Barrier Reef Marine Park Authority (Figure 3.2). The model displays, colour-codes and updates tabled scores for criteria as different planning units are selected. This tool provided information on the areas and percentages of different bioregions and habitats represented in different marine park zoning options, as well as the potential costs for different commercial fisheries. This prototype, stand-alone GIS tool was designed to be widely distributed to managers, scientists, stakeholders and communities. The NSW National Parks and Wildlife reserve selection tool, C-Plan (NPWS 2001) has a similar capability but also estimates the statistical irreplaceability of a site in contributing new species and habitats to existing networks (Pressey and Nicholls, 1989). C-Plan and other related techniques are reviewed in the following section.
Figure 3.2. Screen view of the prototype MARES GIS decision support tool developed by Pantus (1998a). The display shows selected hypothetical no-take zones in yellow on a background map of the Queensland IMCRA bioregions. The spreadsheet shows the areas and percentages of bioregions selected, fisheries catches likely to be affected, threshold targets and performance in meeting or exceeding thresholds.
3.3.4 Selection algorithms, complementarity and irreplaceability

To meet criteria for representativeness, a system of MPAs should aim to represent a comprehensive range of different habitats and species. However, there are usually economic, social and political constraints on how much of the marine environment can be included within MPAs. There are now several computer assisted techniques that aim to represent multiple habitats, species or other conservation features while minimising costs to competing activities. These approaches will generally achieve these goals more efficiently than simple scoring techniques or ad hoc selection (Lomolino 1994).

**Maximum covering problems and minimum set problems**

Mathematical reserve selection algorithms either aim to select areas that maximise representation of a set of conservation features for a given cost constraint (a maximal coverage problem) or conversely, meet threshold targets for a set of conservation features while minimising costs (a minimum set or minimum area problem, Cabeza and Moilanen 2001). For situations involving a few areas and values, this can be done manually by inspection. However, with increasing numbers of sites and features, the problem rapidly becomes more complex and mathematical algorithms are best used to identify solutions from potentially millions of possible combinations of sites (Possingham et al. 1999).

The techniques can be categorised as either exact or inexact (Cabeza and Moilanen 2001). Linear integer programs use branch and bound algorithms to solve for the exact combination of sites that optimises representation for a given cost. This method was used by Kirkpatrick (1983) to identify representative areas of Tasmanian forest to be protected from logging. This approach has been demonstrated to provide optimal solutions for relatively small (Underhill 1994, Church et al. 1996) to medium size data sets (Fisher and Church 2005), but as the number of sites and features increase, the time taken to reach a solution can become impractical.

Inexact methods include iterative heuristic algorithms that repeatedly apply a set of rules in a step-wise manner to sequentially add sites to a reserve network until a stopping condition is met. Inexact methods also include ‘stochastic global search’ methods (Cabeza and Moilanen 2001) like the simulated annealing algorithm included in the reserve selection software Spexan and Marxan (Ball and Possingham 1999, 2000).

**Iterative heuristic algorithms**

In iterative heuristic algorithms, decision rules within each iteration are used to resolve ties between planning units and to prioritise criteria. Rules are applied in order of priority, so that if several sites of equal value ‘tie’ for first place according to one rule, a second rule is used to choose the best site using a different criterion.
Many variations and sequences of these rules can be used including, selecting those sites with the most unrepresented features (a greedy algorithm), the rarest features or the highest irreplaceability. Different rules can also be used to select for costs, vulnerability (Faith and Walker 1996b, Pressey and Taffs 2001) and reserve design by selecting the nearest sites to existing reserves (Nicholls and Margules 1993, Briers 2002).

As an example, these sequential decision rules might include the following:

Rule 1. Select the planning unit that includes the rarest habitat not already included.

Rule 2. If there is more than one unit that satisfies Rule 1, select the unit with greatest number of unrepresented habitats.

Rule 3. If there is more than one unit that satisfies Rule 2, select the unit that occupies the least area…and so on.

After finally selecting a site, scores are adjusted to take into account the features added to the reserve network and the rules are applied again. In this way, the algorithm adds sites that complement the features already represented in the network and prioritises those features required to meet targets.

**Irreplaceability**

The potential value of a site in meeting targets will change as each new site contributes additional features to the network. Pressey et al. (1994) define this value conceptually as ‘irreplaceability’, “...the likelihood that an area will be required as part of a conservation system that achieves the set of targets”; or “...the extent to which the options for achieving the set of targets are reduced if the area is unavailable for conservation.”

In explaining this concept Pressey et al. state that: “If an area is totally irreplaceable, then no matter how a system of conservation areas is designed for a region, it will have to include that area. Put the other way, if that area loses its conservation values, one or more of the conservation targets for the study area will become unreachable.”

Csuti et al. 1997 estimated irreplaceability by counting how frequently planning units occurred in different optimal solutions from a branch and bound algorithm. However, these algorithms can be slow for large problems. Ferrier et al. (2000) however, developed rapidly calculated statistical estimators of irreplaceability. These irreplaceability statistics are incorporated in the C-Plan reserve selection with links to GIS displays in ArcView.

**C-Plan**

C-Plan generates an initial view of the predicted irreplaceabilities of all plan units that rapidly provides an indication of which sites are likely to achieve targets efficiently. However, as sites are added to the reserve network, targets are gradually met for some features. As a consequence, planning units containing those features decrease in irreplaceability and this is reflected in
Methods to identify MPAs

negative changes in the colour scale of these units mapped in the GIS. Conversely, irreplaceability for units with features not represented tends to increase as options for reserves are gradually used up, and this is reflected by positive changes in their colour scale.

When all targets for features in a site have been met, irreplaceability for that site approaches zero and the colour of the planning unit in the GIS display fades to white. Units with unrepresented targets increase in value and are highlighted in the map, and units likely to contribute most towards specific, individual feature targets can also be identified and highlighted.

Site irreplaceability and summed irreplaceability, are two of several different measures that can be calculated. Site irreplaceability is a measure of the overall likelihood that an area will be required as part of a conservation system to achieve a set of conservation targets. Values for site irreplaceability range from totally irreplaceable (1.0) to zero irreplaceability (0.0) and sites can have any value between these extremes.

A unit with a site irreplaceability of 1.0 may indicate that a planning unit is irreplaceable for one or perhaps several conservation targets. However, summed irreplaceability is also related to how many conservation targets a planning unit is likely to contribute to. It is derived by summing the individual site irreplaceabilities estimated for each individual conservation target. Summed irreplaceability can therefore range from 0 to numbers greater than 1, depending on how many feature targets are set.

C-Plan allows an operator to add and subtract plan units to and from the reserve system while immediately seeing the effect of these changes on irreplaceabilities, percentage goals and the area protected for different features and sites. The program is therefore a powerful ‘participatory GIS’ tool that enables managers and community representatives to interact directly with the goals, data and spatial boundaries of proposed reserves. The program can also save a significant amount of time for operators and technical staff in building and analysing GIS coverages of alternative proposals.

**Marxan and simulated annealing**

Simulated annealing is a type of algorithm included in Marxan, a computer program adapted by Ball and Possingham (2000) for the Great Barrier Reef Marine Park Authority (Day et al. 2002). It is based on a program developed for terrestrial conservation (Spexan) and written in C, but derived from an earlier, less accessible version, SIMAN, written in FORTRAN at the Department of Applied Mathematics at Adelaide University.

Like C-Plan, Marxan works from a basic data matrix of values (areas or occurrences) for the conservation features (e.g. habitats or species) represented within each planning unit (site) and a related table of costs associated with the decision to include each plan unit in a reserve network. Recent versions of C-Plan are able to use the same main data files as Marxan and call and
display Marxan simulations from a C-Plan interface. The two programs complement each other in a number of ways. While both include a selection of heuristic algorithms, C-Plan allows a user to manually select, query, include and exclude plan units and rapidly update statistical irreplaceability maps in ArcView GIS.

Marxan however also includes the ability to influence the spatial arrangement of the planning units selected by the reserve design algorithm. This includes the capacity to:

- minimise boundary length to area ratios and produce a network of more compact reserves (that is less likely to be influenced by edge effects and may be easier to enforce and manage)
- specify minimum reserve sizes, numbers of reserves and the replication of features in reserves
- specify minimum separation distances between reserves for independent replication and representation across geographic gradients.

Algorithms in Marxan aim to minimise an objective function (Equation 1) of the sum of the costs of the plan units in a given reserve system and the sum of the penalties incurred for not meeting specified targets for conservation features.

Equation 1. \[ \sum_{\text{sites}} \text{Cost} + \text{BLM} \times \sum_{\text{sites}} \text{Boundary} + \sum_{\text{CFPF}} \times \text{Penalty} + \text{Cost Threshold Value}_{(t)} \]

The Cost of a plan unit can be measured as the area it includes or some other measure (such as fisheries catches). A cost can also be assigned to the boundary length (or any other boundary cost) of adding a planning unit. A coefficient known as the boundary length multiplier (BLM) adjusts the relative importance of minimising boundary length over other costs and penalties.

The Conservation Feature Penalty Factor (CFPF) is used to weight the relative importance of meeting targets for conservation features. The penalty is roughly the additional cost and modified boundary cost needed to represent features not already adequately represented in the reserve system. A very small or zero value boundary length multiplier tends to generate a reserve system that is highly fragmented but efficient in terms of occupying a small area. A large boundary length multiplier aggregates plan units into larger clumps, but sometimes at an increasing cost in area and, where the CFPF is low, possible failures in meeting conservation targets. The Cost Threshold Value is an optional feature that applies an additional penalty once a specified or time dependent threshold cost has been exceeded (Ball and Possingham 2000).

The program includes variations of different stepwise heuristic algorithms, an iterative improvement algorithm and a simulated annealing algorithm. The heuristic algorithms work by sequentially adding sites to a reserve network according to stepwise criteria until a stopping condition is met. Iterative improvement algorithms randomly add, subtract and/or swap plan units that improve an initial ‘seed’ network to find a local minimum.

Simulated annealing works in a similar manner to iterative improvement. However, in the early stages of simulated annealing, when a parameter described as the ‘temperature’ is set high, both ‘good and bad changes’ to minimise the objective function are accepted. As the algorithm
progresses through a set number of iterations, the ‘temperature’ gradually decreases, and changes that do not decrease the objective function are rejected more frequently until only improvements in the solution are accepted. By randomly accepting many different plan units early in the algorithm, the program avoids local minima and can potentially identify a greater number of near optimal solutions.

For problems with many sites, measures like irreplaceability and heuristic selection methods are more likely to find efficient solutions (represent more conservation values for less cost) than simple scoring or *ad hoc* approaches. However, unlike branch and bound methods, heuristic algorithms are less likely to find perfectly optimal solutions (Cocks and Baird 1989; Underhill 1994). However, they can rapidly find approximate solutions for relatively complex problems. In addition, these approaches may identify a potentially greater variety of near optimal, alternative solutions, which can be advantageous where flexibility is important.

If the simulated annealing algorithm is run repeatedly (e.g. 100 times), it will generate 100 near optimal solutions to the problem. This variety of options may therefore be more flexible in providing solutions to meet other design criteria and in suggesting compromises between conflicting conservation and stakeholder requirements. The frequency with which planning units occurs in a set of solutions from many runs also provides a readily interpreted measure of irreplaceability (e.g. 0-100% of runs) that can be mapped. The ‘best’ solution of all runs as well as any other near optimal solution can also be mapped individually.

**Zonation**

Other approaches to incorporating reserve design in selection algorithms are recommended by Moilanen *et al.* 2005. These are implemented in a backward step-wise algorithm called ‘Zonation’. The algorithm starts from a set of all sites, then iteratively discards low value sites from the edges of the remaining area to ‘maintain the structural connectivity’ of the remaining habitat. Sites are removed gradually, leaving the most important sites till last. A ‘nested zoning’ which reflects the order of site removal can then be mapped to indicate site priority (Moilanen and Wintle 2006).

Aggregated reserves can also be obtained by smoothing species distributions before selection procedures and incorporating probabilistic measures of uncertainty, persistence and dispersal (Araujo *et al.* 2004, Cabeza *et al.* 2004, Moilanen and Cabeza 2005). Species persistence can be iteratively defined as species’ responses to habitat loss change with the changing structure and quality of the evolving reserve network (Cabeza 2003, Moilanen and Wintle 2006).

For computational reasons, all of the above methods require specific thresholds or targets to be set for either costs or conservation features. These can be used prescriptively according to agreed policy or used to explore the consequences of alternative scenarios. Targets can be set identically for all features or set individually to reflect priorities for specific features and situations. While specific targets are required for the computation of most reserve selection
algorithms, reaching agreement among managers and stakeholders on the ‘correct’ targets to use can be difficult. While some standard targets have been recommended, these might logically be quite different for various habitats and species, for different contexts (e.g. targeting an absolute area or a % of a habitat, a park, a bioregion or some other region) or different purposes (e.g. fisheries or biodiversity conservation). A complete reliance on these statistics can also obscure the importance of other criteria (e.g. Figures 2.1 to 2.4) which might not be summarised so easily (Agardy et al. 2003).

In the assessments conducted in this thesis, specific thresholds were not pre-determined, but a range of suggested targets were trialled in consultation with managers and stakeholder representatives. These methods are well suited to exploring the outcomes of using a range of targets, although it is difficult to portray all scenarios within a written document. For simplicity, examples using arbitrary targets or a range of targets of equal value for each conservation feature are described. The methods however, allow for much greater flexibility and this capability is ideally used in conjunction with input from managers, scientists and stakeholders.

**Environmental Diversity**

Faith et al. (1996a) describe a quite different approach that avoids splitting continuous environmental variation into surrogate categories and having to set targets for an arbitrary number of landscapes or habitat types. They suggest deriving an ‘environmental space’ from ordinations of environmental factors and species distributions and using distances in this space to select sites that best span this environmental diversity.

The rationale is that “the number of species represented by a set of areas will be large to the extent that on average, the distance from any point in the space to its nearest protected area is small. The expected complementary value of an area (the relative number of additional species it contributes) is indicated by the extent to which addition of the area to a partial set reduces the sum of these distances.”

This is a special case of a ‘p-median criterion’ and forms the basis of a number of reserve selection algorithms which can also incorporate various costs, vulnerabilities and the results of multiple criteria analyses reflecting the preferences and attitudes of interest groups.

**3.3.5 Hierarchical multiple criteria analysis models.**

The conceptual ‘trees’ of MPA goals and criteria described in Chapter 2 can be used to model the degree to which alternative sites meet an overall goal calculated as a function of their scores against prioritised criteria. There are many forms of these analyses and they have been reviewed in detail by Mardle and Pascoe (1999). The methods have been widely applied in business management and environmental impact assessment (Edwards 1977, www.expertchoice.com), with some applications in fisheries (Mardle and Pascoe 1999) and in selecting protected areas (Bakus 1982, Fernandes 1996, Rothley 1997, Guikema and Milke 1999, Villa et al. 2002). The
techniques can incorporate weighting of criteria, calculation of tradeoffs, representation of uncertainty, sensitivity analyses of the relative influence of different criteria, and the ability to combine and assess alternative models, data and sources of opinion.

The simplest is the Simple Multi Attribute Rating Technique (SMART) which evaluates alternatives according to hierarchical tree of detailed criteria nested within several levels of more general criteria which ultimately converging to single, broad goal. For each alternative, standardised scores are assigned to the most specific criteria. The overall aggregate score in achieving the main goal is calculated as a function of the standardised scores for the many sub-criteria, weighted according to priorities assigned to criteria at each level in the tree. Guikema and Milke (1999) use this technique to prioritise projects for the New Zealand Department of Conservation.

The Analytical Hierarchy Method (AHP, Saaty 1980) is a similar technique which uses the dominant eigen vectors from matrices of pair-wise comparisons among criteria to determine the relative importance of weights and, or scores. The pair-wise comparisons are usually derived from questionnaires or interviews of stakeholders or others involved in decision making. This approach was used by Fernandes (1996) to assess community preferences in the management of Saba Marine Park and by Villa et al. (2002) to develop a zone plan for the Asinara Island National Marine Reserve.

In either method, the scores and priorities for criteria and alternatives can be derived from quantitative or qualitative data and the method permits information from widely different sources to be integrated within a single analysis. Priorities for criteria within each level of the tree may represent policy, law or stakeholder preferences or the reliability and relevance of different data sets for each criterion. Sensitivity and trade-off analyses can also be used to determine how the priority for a criteria would need to change before it produces a different outcome. In this way, criteria that are most likely to influence decisions can be isolated and alternatives likely to satisfy a range of different priorities can be identified as compromises.

In this thesis, the SMART technique is used to assess goals and criteria for all estuaries and sections of coast and ocean in the Manning Shelf bioregion. In the Hawkesbury, Batemans and Twofold Shelf assessments, this technique is used to compare nine different options for large marine parks.

3.3.6 Expert advice, anecdotal information and delphic workshops

The advice of scientists, managers, stakeholders and communities should be part of any process to identify and select MPAs. Much of the scientific information used to assess MPA options is collected for purposes other than the selection of MPAs. It is therefore prudent to have experts familiar with the methods and phenomena to interpret the strengths and limits of the data. They can also advise on additional sources of information and where data is sparse, they can provide
expert judgements on the predicted distributions of species, communities, processes and sites of special ecological significance. They may also be best qualified to recommend what protective measures are likely to succeed in conserving biodiversity and ecosystem processes.

For many criteria there will be few, if any, data. There may however, be substantial knowledge to be gained from unpublished scientific observations and the experiences of many stakeholders. To estimate social and economic costs, information is also required on the locations and natural features most valued by stakeholders. Systematic survey data on these values is rarely available at the scales required for the selection of MPAs, and thus the people involved in marine activities are often the best, and sometimes the only, source.

These ‘informal’ sources can accessed through surveys and interviews and through workshops of representatives. The main challenges here are dealing with the potentially large number of possible contributors, and obtaining an unbiased representation of views. Voluntary submissions are often required as part of official consultation processes. These provide an opportunity to obtain information on specific values and locations of concern to different individuals and groups. Such submissions can however, be dominated by those most likely to be affected or concerned by changes in management. They should therefore be interpreted cautiously and where possible, independently verified. Statistically designed social and economic surveys, field observations and other independent sources can help in checking and supporting anecdotal information.

Workshops require guidance and direction to be useful, especially given the number of individuals involved and their often varied backgrounds and perspectives. Clear objectives and terms of reference are needed and an independent facilitator is desirable. Good information, tools and specific tasks to complete can also help to focus discussion on achieving outcomes.

3.4 Regional marine biodiversity classifications

There is an increasing trend towards coordinated programs to identify and establish systems of marine protected areas. These include global assessments, regionalisations (Kenchington and Bleakley 1994, Kelleher et al. 1995), databases (Grassle 2000) and many programs aiming to establish networks of MPAs. For many areas, a classification of marine regions has been a first step in describing broad patterns in marine ecosystems (Ray et al. 1975, Dethier 1992, McDonald and Cocks 1993, Hamilton et al. 1995, Hamilton and Cocks 1995ab, Walls 1995, IMCRA 1998, Day and Roff 2000, Roff and Taylor 2000, Roff et al. 2003, Fernandes et al. 2005).

All of these regionalisations provide a simplified representation of the environment and presumably the associated biota. They are, as such, only approximate surrogates. It is often possible that several different versions of a regionalisation are developed that emphasise one or more characteristics. This is not surprising given the aim is to summarise patterns for many species, habitats and processes in just two dimensions. What is critical is whether the
regionalisations are fit for a particular use, and that they are understandable, explainable and defendable (Thackway 1995, 1996). In the following sections, I will review some national marine regionalisations developed for Australia, and the classifications and MPA assessment processes developed within several State and Commonwealth jurisdictions in Australian waters.

3.5 Australian national marine classifications

Several early marine and coastal classifications of Australian waters have been proposed on the basis of both physical and biological characteristics (Ekman 1953, Knox 1963, Wilson and Gillett 1971, Wilson and Allen 1987, Gill 1974, Galloway et al. 1980, McDonald and Cocks 1993). However, the first apparent marine regionalisation developed specifically for marine and estuarine protected area planning resulted from a workshop endorsed by the Council of Nature Conservation Ministers (CONCOM 1985).

It proposed a classification of three coastal and offshore zones with geographic regions in each zone. These were further subdivided by substratum and then by biotic descriptors (Table 3.1). This regionalisation was then modified by the Australian Committee for the World Conservation Union (ACIUCN) in their proposal for a national system of coastal and marine protected areas.

At their first workshop in 1994, the newly formed Australian and New Zealand Environment and Conservation Council (ANZECC) identified the need to develop a regional or ‘meso-scale’ regionalisation, to be known as the Interim Marine and Coastal Regionalisation of Australia (IMCRA 1998). This involved both Commonwealth and State agencies.

The Commonwealth defined provinces for the Australian coast and for offshore territories in the Antarctic Ocean and the Kerguelen, Christmas, Cocos, Macquarie, Norfolk and Lord Howe Islands and Elizabeth and Middleton Reefs. For the Australian continental shelf, the Commonwealth developed separate pelagic and demersal provinces for waters inshore of the 200 m isobath and offshore of 200 m. The inshore regionalisation included not just provinces, but also ‘biotones’ or regions of overlap where species assemblages underwent gradual change between provinces. These regionalisations were based primarily on differences in fish species composition and richness, physical oceanographic data and benthic topography and sediment type.

Each State developed meso-scale (100’s – 1000 km) regionalisations within each inshore province for all waters out to the 200 m isobath. The Queensland component of the inshore IMCRA was derived from a cluster analysis of eleven biological and physical variables assigned to 30 arc second grid cells (Page and Stevens 1995, Stevens 1995, Stevens 1998). The variables used included sedimentary basins, carbonate and mud fractions in sediments, bathymetry, cyclone incidence, rainfall, tidal range, reef morphology, mangrove, saltmarsh, littoral crab biogeography and hard coral genus richness.
In Western Australia, physical and biological data were used in delphic workshops to develop bioregions (Chevis 1995, Wilson 1995). In South Australia, physical and biological data and delphic workshops of experts were also used to derive bioregions (Edyvane and Baker 1995). In Tasmania, multidimensional scaling (MDS) was used to develop bioregions from temperature data and systematic sampling of biological communities (Edgar et al. 1997). The data collected in these surveys also provided a basis for the selection of MPAs and for subsequent monitoring programs (Edgar and Barrett 1997). In the Northern Territory, data for fish assemblages, mangroves and a range of physical characteristics were overlayed in a GIS to derive biophysical regions (Ferns and Bilyard 1995).

In Victoria, a biophysical regionalisation was supported by remote sensing, collation of existing seabed data (Jenkins 1999a, Jenkins and Catlin 1999) and ground truthing of coastal marine habitats to produce a habitat classification. This was subsequently used to help select a system of 13 ‘no-take’ Marine National Parks and 11 Marine Sanctuaries covering 540 km² or 5.3% of state waters (Ferns 1999, Ferns and Hough 1999). In NSW, Pollard, Ortiz and Pethebridge (1997) defined bioregions using multivariate analyses for a range of different physical and biological data sets including information on the distributions of fishes, invertebrates and algae.

The resulting combined, national IMCRA (1998, Figure 3.3) includes 65 different marine bioregions and provinces to help plan a national system of marine protected areas. By including the characteristic biodiversity of each bioregion within a network of MPAs, the program aims to ensure that marine ecosystems are effectively managed for the conservation of biodiversity and sustainable use.

Commonwealth jurisdictions are currently developing regional plans to manage a range of use using a variety of tools including MPAs. These plans use habitat classifications, biological data, maps of economic and social values and extensive consultation to develop strategies. A plan for the South East Region has recently been completed and work is now focussing on other areas. The Commonwealth has also implemented MPAs for specific purposes such as complementing state marine parks with Commonwealth Reserves in waters offshore of Lord Howe Island and the Solitary Islands and providing protection for a major Grey Nurse Shark aggregation site at the Cod Grounds, near Laurieton, off NSW.

The Commonwealth Great Barrier Reef Marine Park Authority also recently completed a systematic assessment and new zone plan for the Great Barrier Reef Marine Park. The new plan allocated additional ‘no-take’ zones increasing the area closed to fishing from 4.5% to 33% of the total area. These new zones aim to protect a more representative selection of biodiversity by including a minimum of 20% of the area of each of 70 bioregions developed for the assessment. The following section briefly describes the biodiversity classification phase of this project as a case study that incorporates many of the methods reviewed in this chapter and led to the development of the Marxan reserve planning software.
Methods to identify MPAs

Table 3.1. Proposed classification scheme to assist in the identification of major marine habitats for the selection of marine and estuarine protected areas: habitat categories (CONCOM 1984).

<table>
<thead>
<tr>
<th>Level 1</th>
<th>Geographic zone</th>
<th>Level 2</th>
<th>Substratum</th>
<th>Level 3</th>
<th>Biotic descriptor</th>
<th>Level 1</th>
<th>Geographic zone</th>
<th>Level 2</th>
<th>Substratum</th>
<th>Level 3</th>
<th>Biotic descriptor</th>
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<tbody>
<tr>
<td>A.</td>
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<td>B.</td>
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Figure 3.3 Interim Marine and Coastal Regionalisation of Australia (IMCRA).
3.6 Classification of marine biodiversity in the Great Barrier Reef World Heritage Area

This classification formed part of the Great Barrier Reef Marine Park Authority’s (GBRMPA) Representative Areas Program (Kerrigan et al. 1999, Day et al. 2002, Lewis et al. 2003, Fernandes et al. 2005) to assist in allocating varying levels of protection within different zones of this multiple use marine park. The classification phase was based on literature reviews of systematic protected area assessments (Margules and Redhead 1995, Phillips 1996, 1998, ANZECC 1998ab, 1999, Pantus 1998b, Ward et al. 1998, Breen and Lloyd 1999, Day and Roff 2000) and designed to provide information for systematic reserve selection tools, consultation and planning. The major planning question addressed was where to locate representative, highly protected ‘no-take’ areas and other zones in the World Heritage Area to provide for the conservation of biodiversity and the management of sustainable use.

This classification and overall planning process was overseen by a scientific steering group of representatives from regional scientific and tertiary education institutions with expertise across a range of disciplines in marine science (Appendix 5). This committee provided critical reviews of the methods proposed and support for the intent of the project.

An initial gap analysis to assess the representation of IMCRA bioregions within the existing system of zoning (Figure 3.4) indicated that no-take zones occupied less than 5% of the marine park and this area included mainly shallow coral reef and very little intervening sediment, deep reef or other habitats. Most of this protection was allocated among a few bioregions with the largest area of protection within a single cross shelf transect in the Far Northern Section of the park (Poiner et al. 1999). This analysis clearly indicated that the existing system of zoning was highly unrepresentative of the range of habitats and biogeographic regions. However, it was determined that a finer scale regionalisation than the IMCRA would be required to identify new representative areas for additional levels of protection.

A physical classification derived from GIS overlays of interpolated depth (Figure 3.5), slope (Figure 3.6), substratum (Figure 3.10), exposure (Figure 3.8) and temperature (Figure 3.9) was initially trialled. Depths estimated from a 30 arc second (~ 900 m) gridded bathymetry data set for Australia (Buchanan 1999) were used to map broad depth zones, and derive estimates of slope and aspect using the spatial analyst extension in ArcView 3.0 (Figure 3.7). Aspect was then used to classify the coast, islands, mid shelf reefs and outer reefs as exposed or sheltered. Previous projects had already digitised and classified most shallow reefs, coast and islands from satellite imagery and aerial photos. Maxwell’s (1968) ‘Atlas of Great Barrier Reef’ included maps of mud, sand and carbonate fractions which had previously been digitised for GIS. Approximate summer and winter mean isotherms (Figure 3.9) provided a general indication of seasonal differences in temperature between the northern and southern ends of the reef and inshore and offshore areas.
These five data sets were ‘unioned’ in GIS overlays to produce physical categories formed by
the intersection of the different classes in each data set. However, if more than a few classes
within each data set were used, many different categories and spurious overlaps resulted. If a
few broad classes in each data set were selected on the basis of their presumed biological
importance, this problem was greatly reduced, but the resulting patterns were still difficult to
substantiate in terms of species assemblages or ecosystem processes (Figure 3.10). Moreover,
geophysical classifications of the region had already been proposed (Hopley 1982, 1983, Hopley
et al. 1989) and it was apparent that extensive data sets for many taxa and descriptions
of cross shelf and other spatial patterns in biodiversity existed for at least some areas (Done
institutions in the region also had well developed research programs and scientists with
experience in these waters. Some of these scientists (T. Done, D. Williams and A. Ayling) had
already provided delphic maps of patterns in the distributions of corals, fishes and benthos for
the marine park and there were several regional data sets suitable for numerical modelling.

A systematic search for all available broad scale biological and physical data sets of the region
was therefore conducted in conjunction with interviews with over 70 marine science and reserve
design experts. A questionnaire (Appendix 6) was sent to the scientists prior to the interviews,
and the results recorded and transcribed for later reference. The interviews assisted in providing:

- access to additional data sets
- information on physical drivers limiting and controlling the distributions of different taxa
- maps of the distributions of biota and important feeding, breeding, migration or other
  special areas and threats to these biota and processes
- anecdotal information for areas (e.g. the continental slope) where few data were available
- reserve design requirements for different species
- software trials with DIVERSITY, C-Plan and Spexan (the precursor to Marxan)
- contacts with other scientists with information
- potential workshop participants
- research priorities and opportunities to address knowledge gaps.

From this process, over 80 different data sets were mapped and some additional projects were
funded to provide data from cross-shelf surveys of inter-reef seagrasses, algae and epifauna
(Coles et al. 2000), mapping of *Halimeda* bioherms from previous acoustic surveys (Drew and
Abel 1988), and collation of fare sheet, public works and other geological seabed data and
facies (Jenkins 1999b).
Methods to identify MPAs

Figure 3.4. IMCRA (marine) and IBRA (terrestrial) bioregionalisations for Queensland and previous GBRMPA zone plan with no-take areas protected from fishing coloured green (4.5%) and bright pink or orange (<1%) (D. Breen. Unpublished map GBRMPA 1999).
Figure 3.5. Gridded bathymetry (30 arc second) for the Great Barrier Reef region and adjacent offshore areas (data from Buchanan, Australian Geological Survey Organisation 1999).
Figure 3.6. Average slope derived from the 30 arc second gridded bathymetry data (derived from data from Buchanan, Australian Geological Survey Organisation 1999).
Figure 3.7. Aspect (degrees from north) derived from the 30 arc second gridded bathymetry data (derived from data from Buchanan, Australian Geological Survey Organisation 1999).
Figure 3.8. Exposure of coast, islands, midshelf and outer reefs modelled from aspect (derived from data from Buchanan, Australian Geological Survey Organisation 1999).
Figure 3.9. Ten year mean monthly isotherms for the Great Barrier Reef (data from CSIRO Division of Marine Research).
Methods to identify MPAs

Figure 3.10. Selected depth and sediment categories within the previous GBRMPA zone plan (derived from data on sediments digitised from Maxwell 1968, depths from Buchanan 1999, bioregions from IMCRA 1998).
Figure 3.11. Morphological reef types of the Great Barrier Reef (data from Hopley 1983).
Figure 3.12. Regionalisation of reef morphologies for the Great Barrier Reef (data from Hopley 1982).
Figure 3.13. Numerical classification of reef morphologies for the Great Barrier Reef (data from Hopley et al. 1989).
Methods to identify MPAs

The Authority contracted a biostatistician to undertake statistical modelling for those data sets with sufficient species observations throughout the region. Multivariate classification and regression tree analyses (MCART, De’Ath 1999, De’Ath and Fabricius 2000, De’Ath 2002) were used to relate species assemblage data for soft corals, hard corals (two data sets), fishes (two data sets), reef benthos, algae, seagrasses, epifauna (Coles et al. 2000) and sediments to two spatial predictors, cross-shelf and along-shelf position.

Coordinates for the centre point of each reef or inter-reef GIS polygon were first standardised on a scale across the shelf from between 0 at the coast and 1.0 at the 200 m isobath\(^1\). Positions along the longest axis of the region (approximately north north west) were standardised to between 0 at the far northern end to 1.0 at southern extreme\(^2\). These spatial predictors provided greater predictive power than other physical explanatory variables such as sediment or distance to river mouths. For each of the ten data sets, MCART analyses were used to successively split the region by cross shelf and along shelf positions into smaller areas containing increasingly homogenous assemblages of organisms until permutation tests indicated that predicted groups were not statistically different (De’Ath 1999).

The resulting models were then used to interpolate bioregions from the positions of reefs and inter-reef areas and assign levels of uncertainty dependent on the distance to the nearest survey site where data were collected. An aggregate MCART analysis for all reef organisms was then run using the membership of each site in different hard coral, soft coral, fish and benthic assemblages as the response variables (De’Ath 1999).

Separate two-day workshops were then convened for a panel of reef experts and a panel of inter-reef experts to interpret data and analyses and define and justify regionalisations that explained the dominant patterns in fauna, flora, habitats and processes throughout the Great Barrier Reef World Heritage Area.

The groups were provided with defined objectives and terms of reference and the results of the MCART analyses were explained by the consultant biostatistician, then reviewed, discussed and generally endorsed by the expert panels. Almost all data and analyses were mapped in ArcView and ARCINFO GIS and provided to the panels as large, A0 sized paper maps and reports that included maps, metadata and descriptions of all the data sets collected.

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\(^1\) i.e. \(x = (\text{distance of each point from coast}) / (\text{total distance between coast and 200 m isobath})\).

\(^2\) i.e. \(y = (\text{distance of each point from north end of reef}) / (\text{total distance along longest axis of the Great Barrier Reef})\).
Methods to identify MPAs

Table 3.2. Physical and biological data sets mapped and documented for the GBRMPA Representative Areas Program.

<table>
<thead>
<tr>
<th>Physical</th>
<th>Regionalisations</th>
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<tr>
<td>Depth, slope, exposure &amp; substrate overlays</td>
<td>48. Interim Marine and Coastal Regionalisation of Australia</td>
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<td>Regionalisation of reef morphology</td>
<td>49. Australian coastal regionalisation</td>
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<td>Numeric regionalisation of reef morphology</td>
<td>50. Delphic reef regionalisation</td>
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<td>Galloway coastal classes</td>
<td>Plants</td>
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<td>Bathymetry</td>
<td>51. Inter-reef algae</td>
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<td>Depth &amp; elevation model</td>
<td>52. Halimeda-bed coverages</td>
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<td>Gridded bathymetry (15 and 30 arc second)</td>
<td>53. Inshore seagrasses</td>
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<td>Seafloor aspect</td>
<td>54. Deepwater seagrasses</td>
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<td>Slope</td>
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<td>Exposure</td>
<td>56. Reef macro algae</td>
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<td>Queensland coastline</td>
<td>57. Soft coral surveys</td>
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<td>Intertidal areas</td>
<td>58. Hard coral surveys</td>
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<td>Sediment grain size</td>
<td>59. Long-term monitoring reef surveys</td>
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<td>Halimeda sediments</td>
<td>60. Surveys of reef biota</td>
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<td>Percent mud, carbonate, sand, gravel and rock</td>
<td>61. Museum collections</td>
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<td>Biological facies</td>
<td>Echinoderms</td>
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<td>Past shorelines and river valleys</td>
<td>62. Museum specimen data</td>
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<td>Estuaries</td>
<td>63. Echinoderms from Cairns Section</td>
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<td>Coastal rivers</td>
<td>Epibenthos</td>
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<td>Australian drainage basins</td>
<td>64. Cross transects off Townsville</td>
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<td>Islands</td>
<td>65. Far Northern Section effects of trawling survey</td>
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<td>19. Islands</td>
<td>66. GBR seagrass and interreef surveys</td>
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<td>20. Cays</td>
<td>Urochordates</td>
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<td>21. Island inventory</td>
<td>67. Museum collections</td>
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<td>22. Classification of islands in GBR</td>
<td>Molluscs</td>
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<td>Reefs</td>
<td>68. Museum collections</td>
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<td>23. Reefs</td>
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<td>24. Drying reefs</td>
<td>69. Northeast Australia surveys</td>
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<td>25. Named rocks</td>
<td>70. Museum collections</td>
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<td>26. Reef inventory</td>
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<td>27. Classification of reef morphology</td>
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<td>29. Reef shape</td>
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<td>30. Cross shelf position</td>
<td>74. Long term monitoring</td>
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<td>31. Long shelf position</td>
<td>75. Reef biota surveys</td>
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<td>33. Secchi depth</td>
<td>78. Pelagic fish - Billfish &amp; Marlin</td>
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<td>34. Extents of flood plumes from rivers</td>
<td>79. Museum collections</td>
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<td>35. Water quality sector model of GBR lagoon</td>
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<td>36. Regional seasonal ocean maps</td>
<td>80. Turtle nesting and movements</td>
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<td>37. Australian region oceanography dataset</td>
<td>81. Sea snake database</td>
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<td>38. Exposure to wind</td>
<td>Birds</td>
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<td>82. Seabird atlas</td>
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<td>39. 10 year mean temperature fields</td>
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<td>40. Sea surface temp effects on coral bleaching</td>
<td>83. Whales</td>
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<td>Tides and currents</td>
<td>84. Dolphins</td>
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<td>41. Regional hydrodynamics &amp; dispersal project</td>
<td>85. Dugong</td>
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<td>42. Coral reef and mangroves: modelling &amp; management project</td>
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Methods to identify MPAs

Figure 3.14. Scientists defining bioregions on an image of ArcView GIS projected on a whiteboard photocopier in a workshop for the GBRMPA Representative Areas Program.

All GIS data were preformatted and projected onto a photocopy whiteboard during workshops. Additional ArcView extensions were used to synchronise views for different data sets, record and return to defined extents, and rapidly record, edit and annotate the regions defined by the group. The expert panels were able to interactively edit and refine regions and provide descriptions and justifications for different boundaries and regions. This resulting bioregionalisations were recorded as they evolved and updated maps were provided to the panels for checking the following day.

A third, joint workshop of the reef and inter-reef panels was held to consolidate bioregions where possible and confirm the draft boundaries of the new joint bioregionalisation (Figure 3.15 and Figure 3.16, Kerrigan et al. 1999). Gap analyses indicated that most of the new bioregions were clearly under-represented in the existing network of highly protected areas (e.g. Figure 3.15). Participants were also able to identify sites of special significance for diversity, uniqueness, productivity, feeding, breeding and migration. The regionalisations were then reviewed externally and a number of changes made to different boundaries.

The expert groups also developed a set of reserve design guidelines or ‘biophysical operating principles.’ These included minimum specifications for size, shape, representation, configuration and replication of no-take areas in each bioregion. A workshop was then held to review reserve planning tools and two new tools, Marxan (based on the terrestrial software Spexan, Ball and Possingham 1999b, 2000) and Trader (De’Ath 1999), were commissioned for the project.
Methods to identify MPAs

The bioregionalisation and supporting data were then used with Marxan, Trader and social, economic and cultural information to plan and consult for a new representative zone plan for the marine park (Figure 3.17). The biodiversity classification process for the Representative Areas Program was only one stage of the zone plan review (Day et al. 2002). The subsequent reserve selection processes, tools, consultation, negotiations, assessments and mapping took a large team several more years to complete (Fernandes et al. 2005). However, the use of a systematic, scientifically justified process, the best available information and advice, and the support of the scientific community, provided substantial support for zoning decisions and a more effective basis for planning.

The process was remarkable for the synthesis of ideas that occurred among scientists from different disciplines and institutions at the workshops. In many cases, boundaries emerged with input from several individuals using data from different sources and from conceptual models of the likely responses of organisms to their environment. The situation was relatively unique in the way that data, experience and intuition were used with graphic tools to address specific tasks in a systematic and cooperative manner.

From a broader perspective, the program increased the awareness of many existing and new data sets and the value of using science to support conservation management. The benefits of these systematic techniques extend beyond the Great Barrier Reef Marine Park. Such tools and techniques are also being used to assist planning for MPAs in Moreton Bay (Buxton 2005) and elsewhere in Queensland (Banks et al. 2005), South Australia (Stewart et al. 2002, Stewart and Possingham 2003, 2005), Florida (Leslie et al. 2003), California (Airame et al. 2003), New England (Cook and Auster 2005), Wales (Richardson et al. in press), Fiji (D. Breen, unpublished data), New Zealand (D. Breen, unpublished data), and NSW (Breen et al. 2002, 2004, 2005, 2006).
Methods to identify MPAs

Figure 3.15. Draft reef regionalisation from the reef expert workshop for the GBRMPA Representative Areas Program and highly protected no-take zones from the previous zone plan for the Marine Park.
Methods to identify MPAs

Figure 3.16. Draft inter-reef regionalisation from the expert workshop for the GBRMPA Representative Areas Program.
Figure 3.17. Distribution of no-take zones (green) in the Great Barrier Reef Marine Park before and after the Representative Areas Program and the new zone plan in 2004 (Maps from www.gbrmpa.gov.au).

Statement of the contribution of others in Chapter 4.

Ron Avery and I together reviewed legislation, objectives and the application of MPAs in NSW and this is reflected in the content of the following chapter.
4 Marine Protected Areas in NSW

The earliest marine protected areas in NSW were, arguably, sections of terrestrial National Park that extended below mean high tide (McNeill 1995) and several of these were declared between 1967 and the early 1980s. However, it took some time before protected areas dedicated solely to marine conservation were created. Pollard (1997) provides a comprehensive account describing the background, history and justification for these MPAs and parts of this are summarised below.

The first proposal for a protected area dedicated to marine conservation was made in 1969 for parts of the Solitary Island group, north of Coffs Harbour on the northern NSW coast. However, it took over 20 years for a reserve to be declared at this site. Other early proposals for marine protected areas were made at Julian Rocks in 1972 (proposed by William Sylvester), at Long Reef in 1974 (Isobel Bennett), at Fly Point-Halifax in 1974 (D. Harris) and at North Sydney Harbour in 1979 (David Stead). In 1971, the NSW government extended Bouddi State Park over an area of inshore water to be managed jointly by the NSW National Parks and Wildlife Service and NSW Fisheries. The unique management arrangement at this site persists today with NSW Fisheries (NSW Department of Primary Industry) providing protection from fishing through closures which must be renewed each five years.

In 1980, Pollard notes that “NSW State Fisheries has for a number of years been carrying out a program of investigation into the suitability of areas along the NSW coastline as potential sites for marine and estuarine reserves…resulting in the identification of some 40 or so potential sites. Preliminary surveys of some 30 or so of these specific sites have since been carried out and 15 have now been surveyed in considerable detail.”

After amendments to the Fisheries and Oyster Farms (Amendment) Act 1979, the first Aquatic Reserve in NSW was declared at Long Reef (1980), shortly followed by declarations at Julian Rocks (1982), North Sydney Harbour (1982), Bushrangers Bay (1982), Shiprock (1982) and Fly Point / Halifax (1983). In 1987, a relatively large area (1400 ha) of mangrove, seagrass, saltmarsh and estuary in Botany Bay was protected in Towra Point Aquatic Reserve. In 1991, a large 100,000 ha MPA was declared in State and Commonwealth waters around the Solitary Islands, 22 years after the original proposal.

After the formation of the NSW Marine Parks Authority, large, multiple use marine parks were established at the Solitary Islands in 1997, at Jervis Bay in 1998, around Lord Howe Island in 1999 and then at Cape Byron in 2002. Additional small aquatic reserves were also established by NSW Fisheries at Cook Island in 1998, and at Barrenjoey Head, Narrabeen Head, Cabbage Tree Bay, Bronte-Coogee, Cape Banks and Boat Harbour in 2002. Many additional areas of national park and nature reserve now also include marine habitats, but only the Bouddi marine extension has any direct protection from fishing.
According to the definition adopted by the NSW Government (IUCN 1994), three types of marine protected areas now exist in NSW:

- **marine parks** – managed under the *Marine Parks Act 1997* by the NSW Marine Parks Authority. The Authority comprises the Director-General of the Premier’s Department, the Director-General of Department of Environment and Conservation and the Director-General of NSW Fisheries, Department of Primary Industry.

- **aquatic reserves** – managed by NSW Fisheries under the *Fisheries Management Act 1994*.

- the marine components of **national parks and nature reserves** – managed by NSW National Parks (Department of Environment and Conservation) under the *National Parks and Wildlife Act 1974*.

The locations of marine parks, aquatic reserves, and those sections of national parks and nature reserves established up until 2006 are shown in Figure 4.1. Since the completion of the assessments described in this thesis an additional two large marine parks were established in 2006 in the Port Stephens and Batemans Bay regions.

Prior to these assessments, legislation for the three types of MPAs was assessed for its ability to meet the goals and criteria described in Chapter 2. All three MPA types had significant powers to conserve biodiversity and ecological processes, but differed in how this could be achieved. It was apparent that under the right circumstances the different MPA types could complement each other in a network of MPAs.

Large, multiple use marine parks could be used to create extensive networks of no-take areas embedded within areas of sustainable, integrated management at key locations. Aquatic reserves could be used to address local issues, species and habitats at specific sites; and national parks and nature reserves could provide extensive protection for shorelines, catchments, wetlands and, with additional legislation adjoining areas of open water. The following section describes the objectives of the different types of MPA in NSW and how they are applied. Further information on MPAs in NSW is found in “*Developing a representative system of marine protected areas - an overview*” (NSW Marine Parks Authority MPA Strategy Working Group 2001) and in reviews by Parker (1995) and Pollard (1997).

### 4.1 Marine parks

The *Marine Parks Act* aims to conserve marine biological diversity, habitats and ecological processes in marine parks. Where consistent with these objectives, it also aims to provide for the ecologically sustainable use of fish and marine vegetation (including commercial and recreational fishing) and provide opportunities for public appreciation, understanding and enjoyment of marine parks. Integrated management of these marine parks also aims to assist in managing pollution, visitor use, activities on adjacent lands, marine pests and a wide range of human activities, environments and species.
Four marine parks were declared in NSW before the assessments described in the following chapters (Figure 4.1). These are large, multiple use MPAs, ranging from approximately 200 km$^2$ to 700 km$^2$ in area. These marine parks are zoned to allow for a range of human activities, including commercial and recreational fishing. They also include sanctuary zones where plants and animals are fully protected. Approximately 12% of the Solitary Islands, 20% of the Jervis Bay, 27% of Lord Howe and 27.5% of the Cape Byron Marine Park are zoned as highly protected, ‘no-take’ sanctuary.

As well as zoning and other forms of regulation, marine parks use permits, impact assessments, education, consultation, research, monitoring and best practices to manage not just what activities occur, but how activities can be carried out sustainably.

Large marine parks attempt to include a range of interconnected ecosystems and habitats, and aim to provide protection from external threats, protection for mobile or widely dispersed populations and have the capacity to manage a wide range of impacts. Including many features within a large marine park also provides for greater flexibility in multiple use zoning, with more opportunities to meet community and stakeholder requirements, while still meeting conservation goals.

### 4.2 Aquatic reserves

Aquatic reserves also aim to conserve the biodiversity of fish and marine vegetation, but may also be established for specific objectives including the protection of fish habitat, protection of threatened species, populations and ecological communities, and conservation for educational activities and scientific research.

There are currently 13 aquatic reserves in NSW. Most are relatively small (2–150 ha), and with the exception of Towra Point (1,400 ha) and North Sydney Harbour (250 ha), mainly protect rocky intertidal shores and inshore reefs. They have a role in complementing the range of ecosystems found in other MPAs, and in addressing specific local issues and ecological features.

### 4.3 National parks and nature reserves

Marine protected areas also occur where national parks and nature reserves are specifically gazetted over subtidal or intertidal areas. In NSW, there are around 50 national parks or nature reserves with recognised marine components. These areas can protect animals, vegetation and substrata, but cannot directly protect fish or aquatic invertebrates as defined in the *Fisheries Management Act 1994*. National parks and nature reserves differ substantially from aquatic reserves and marine parks in that there is no zoning or regulation for ‘multiple use.’

Conservation of marine species can be enhanced through the protection of habitat and the general management of use including control over moorings, motor vessel access and the construction of marinas and other structures, and by protecting adjacent terrestrial habitat.
Direct protection for fish and other marine life can be achieved through arrangements with NSW Fisheries. MPAs within national parks and nature reserves exist as components of a broader terrestrial reserve system. Generally, they include large areas of intertidal and subtidal estuarine habitat and smaller areas of intertidal and subtidal coastal habitat. Offshore ecosystems have not been included within these MPAs, and are not currently targeted for protection.

As a part of an integrated system of MPAs, national parks and nature reserves make a complementary contribution to comprehensiveness, particularly in coastal and estuarine areas. The number and area of MPAs in national parks and nature reserves is substantial and several are large enough to include a range of marine habitats and ecosystem processes. For example, Myall Lakes National Park extends over 97 km² of estuary and ocean coast, although this area is now included in the newly declared Port Stephens - Great Lakes Marine Park.

Because of their relationship with terrestrial reserves, MPAs in national parks and nature reserves bridge gaps in protection for saltmarsh, mangrove, rocky shore, beach and other coastal transition areas. The associated terrestrial reserves also provide protection for catchment and coastal ecological processes that critically affect biodiversity in the land and sea. In particular, the protection of shoreline and catchment from habitat clearing, development and pollution provides an important buffer for near shore environments. Agreements with other management agencies (e.g. NSW Fisheries) can provide direct protection for fish in MPAs and assist in integrating conservation strategies for marine and terrestrial environments.

The National Parks and Wildlife Act requires a plan of management to be prepared for national parks and nature reserves, as soon as is practicable after reservation. However, these plans do not usually deal specifically with the management of marine protected areas or the aquatic biodiversity present. Current issues for NSW National Parks in this area include the control of vehicles and dogs on shores and intertidal areas and the effects of these threats on seabirds, waders and other biodiversity.

4.4 Commonwealth MPAs

The NSW state jurisdiction includes only sea within 3 nautical miles of the coast and islands. MPAs beyond the three nautical mile limit of state territorial waters are managed for the Federal Government by the Department of Environment and Heritage. These areas include important habitats and ecological gradients across the shelf, shelf break and slope associated with a distinct but largely undescribed diversity of organisms and processes of widespread importance to oceanic ecosystems. Dramatic changes in depth and sea floor topography here create unique habitats for benthic invertebrates and demersal fishes and produce unique oceanographic conditions. Currents and up-welling in these areas have important roles in the feeding and migration patterns of many fishes, invertebrates, birds, reptiles and mammals.
Currently the state Solitary Islands and Lord Howe Island Marine Parks are declared out to the three nautical mile state limit but have complementary Commonwealth MPAs established further offshore. A small section of bay inside Bowen Island at the southern headland of Jervis Bay is also managed as an MPA ‘in sympathy’ with the adjoining Commonwealth National Park. A Commonwealth Marine Reserve has also been declared at the Cod Grounds, off Laurieton on the NSW north coast to protect the endangered Grey Nurse Shark (*Carcharias taurus*).

Systematic planning for the Commonwealth sections of Australia’s marine bioregions is currently underway through the Department of Environment and Heritage (www.deh.gov.au).

**Statement of the contribution of others in Chapter 5.**

Ron Avery developed the original environmental classification with the NSW Marine Parks Scientific Committee and provided much of the research to justify the environmental categories chosen and identify available sources of data. Ron also developed and implemented the methods to digitise near shore environments from aerial photographs.
Figure 4.1 Marine protected areas in NSW before 2006 (map provided by Rodney James, Department of Environment and Conservation).
5 Methods to identify MPAs in NSW

This chapter describes the methods used to assess broad scale patterns in marine biodiversity and identify potential sites for MPAs in NSW. More detailed accounts can be read in Breen et al. (2003, 2004, 2005, 2006), Creese and Breen (2003) and Appendix 1. The approach used in NSW differs substantially from the process used for the Great Barrier Reef Marine Park Representative Areas Program in the amount of detailed biological information available and the level of public and scientific consultation.

The mapping methods used in the following assessments are derived from those adopted by Avery (2001) for a marine protected area assessment of the Tweed-Moreton bioregion. Mapping was based largely on the modification of existing data into an appropriate GIS format. The major exceptions were maps of near shore reefs and rocky shores digitised by Ron Avery from high resolution aerial photography. A major constraint for the assessments was the scarcity of biological data at the community and species level for large areas, and the absence of detailed maps of subtidal seabed beyond the near shore zone. The assessments focused on mapping broad scale variation as coarse scale ‘ecosystem’ and ‘habitat’ surrogates supplemented with available species data and derived measures of condition and vulnerability.

Options for marine protected areas in each bioregion were identified according to three major criteria: comprehensiveness, representativeness and adequacy. These broad criteria were applied in the assessment through the many, more detailed sub-criteria described in Chapter 2. While data were available to assess most measures for comprehensiveness as defined in Figure 2.2, much less information was available to assess representativeness at the community and species level (Figure 2.3).

To assess adequacy, quantitative measures of the potential level of human impact were calculated as general indicators of the likely condition and vulnerability of different areas (Figure 2.4). These measures were based largely on patterns of adjacent land use, as little information was available to consistently measure disturbance of marine environments at broad spatial scales.

Ultimately, the location of MPAs will be strongly influenced by meeting criteria for comprehensiveness and representativeness. However, the ability of reserves to adequately protect biodiversity and ecosystem processes will also be influenced by reserve designs that consider biological characteristics such as home range, migration, habitat complexity, disturbance and connectivity among species, habitats and processes. Criteria to address these factors were conceptually grouped under criteria for reserve design and adequacy in Figure 2.4 and include consideration of the size, shape, configuration and replication of reserves in a MPA network. Although these factors were not assessed analytically, guidelines summarised from the scientific literature were compiled by Ron Avery and are presented in Appendix 1. The decision
support tools used in Chapters 6 and 10 provide ways to explore and implement these principles.

Criteria were first assessed as a function of measures for individual criteria and then combined in irreplaceability and multiple criteria analyses. Each information source, the criteria addressed and the derived measures are described in Chapters 6-8 with an assessment of the degree to which different planning units met those criteria.

The methods reflect the paucity of marine biological data for large areas and a significant by-product of this work is the identification of gaps in our knowledge of marine biodiversity in NSW. However, by basing the assessment on a broad scale environmental classification, the project avoids many of the biases inherent in examining only those areas where detailed research information is available. The ecological classification presented here lays the foundations for future research and is general enough to incorporate new information as it becomes available. The information used to identify MPAs in following three chapters can be summarised as:

- MPA goals and identification and selection criteria (Chapter 2)
- an environmental classification of marine ecosystems, habitats and communities
- available broad scale surveys of marine communities and species
- derived measures from related conservation assessments, and
- maps created of existing marine protected areas.

The methods used to help identify candidate MPA locations from this information include:

- summary statistics displayed in graphs and tables
- Geographical Information System (GIS) maps and spatial analyses
- irreplaceability analyses in C-Plan (NPWS 2001) and Marxan (Ball and Possingham 2000)
- multiple criteria analyses, and
- reviews of literature.

### 5.1 A marine environmental classification for NSW

The broad scale assessments are based primarily on an environmental classification developed in conjunction with Ron Avery and the NSW Marine Parks Research Committee. The surrogate biodiversity measures in this classification were used to represent progressively finer scales of biological variation in marine environments. Levels in the hierarchy were:

- IMCRA bioregions (IMCRA 1998)
- ‘ecosystem’ units - estuary types and cross-shelf depth zones (Figure 2.2 and Figure 6.2)
- ‘habitat’ units - seagrass, saltmarsh, mangrove, reef, rocky shore, beach, island and sediment (Figure 2.2 and Figure 6.8)
- finer scale ‘community’ units from other physical predictors, dominant biota and species assemblages (Figure 2.3)
- estimated distributions and abundances of species and populations (Figure 2.3).
Categories in the classification are described in the following section with a brief justification based on a review of the marine research literature researched mainly by Ron Avery. More detailed descriptions are available in Breen et al. (2003, 2004, 2005, 2006).

5.1.1 Estuary ecosystem classes
Coastal water bodies from the NSW Waterways GIS coast coverage were classified on the basis of coastal morphology, entrance type and tidal exchange according to Roy et al. (2001) who associates these differences with characteristic ecosystem processes and related assemblages of organisms. The estuary classes are:

I. Ocean embayments. These semi-enclosed bays are transitional zones between estuaries and the ocean and include communities of both environments. They generally have low turbidity, ocean tidal ranges and salinities, and include sandy areas with seagrass beds in protected locations (e.g. Botany Bay, Jervis Bay, Batemans Bay and Twofold Bay).

II. Tide dominated, drowned river valleys. These are tidal, generally deep, narrow estuaries with rocky sides, and sometimes with large, submerged, sand deltas extending up the estuary (e.g. Port Stephens, Sydney Harbour, Hawkesbury River and the Clyde River estuary).

III. Wave dominated, barrier estuaries. Young barrier estuaries in the early stages of infilling have large shallow lagoons with dense seagrass beds away from the main tidal channels (e.g. Wallis Lake, Lake Macquarie, St Georges Basin and Tuross Lake). Mature estuaries in the late stages of infilling form a riverine estuary with extensive flood plains and coastal wetlands. They often have narrow, elongated entrance channels and broad barrier sand flats (e.g. Tweed, Richmond, Clarence, Macleay, Hastings, Manning and Shoalhaven Rivers).

IV. Intermittent lagoons and creeks. These water bodies are intermittently open to the ocean, are usually associated with small catchments and small fluvial inputs, and are often non-tidal and brackish. Mangroves are generally absent, with sea rush (Juncus kraussi) often dominant. Benthic species diversity is often low, but there are sometimes extreme variations in abundance (e.g. Durras Lake, Narrabeen Lakes, and Smiths Lake).

V. Brackish barrier lakes. These bodies of fresh to brackish water have only a tenuous connection to the sea and are dominated by freshwater species. They are relatively rare in NSW (e.g. Myall Lakes).

5.1.2 Ocean ecosystem classes
Oceanic ecosystem types were derived from depth contours digitised by NSW Waterways from Australian Hydrographic Office nautical charts. The contours were used to divide the shelf into four broad depth zones between 0-20 m, 20-60 m, 60-200 m and waters deeper than 200 m. These zones aim to account for biotic and abiotic variation across the shelf in algae (Womersley 1984), sponges (Roberts and Davis 1996), benthic fauna (Coleman et al. 1997, Gray et al. 1997), fish assemblages (Andrew et al. 1997), light, wave action, sediments, currents,
Methods to identify Marine Protected Areas in NSW


5.1.3 Seagrass, mangrove and saltmarsh habitats

The distributions of seagrass, mangrove and saltmarsh habitats were estimated from a GIS coverage digitised by the National Parks and Wildlife Service from paper maps produced by West et al. (1985). Mangrove and salt marsh communities contribute to estuaries through nutrient cycling, trapping of sediments and detritus and providing habitat for characteristic and highly diverse assemblages of fish, birds and invertebrates (Hutchings and Recher 1982, Saenger 1999). Seagrass beds are widely recognised for their role in providing habitat for diverse assemblages of flora and fauna (Bell and Pollard 1989, Howard and Edgar 1999, Hannan and Williams 1998). The maps are now being updated through digital GIS classifications of orthorectified high resolution aerial photographs (1:10,000-1:25,000, R. Williams and G. West pers. comm. Fisheries, NSW Department of Primary Industry).

5.1.4 Subtidal reef habitats

Areas of shallow near shore reef systems and intervening sediment patches were estimated from GIS coverages mapped to a depth of 10-20 m by Ron Avery (NSW Department of Environment and Conservation, DEC) using 1:10,000 – 1:25,000 scale aerial photographs provided by the NSW Department of Infrastructure, Planning and Natural Resources (DIPNR). Shallow areas of reef and shoal further offshore were digitised from Australian Hydrographic Office nautical charts.

Subtidal rocky reef areas in NSW provide habitat for distinctive assemblages of invertebrates, algae and fishes (Underwood et al. 1991, Andrew 1999). However, the use of aerial photographs to map subtidal habitats is limited to near shore areas and hydrographic charts focus only on those reefs and shoals that approach the sea surface and pose a hazard for shipping. It is recommended that a more comprehensive assessment of existing seabed data is made and that, where required, additional seabed surveys are carried out to accurately characterise these environments.

5.1.5 Island habitats

Areas of islands and emergent rocks were estimated from a GIS coverage generated for the Australian Maritime Boundary Information System (AMBIS) held by Geoscience Australia (Commonwealth of Australia 2001). Islands, emergent rocks and surrounding waters provide unique and important habitats for seabirds, marine mammals, fish, invertebrates and other species. Fronts, wakes and other oceanographic features that extend beyond rocks and islands (Cresswell et al. 1983) are important for the feeding ecology of many species and the transport

3 now within the NSW Department of Environment and Conservation (DEC)

5.1.6 **Subtidal sediment habitats**

Areas of near shore subtidal sediments were estimated from the GIS coverage described above for subtidal reef. Benthic fauna are known to vary significantly with depth and grain size (Poore et al. 1985 in Ward and Blaber 1994, Coleman et al. 1997), but for many areas there is currently little compiled information on the distribution of sediments that can be easily accessed. While cross shelf variation in sediment distribution is at least partly represented by ocean depth zones, further research and collation of existing data is required.

5.1.7 **Intertidal beach habitats**

Topographic maps (1:25,000) and the calculated area between the high and low water lines in the Digital Cadastre database from the Land and Property Information Division (LPI) of the NSW Department of Lands were used to produce a GIS shape file of intertidal areas and estimate areas of intertidal beach habitat. Justification for the classification of beaches in NSW is provided by Hacking (1997, 1998) based on relationships described in McLachlan and Hesp (1984), McLachlan (1985, 1990), Brown and McLachlan (1990) and McLachlan et al. (1996). 

5.1.8 **Intertidal rocky shore habitats**

The intertidal GIS coverage described in section 5.1.7 for beach habitats was also used to estimate the area of intertidal rocky shore habitats. Field surveys by Otway (1999) and Otway and Morrison (in prep.) were used to score sections of rocky shore for the presence of five ‘community’ level substrata (platform, boulder, cobble, pool, and crevice) that were correlated with the number of species present for a given shore. Other previous conservation assessments (Quint 1982, Short 1995) also provided information on rocky shores.

5.2 **Data for individual species**


Other, data sources for species included analyses of commercial fish catch data (Pease 1999), and sightings databases kept by the NSW Department of Primary Industry, the NSW Department of Environment and Conservation and the Commonwealth Department of Environment and Heritage. Data from museum collections were not available at the time of the assessments, although they had been previously reviewed for these projects (Avery 2001).
5.3 **Condition, vulnerability and previous assessments**

There was little direct information available on condition, threat or vulnerability for marine environments across whole bioregions. However, data sets indicative of condition, potential threats and vulnerability were available for adjoining terrestrial areas. These included GIS maps of national parks and nature reserves, state forest, wetlands, wilderness, land capability, built-up areas, acid sulphate soils, and the Australian river and catchment condition database (Stein *et al.* 2000). Indices of the percentage area of these features along shorelines and in catchments were calculated for estuaries and sections of coast.

The results of previous conservation assessments for wetlands (ANCA 1996), estuaries (Bell and Edwards 1980, Digby *et al.* 1998, Frances 2000, Healthy Rivers Commission 1998, 1999, 2002ab), and rock platforms (Short 1995, Otway 1999) were also summarised and related to MPA identification and selection criteria along with descriptive information from coastal management plans (NPWS 1995ab, 1998ab).

5.4 **Systematic methods to identify MPA options in NSW**

A systematic approach was used to document the conservation values of alternatives, and interpret the many criteria and sources of information used to assess options for MPAs. The methods used included summary statistics, Geographical Information System (GIS) maps and spatial analyses, irreplaceability analyses and reviews of literature and existing conservation plans.

5.4.1 **Planning units**

Two types of spatial planning units were used to summarise information: fine scale (1 to 4 km²) hexagonal plan units (Figure 5.1, Figure 6.37, Figure 7.24 and Figure 8.33) and relatively large, broad scale units representing whole estuaries and sections of coast and shelf (Figure 6.2, Figure 7.4 and Figure 8.6). The small planning units were useful for summarising local patterns, and for identifying small scale planning options. The large planning units were more useful for summarising broad scale regional patterns, analysing patchy data and identifying MPA options at wider scales.

5.4.2 **Graphical summaries and qualitative scores**

Broad scale planning units were used to summarise regional patterns in the number and size of different ecosystems, habitats, communities and populations. These and other measures were graphed, mapped and summarised in tables with the results of related conservation assessments. Each information source, the criteria addressed, the derived assessment measures used and values for broad scale planning units are presented in Chapters 6-8.
Fine scale planning units (Port Stephens area only) derived from 1 square kilometre hexagons and major marine and terrestrial features. The units are linked to relational databases holding surrogate biodiversity measures for ecosystems, habitats, communities and species, values indicating environmental condition, threat, and vulnerability and to measures of current patterns of human use and management. The units provide an interactive graphical interface to spatially query databases and provide input values to, and display output from reserve selection modelling tools like CPLAN (NPWS 2001).

Figure 5.1. Fine scale planning units derived from 1 km² hexagons and coastal features.
5.4.3 Irreplaceability

A number of computer-assisted techniques take into account the ‘complementarity’ (Pressey et al. 1994) of different areas in jointly achieving targets for a range of conservation features. These targets are usually defined in terms of a desired area or proportion of a habitat, or representation of a given number of occurrences of a species.

The complementary value of a site to a reserve network is not only related to how many features (e.g. species or habitats) it includes, but also, to how that site complements the range of features already represented in protected areas. While an area may include many species and habitats, it may not be able to add anything to an existing network if conservation goals have been met for those features. However, a site with one feature not found elsewhere may be virtually irreplaceable and be an essential requirement if a system of reserves is to meet a particular conservation goal.

Including new locations in a reserve continually alters the potential value of remaining areas in meeting overall goals, and a site’s value is affected by the order in which new areas are included. These changing values are difficult to quantify within a static measure, but a statistical estimate can be made of each site’s ‘irreplaceability’. This measure represents “the likelihood that an area will be required as part of a conservation system that achieves the set of targets” or “the extent to which the options for achieving the set of targets are reduced if the area is unavailable for conservation” (Pressey et al. 1994).

Statistical estimators for irreplaceability can be computed relatively quickly (Ferrier et al. 2000). Other goal seeking algorithms such as integer linear programming (Cocks and Baird 1989; Underhill 1994), iterative heuristic algorithms (Margules et al. 1988, Nicholls and Margules 1993, Pressey et al. 1997) or simulated annealing (Possingham et al. 1999a) may have longer computing times, particularly where there are many sites and criteria.

The NSW National Parks (DEC) reserve selection software ‘C-Plan’ (NPWS 2001) was used in the assessments to compute irreplaceability for the following data sets:

- ecosystem types (estuary type and ocean depth zone)
- habitat types (seagrass, mangrove, saltmarsh, rocky intertidal shore, beach, subtidal reef and islands)
- juvenile estuarine fish and invertebrate survey data (Fisheries, NSW DPI)
- commercial fish catch data for estuaries and ports of landing (Fisheries, NSW DPI)
- bird sightings data (NSW National Parks (DEC))
- threatened species data (NSW National Parks (DEC))
- surveys of species presence on rock platforms (Griffiths 1982).

Hypothetical conservation targets of 20% of the total area of each ecosystem and habitat in the bioregion were used to calculate and map irreplaceability indices. C-Plan requires a
conservation target to be set as a computational necessity. For the species data sets, a hypothetical target to represent each species at least once in an MPA system was used to calculate relative irreplaceability values. For area targets, 20% goals were set as arbitrary targets to allow the calculation of relative irreplaceability values. If targets are set at 100%, all planning units are irreplaceable; if set at 0% no locations are required. Setting a target between these extremes allowed the relative merits of alternative MPA systems to be assessed for a range of scenarios. However for presentation in print, only a few representative scenarios will be shown.

Two different irreplaceability measures, site irreplaceability and summed irreplaceability, were calculated for the broad scale and fine scale planning units. Site irreplaceability is a measure of the overall likelihood that an area will be required as part of a conservation system to achieve a set of conservation targets. Irreplaceability values can range from totally irreplaceable (1.0) to zero irreplaceability (0.0).

In explaining this concept, Pressey et al. (1994) state: “If an area is totally irreplaceable, then no matter how a system of conservation areas is designed for a region, it will have to include that area. Put the other way, if that area loses its conservation values, one or more of the conservation targets for the study area will become unreachable. Areas with progressively lower irreplaceabilities have progressively more replacements in the region, less likelihood of being required as part of a system of conservation areas, and less impact on the achievement of targets if destroyed or unavailable for conservation. Areas with zero irreplaceability contain only features that have already had their conservation targets met in existing protected areas.”

While a site irreplaceability of 1.0 may indicate that a planning unit is irreplaceable for one or perhaps several conservation targets, summed irreplaceability is also related to how many conservation targets a planning unit is likely to contribute to. It is derived by summing the individual site irreplaceabilities estimated separately for each conservation target and can therefore range from 0 to greater than 1.

For most data sets, site irreplaceability values indicated that many options were available for meeting feature targets and only a few planning units were totally irreplaceable for some targets. In general, we present results for summed irreplaceability of the broad scale plan units as these provided the most easily summarised, general interpretations of the data. However, for resolving small-scale pattern and building detailed reserve networks, the fine scale units and a variety of diagnostic measures and tools can be used.

Links between C-Plan and ArcView GIS allow operators to quickly map the results of analyses and include or exclude potential sites from MPA networks while assessing the consequences of alternative decisions. These rapid display and analysis capabilities of C-Plan make it a useful tool for exploring potential scenarios during decision-making.
5.4.4 Multiple Criteria Analyses

Criteria, data and options identified in the assessments were also used in multiple criteria analyses (Criterium Decision Plus, InfoHarvest 2000) to assess options for MPAs as a function of the combined scores for many criteria and priorities. These methods have been widely applied elsewhere in management, environmental impact assessment (Edwards 1977, www.expertchoice.com), fisheries (Mardle and Pascoe 1999) and in the selection and management of reserve networks (Fernandes 1996, Rothley 1997, Villa et al. 2002). The techniques allow for the weighting of criteria, calculation of trade offs, representation of uncertainty, sensitivity analyses of the relative influence of criteria, and the ability to combine and assess alternative models, data and sources of opinion.

In the Manning Shelf assessment, the Simple Multi Attribute Rating Technique (SMART) was used to assess goals and criteria for each individual estuary and section of coast and ocean. In the Hawkesbury, Batemans and Twofold Shelf assessments, the technique was used to compare nine different options for large marine parks.

Statement of the contribution of others in Chapter 6.

Ron Avery was a partner in the Manning Shelf bioregional assessment and in developing a systematic approach to researching and assessing data, mapping the environmental classification and mapping a range of other data sets, including existing marine protected areas and other management sites. Ron also mapped near shore reefs for all of NSW and rocky shores for the Manning Shelf bioregion from aerial photograph and assisted in identifying options for MPAs.