Identification of intertidal marine reserves – using habitat types to identify areas of high conservation value

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DECLARATION

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Banks SA, Skilleter GA. (in press) Implementing marine reserve networks: a comparison of approaches in New South Wales (Australia) and New Zealand. Marine Policy – Greg Skilleter was responsible for reviewing and editing draft versions of the manuscript.

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ABSTRACT

The goal of biodiversity conservation has been described as the conservation of diversity at three levels: ecosystem, species and genetic diversity. Developing a representative system of marine protected areas is considered an effective way to achieve this goal in the marine environment. The growing concern associated with threats to the marine environment has resulted in an increased demand for marine reserves (i.e. no-take areas) that conserve representative and adequate examples of biodiversity. Often, the decisions about where to locate reserves must be made in the absence of detailed information on the patterns of distribution of the biota. Alternative approaches are required that include defining habitats as surrogates for biodiversity.

The development of biodiversity surrogates at fine-scales (i.e. habitats) will have an increasingly important role in the identification of sites that will contribute to a representative system of marine protected areas. This is because it will increase the likelihood that the system will adequately achieve biodiversity objectives by ensuring protection of a greater range of habitats and species. Surrogate measures of biodiversity enable decisions about where to locate marine reserves to be made more reliably in the absence of detailed data on the distribution of species. There is concern, however, about the reliability of surrogate measures to represent biotic diversity and the use of such measures in the design of marine reserve systems. Currently, surrogate measures are most often based on broad-scale (100s to 1000s of kilometres) bioregional frameworks that define general categories (sandy beach, rocky shore) for intertidal systems. These broad-scale categories are inadequate when making decisions about conservation priorities at the local level (10s to 100s of metres).

This study provides an explanation of an intertidal shoreline habitat surrogate (i.e. shoreline types) used to describe 24,216 kilometres of Queensland’s coastline. The protective status of shoreline types was evaluated to assist with designing a representative system of intertidal marine protected areas. The shoreline types derived using physical properties of the shoreline were used as a surrogate for intertidal biodiversity to assist with the identification of sites for inclusion in a candidate system of intertidal marine reserves for 17,463 kilometres of the mainland coast of Queensland, Australia. This represents the first systematic approach, on essentially one-dimensional data, using fine-scale (10s to 100s of metres) intertidal habitats to identify a system of marine reserves for such a large length of coast. A range of solutions would provide for the protection of a representative example of shoreline types in Queensland.

Shoreline types were used as a surrogate for intertidal biodiversity (i.e. habitats, microhabitats) to assist with the identification of sites to be included in a representative system of marine reserves in south east Queensland. The use of local-scale shoreline types increased the likelihood that sites identified for conservation achieved representation goals for the mosaic of habitats and microhabitats, and therefore the associated biodiversity present on rocky shores, than that provided by the existing marine reserve protection in south east Queensland. These results indicate that using broad-scale surrogate measures (rocky shore, sandy beach) for biodiversity (habitats, microhabitats and species) are likely to result in poor representation of fine-scale habitats and microhabitats, and therefore intertidal assemblages in marine reserves. When additional fine-scale data were added to reserve selection the summed irreplaceability of 24% (for spatial extent of habitats), and 29% (for presence/absence of microhabitats) of rocky shore sites increased above zero, where a value close to one means a site is necessary, for inclusion in a reserve system, to meet conservation targets. The use of finer-scale physical data to support marine reserve design is more likely to result in the selection of reserves that achieve representation at habitat and microhabitat levels, increasing the likelihood that conservation goals will be achieved. The design and planning of marine and terrestrial protected areas systems should not be undertaken independently of each other because it is likely to lead to inadequate representation of intertidal habitats in either system. The development of reserve systems specially designed to protect intertidal habitats should be integrated into the design of terrestrial and marine protected area systems.

Marine reserve networks are a necessary and effective tool for conserving marine biodiversity. They also have an important role in the governance of oceans and the sustainable management of marine resources. The translation of marine reserve network theory into practice is a challenge for conservation practitioners. Barriers to implementing marine reserves include varying levels of political will and agency support and leadership, poorly coordinated marine conservation policy, inconsistencies with the use of legislation, polarised views and opposition from some stakeholders, and difficulties with defining and mapping conservation features. The future success of marine reserve network implementation will become increasingly dependent on: increasing political commitment and agency leadership to remove conflicts within and between government agencies involved in site identification and selection; greater involvement and collaboration with stakeholders; and the provision of resources to define and map conservation features. Key elements of translating marine reserve theory into implementation of a network of marine reserves are discussed based on approaches used successfully in New Zealand and New South Wales (Australia).

Keywords

intertidal habitats, conservation planning, marine reserves, microhabitats, reserve design, rocky shore, siting algorithm

Australian and New Zealand Standard Research Classifications (ANZSRC)

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LIST OF ABBREVIATIONS

BLM	Boundary Length Modifier
IMCRA	Interim Marine and Coastal Regionalisation for Australia (IMCRA Technical Group 1998)
Marine reserve	No-take areas; no-take zones, sanctuary zones, national park zones in multiple-use marine protected areas
MPAs	Marine Protected Areas – referring to areas managed for multiple-uses
NRSMPA	National Representative System of Marine Protected Areas
System	Collection of marine reserves or marine protected areas. Also referred to as a network of marine reserves or marine protected areas

CHAPTER 1 — GENERAL INTRODUCTION

1.1 THREATS TO MARINE AND COASTAL BIODIVERSITY

Internationally there is broad recognition that the marine and coastal environment is under increasing threat and stress (see for example Boersma & Parish 1999; Worm et al. 2006; Young et al. 2007). There are some threats that are considered to have global implications such as climate change and sea-level rise, which have particular implications for the coastal zone (Kelleher 1999; Bradbury et al. 2001; McGoodwin 2007). In an analysis of threats to the marine environment, Gray (1997) concluded that there were few threats to the open ocean, and that threats are generally concentrated in coastal areas. This may however reflect the state of knowledge of the marine environment, which is limited for open ocean ecosystems (Worm et al. 2006). Many human activities are compromising marine ecosystems, leading to large-scale alteration or degradation of the coastal environment (Attwood et al. 1997a, 1997b; Gilman 1997; Micheli 1999; Hooper et al. 2005). Threats to coastal areas have been broadly grouped as:

- (i) climate change;
- (ii) coastal development;
- (iii) pollution;
- (iv) over-exploitation of coastal resources; and
- (v) species introductions (Boersma & Parish 1999).

1.1.1 Effects of climate change

The impacts of climate change on social, economic and environmental infrastructure around the world is becoming an increasingly important issue with growing concern about its consequences (Intergovernmental Panel on Climate Change 2001, 2007; Opdam & Wascher 2004; Few et al. 2007; Hennessy et al. 2007). Scientific evidence is predicting continued rise in global average near-surface temperatures, which is expected to lead to rises in sea level, increased heavy rainfall, retreat of glaciers, and alterations in patterns of river runoff and storm events (Houghton et al. 2001; McCarty 2001; Intergovernmental Panel on Climate Change 2002; Arnell 2004; Woodworth et al. 2005; Commonwealth Scientific and Industrial Research Organisation 2006; Few et al. 2007). There are concerns that the consequences of such changes will be devastating to species dependent on marine and coastal habitats (Intergovernmental Panel on Climate Change 2001, 2007).

In the coastal zone there is expected to be a greater frequency of storm surges, changes in weather patterns, ocean currents and ocean temperatures (Hennessy et al. 2007). As a consequence, these changes are expected to lead to increased coastal erosion, changes in sediment supply and a loss of biodiversity (Intergovernmental Panel on Climate Change 2002; Hennessy et al. 2007). There are predicted to be major problems for the survival of species including accelerating rates of extinction and changes to species distribution patterns (Brereton et al. 1995; McCarty 2001; Hannah et al. 2002). For example, it is expected that there will be a southward shift (i.e. poleward) in species distributions in Australia, a predicted loss of wetland communities (Commonwealth Scientific and Industrial Research Organisation 2006; Hennessy et al. 2007), and also changes to life cycles of intertidal species, such as the timing of flowering in mangroves (Howden et al. 2003).

1.1.2 Habitat loss and degradation

Many impacts from human alteration to coastal habitats result in obvious changes to their physical structure including loss, damage and the local extinction of habitats and species. Coastal development, including the construction of marinas, harbours and other facilities, result in the destruction of intertidal habitats and may also include the installation of artificial features such as rock walls (Harrison & Parkes 1983; McIntyre 1992; Thompson et al. 2002). These artificial features can have profound effects on adjacent habitats due to changes in current flow and exposure to waves, resulting in further degradation and changes to habitats, and loss of suitable habitat for those species that previously occupied the area (UNEP 1996). With a predicted increased frequency in storm surges, associated with climate change, it is likely that further modifications of coastal habitats to protect human settlements will be required (Hennessy et al. 2007).

Scientists, conservation practitioners and the community are also concerned that threats to habitats and ecosystems in the coastal zone are likely to increase due to the burgeoning world population, which has quadrupled in the past decade, with a trend towards greater occupation of coastal areas (Gray 1997; Thompson et al. 2002). For example, over 80% of Australia’s population lives in the coastal zone (Harvey & Caton 2003). The growing occupation of the coast is likely to lead to a greater need to build coastal defenses to protect human settlements from storm surges and flooding (Hennessy et al. 2007). Despite some knowledge of a range of threats to coastal habitats and species, there is often little specific information about their overall consequences to genetic, species or ecosystem diversity. There is evidence to suggest that there are chronic long-term impacts that result from changes in biological processes related to over-exploitation, pollution and habitat degradation in coastal habitats (Siegfried et al. 1985; Duran & Castilla 1987; Bustamante & Castilla 1990; Castilla 1999).

Other activities that may appear more benign in their effects on coastal systems include, for example, changes in shipping technology and infrastructure, which has resulted in increasing accessibility of reefs by tourism operators and hence an increase in the number of people that can be transported daily to individual locations (Inglis et al. 1999). Increasing use of an area for tourism or recreation may also result in alteration or pollution of coastal habitats affecting species' distributions (Gray 1997; Jennings 2004). Often physical infrastructure to support recreation and tourism activities is required and therefore there would be degradation and alteration to coastal habitats to construct such facilities. These types of changes are likely to increase pressure in areas that are popular for tourist and recreation activities.

Alteration of catchments adjacent to coastal areas has also led to the disruption of marine processes. While physical alteration of coastal habitats may be obvious the impact of human activities in areas distant to the coastal system such as deforestation and mining have been described as threatening habitats (Johannes 1975; Gray 1997).

1.1.3 Pollution of the marine environment

The trend of increasing human occupation of coastal areas has led to increased pollution of nearshore and coastal habitats (Gray 1997; National Academy of Sciences 2001; Thompson et al. 2002). This is because there is a belief that there is a natural assimilation capacity by marine and coastal waters to process sewage, urban and industrial effluent. Many forms of pollution have been reported to alter species compositions both in the water column and benthic communities (Fairweather 1990a; Otway 1995; Gray 1997; Thompson et al. 2002). Intertidal habitats represent an interface between the land and sea where many forms of pollution can have chronic effects over vast stretches of coastline (Thompson et al. 2002).

Oil spills from shipping accidents or offshore oil production have also affected coastal areas. The impacts of oil spills have been found to be substantial, although the long-term consequences on coastal habitats and species remain poorly known (Lubchenco et al. 1995). Oil spills have been described as causing major localised effects on biota (Hockey & Branch 1994). There has been a history of major oil spills that have impacted coastal habitats. For example, the *Amoco Cadiz* resulted in heavy impacts along 440 kilometres of coast and affected a further 260 kilometres of coastline less intensively (Teal & Howarth 1984). It is not only the direct impact of the oil spill but also the impact of the clean-up that has affected habitats and species populations (Clark 1982). One of the most well known oil spills followed the grounding of the *Exxon Valdez* resulting in the spilling of 11 million barrels of crude oil. After 10 years of research on the effects of this incident there is evidence that the oil continues to persist in intertidal and shallow subtidal habitats where many species have not recovered (NOAA 2001).

In response to the threat from oil spills many regions throughout the world have prepared oil spill contingency plans that detail responses to oil spill events in order to minimise damage to sensitive coastal habitats. An important factor in the successful management of a response to an oil spill is having an understanding of the distribution of vulnerable ecosystems and habitats. This would include mapping intertidal shorelines in order to enable decisions to be made about the appropriate response to minimise the potential effects of oil spills. Mapping is required to enable decision-making at a local-level (10s to 100s of metres) over large geographic areas (100s to 1000s of kilometres). Local-scale mapping enables intertidal habitats sensitive to oil spills to be identified for the deployment of a management response.

1.1.4 Over-exploitation of marine resources

The provision of natural resources for a range of exploitative and non-exploitative industries, recreation and subsistence living and the burgeoning world population in coastal areas has led to increasing exploitation of marine resources by humans (Fairweather 1990b; Pullen 1997). The focus of exploitative activities throughout the world is on a wide range of marine resources including fish, invertebrates, seaweed and minerals resulting in a variety of effects, during their collection, to marine and coastal habitats and species (Castilla & Duran 1985; Keough et al. 1993). This use has led to the over-exploitation of many species resulting from increased collection and overfishing of organisms from the intertidal shoreline and adjacent shallow subtidal habitats (Duran & Castilla 1987; Lubchenco et al. 1995).

1.1.4.1 Overfishing

Overfishing relates to the over-harvesting of fish stocks and the direct impacts of different fishing techniques on habitats. For example, the modification of habitat has been described as resulting in the decline in both commercial and recreational fish catches in estuaries (Hockey & Branch 1994). Trawling has been found to impact on habitat resulting in a major effect on target and non-target species (Gray 1997; Hiddink et al. 2006). An important issue concerning the destruction of subtidal habitat is that it largely remains unknown what the long-term consequences of such modification are on benthic species and their population dynamics. There has been recognition that fisheries management practices have largely failed to address the declining fish stocks and destruction of habitat of commercially important species.

Overfishing is a threat that has received a large amount of attention both in terms of developing management tools, and in the research and modelling of fish population dynamics (McIntyre 1992; Hanna 1999; Frank & Brickman 2001; Worm et al. 2006). In some instances this research and management has come too late with over two-thirds of the worlds fish resources fully exploited, over-exploited or depleted (Boesch 1999). Most research has focused on commercially important species and the consequences of over-harvesting on their population dynamics.

There have been dramatic changes in the composition of fish stocks and other marine fauna as a consequence of fishing (Gray 1997). Changes related to the commercial take of species include the decline in target species (e.g. herring and Artic cod) and alterations to the abundance of less commercially valuable species such as sharks (Sherman & Alexander 1990). However the status of most shark species is uncertain and in some locations there are indications of rapid declines in coastal and oceanic shark populations (Baum et al. 2003). These changes also extend to other species such as seabirds and marine mammals that have shown dramatic declines in some species as a result of changes in prey in areas where harvesting occurs (Monaghan 1992; Hamre 1994; Pillans et al. 2007). There have been regulatory measures put in place to manage bycatch resulting from fishing practices, however compliance is very difficult to monitor.

There is less information and data to support the management of recreational angling. The cumulative impacts of such fishing has been recognised as a potential threat to fish stocks however there are limited data that indicate the significance of the impact of recreational fishing on fish species composition. It has been found that species (e.g. blue groper, black fish, red morwong, lobster) targeted by recreational anglers protected by a marine reserve were significantly greater in density in the protected area than adjacent non-protected areas (Gladstone 2001; Shears et al. 2006). In addition there was evidence that the assemblages of fish species differed between protected areas and non-protected areas and that non-target species in the non-protected areas increased in density. The results of the study firstly indicate that recreational anglers impact on target species and secondly that the closure of areas to recreational anglers can result in significant benefits to fish species richness and density of target species.

1.1.4.2 Intertidal harvesting

There is evidence of the dramatic direct impacts of human exploitation on intertidal shoreline community structure (Castilla & Duran 1985; Duran & Castilla 1987; Castilla 1999, 2000; Moreno 2001; Zharikov & Skilleter 2003, 2004; Skilleter et al. 2005, 2006). There is also evidence of seasonal pressure on intertidal organisms resulting from seasonal increases in local human population centres (Duran & Castilla 1987). In addition to over-exploitation generally, it has been documented that there is often a non-discriminatory approach to exploitation and the collection of both juveniles and adult intertidal species. The result of such non-discriminatory over-harvesting has led to changes to biological assemblages present along a coastline.

Research on intertidal systems tends to be short-term with an increasing focus on investigating the differences between areas open to harvesting and areas closed to harvesting (see for example Castilla & Duran 1985; Moreno et al. 1986; Zharikov & Skilleter 2003, 2004; Skilleter et al. 2005, 2006; Martins et al. 2008). The over-exploitation of some intertidal species, for example the Brown Mussel (*Perna perna*) has led to the transfer of effort onto other intertidal organisms, which has also resulted in changes to intertidal community structure (Siegfried et al. 1985). In this particular situation other management tools regulated the capture-size of the organisms collected, however these regulations were found to be largely ignored. Examples such as this highlight the consequences of over-exploitation and limited management, and the need for management or the establishment of intertidal marine reserves where harvesting is prohibited. The use of marine reserves provides a simple and enforceable management approach compared to regulation of capture-size or limits on numbers of organisms. With expected changes in the population demographics in coastal areas the need for simple approaches to management will provide a better means to manage increased pressure on the species and habitats that are the focus of collection activities.

1.2 MANAGING THE COAST

There are many complexities with managing the coastal zone, particularly since the coastal environment is increasingly becoming the focus of human occupation (Holmes & Saenger 1995; Gray 1997). The complexity of coastal management is illustrated by the various levels of government and numerous legislative instruments that are used to influence development, and biodiversity and resource conservation activities in the coastal environment. Internationally and nationally the trend to integrate a range of management tools in an attempt to increase their overall effectiveness in coastal management has evolved. Strategies for the conservation and protection of the coastal zone continue to evolve as scientists, conservation practitioners and the community place increasing importance on the conservation of ecosystems, habitats, communities and species.

1.2.1 Coastal zone management – global trends

Historically the approach to coastal management has been to develop policy and legislative tools to manage or reduce threats (Jennings & Reganold 1991; Pullen 1997; Young & Gunningham 1997;

Young 1998; Kelleher 1999). A key objective of coastal management is the long-term provision of ecosystem services through conservation and management of biodiversity and its sustainable use (Parnell et al. 2006). There has been a sectoral approach to management of problems, which is described as inefficient and contributing to the creation of new environmental problems (Fast et al. 2001). These approaches have largely failed to reduce continuing degradation of ecosystems and habitats. In response, a new paradigm of integrated coastal management has developed to provide a broad framework to manage use, and its impacts on coastal resources and conflicts in the coastal zone (Pullen 1997).

Coastal management has evolved from a simplistic model that had difficulties with the meaning of basic and commonly used terms and phrases to a clearer field of integrated coastal management (Dutton & Hotta 1995). The integrated coastal management framework aims to integrate the management of activities, developments, and resources, the environment and biodiversity in the coastal zone, and to establish a system of regulatory land use measures for the protection of areas of ecological, landscape and amenity value (Pullen 1997). While in theory the concept of integrated coastal management is sound, putting such approaches into practice remains a challenge, and in many instances only components of the integrated coastal management framework have been implemented.

The complexity of the marine environment is basically matched by the complexity of coastal management where few examples are available of a successful integrated approach to managing the coastal zone (Dutton & Hotta 1995). The scale and inter-connectivity of marine ecosystems and habitats further complicate the management and protection of the marine environment. The ability for pollutants, nutrients, sediment, exotic species and fish to move through the water column present special management problems, particularly when determining boundaries for protected areas (Kelleher 1999), or opportunities to mitigate threats.

Progress in coastal management has been through controls on land-based development with less consideration towards biodiversity conservation of the land/sea interface (i.e. intertidal habitats and species). There has been a suggestion that a focus for environmental management should centre on the land-sea interface, which has historically been used as a natural division for planning exercises (Heyman & Kjerfve 1999). The paradigm of integrated coastal management attempts to recognise the importance of the land-sea interface, and the management needs of the coastal marine and coastal terrestrial systems, and the interactions between the two systems.

Approaches to marine conservation and management include regulation of activities such as commercial and recreational fishing or the protection of particular species, ecosystems or habitats considered to be endangered or threatened. There is however a history of fragmented decision-making, limited coordination between management of the marine environment and adjacent coastal lands and tension between local communities, industry and conservation practitioners (Fast et al. 2001; Jones 2006). Managing increasing threats to habitats and species in the coastal zone has been approached through single species management or the *ad hoc* and fragmented declaration of MPAs to conserve high profile ecosystems or species. This has often been undertaken based on little or no ecological data (Vanderklift & Ward 2000). A precautionary approach to management and regulation of coastal activities has developed due to the limited knowledge concerning the impacts of many threats to the coastal zone (Gray et al. 1991). This approach has also been important when considering the conservation of biodiversity, which is an important part of the coastal management framework.

1.2.2 Coastal zone management in Australia

Coastal management in Australia has been described as a complex inter-woven management regime focused on regulating an intensifying range of resource users exploiting the coastal zone (Holmes & Saenger 1995). Throughout Australia there are generally three tiers of Government (i.e. Commonwealth, State and local) that have a role in managing the coastal zone. Added to this complexity are the administrative arrangements, particularly associated with the large number of legislative tools that can be used to regulate and manage coastal activities. This administrative complexity has led to fragmented and *ad hoc* decisions. Coastal management in New South Wales provides an example of the complexity of managing the coastal zone. There are at least seven State Government agencies and an increasing trend for local councils to be involved in coastal management (Underwood & Chapman 1999a). The involvement of local councils in coastal management has been described as inadequate for solving large-scale problems associated with the coastal zone. The New South Wales Government released the NSW Coastal Policy in an attempt to clarify the strategies and role of the different levels of Government in managing the coastal zone, although it has been described as resulting in further fragmentation of decision-making (Underwood & Chapman 1999a).

In Queensland, three levels of Government are responsible for administering over 50 pieces of legislation. The Queensland Government has attempted to rectify this complexity by developing a State coastal management plan (State of Queensland 2001). This plan recognised the need to establish an integrated approach to coastal management and attempted to provide a basis for integrating management, conservation and use of the coastal zone. However, as with the NSW Coastal Policy the Queensland State coastal management plan has not clearly simplified or defined the responsibilities of agencies responsible for managing the coastal zone.

The recognition that managing each activity in isolation in the coastal zone has failed is leading to an integrated approach to coastal management, in which multiple-use MPAs can play an important role in protecting critical areas from exploitation (Guenette & Alder 2007). While there are numerous tools that can be used to regulate activities in the coastal zone, there is international recognition that MPAs have an important role in providing conservation measures to the marine and coastal environment. Marine protected areas (including marine reserves) are one of many tools available for the protection and management of the coast.

1.3 DEVELOPING A SYSTEM OF MARINE RESERVES

1.3.1 Definition of marine protected areas

The concept of MPAs, marine nature reserves or marine reserves has been used to describe an area protectively managed in order to preserve a site of ecological or scientific interest and/or value (Jones 1994). There is no consistency, in different parts of the world, in the use of the terms and definitions to describe MPAs and their design functions (Attwood et al. 1997a). This has led to confusion and misunderstanding in the community as to the intentions and goals of the protected areas. An internationally accepted definition of protected area is:

> "A clearly defined geographical space, dedicated and managed, through legal or other effective means, to achieve the long-term conservation of nature with associated ecosystem services and cultural values" (Dudley 2008).

The IUCN defined the goal of the global system of MPAs in the formal language of the General Assembly Resolutions:

> "To provide for the protection, restoration, wise use, understanding and enjoyment of the marine heritage of the world in perpetuity through the creation of a global, representative system of marine protected areas and through the management in accordance with the principles of the World Conservation Strategy of human activities that use or affect the marine environment" (Kelleher 1999).

Marine reserves have typically been described as no-take MPAs that prohibit the taking of natural resources but allow access for many non-exploitative commercial and recreational activities. For the purposes of consistency, I will use the term marine reserves in the context of establishing a system of no-take areas, whether this is within a multiple-use marine park (i.e. no-take zones), or individual areas that are closed to all extractive activities.

The definitions of MPA have been considered to be too broad and vague to have value (Ballantine 1999; Faye 1999). However, the value of a specific definition is that it is likely to focus politicians and the wider community on marine conservation and the development of initiatives to progress conservation initiatives.

The inconsistent use of terms to describe MPAs and their functions occurs internationally and throughout Australia. For example in Western Australia the term marine nature reserve is used to describe an area of sea protected under specific conditions (e.g. a no-take zone) (Department of Environment and Conservation 2008). Whereas in Queensland the same term is applied in a way which may encompass a variety of different zones (e.g. National Park Zone; Scientific Research Zones) (State of Queensland 2000). In other areas though, Western Australia declares marine parks (e.g. Jurien Bay Marine Park) with four management zones that permit levels of use from no-take to general use zones (Department of Conservation and Land Management 2003), which are similar to those applied in Queensland (State of Queensland 2000). The inconsistent use of terms to describe MPAs and marine reserves is a problem that adds to stakeholder confusion concerning the important role of MPAs as a tool for ecosystem management and marine conservation (McNeill 1994; Faye 1999). The problem with such confusion is that it leads to a misunderstanding of the value of MPAs and therefore it is less likely that conservation of biodiversity will be adequately achieved (Agardy 1999).

To provide an international standard for describing the different types of protected areas the IUCN developed six protected area categories into which protected areas can be allocated, depending on their management objective (IUCN/WCMC 1994; Wells & Day 2004; Dudley 2008). Protected areas will be assigned to one of the following categories which define differences in management approaches:

- **Category Ia –** Strict nature reserve: strictly protected areas set aside to protect biodiversity and also possibly geological/geomorphological features, where human visitation, use and impacts are strictly controlled and limited to ensure protection of the conservation values.

Such protected areas can serve as indispensable reference areas for scientific research and monitoring.

- **Category Ib –** Wilderness area: protected areas that are usually large and unmodified or slightly modified areas, retaining their natural character and influence, without permanent or significant human habitation, which are protected and managed so as to preserve their natural condition.
- **Category II –** National park: protected areas that are large natural or near natural areas set aside to protect large-scale ecological processes, along with the complement of species and ecosystems characteristic of the area, which also provide a foundation for environmentally friendly and culturally compatible spiritual, scientific, educational, recreational and visitor opportunities.
- **Category III –** Natural monument or feature: protected areas set aside to protect a specific natural monument, which can be a landform, sea mount, submarine cavern, geological feature such as a cave or even living feature such as an ancient grove.
- **Category IV –** Habitat/species management area: protected areas that aim to protect particular species or habitats and management reflects this priority.
- **Category V –** Protected landscape/seascape: protected areas where the interaction of people and nature over time has produced an area of distinct character with significant ecological, biological, cultural and scenic value: and where safeguarding the integrity of this interaction is vital to sustaining the area and its associated nature conservation and other values.
- **Category VI –** Protected area with sustainable use of natural resources: protected areas that conserve ecosystems and habitats, together with associated cultural values and traditional natural resource management systems.

1.3.2 Marine reserves – a management tool

The declaration of marine reserves is a developing phenomenon worldwide, although it has been noted that progress continues to be slow (Kriwoken & Haward 1991). Historically, marine reserves have been used to conserve sensitive, unique and significant areas while providing a right of access to the public (Harrison & Parkes 1983; Bernd-Cohen & Gordon 1999; Mascia 1999). The use of marine reserves as a management tool is consistent with the principles of coastal zone management.

Historically the identification and selection of many marine reserves (and other types of MPAs) has been in response to growing awareness of the need to manage and protect natural resources and conserve habitats that are exploited by humans (Alder 1996). This has resulted in the fragmented and opportunistic declaration of areas for protection. The effectiveness of such an approach remains untested and when combined with inadequate on-ground management has led to the term ‘paper parks’, which refers to marine reserves that fail to protect representative samples of biodiversity and usually have limited compliance and enforcement capabilities (Kelleher et al. 1995; Edyvane 1996; Underwood & Chapman 1999a; Gladstone 2001).

The concept and design of national parks for protection of terrestrial systems has been used as the framework for protecting the marine environment, however such a framework does not directly transfer to the marine environment. Terrestrial national parks can be managed sustainably as closed systems, providing they are of sufficient size to maintain ecological processes. National parks can and do exist as isolated natural areas within a matrix of disturbed and impacted areas (i.e. areas altered by human activities). It is generally thought that marine reserves however exist within a natural setting, although in coastal areas the adjacent terrestrial environment may be highly modified. An important difference is that the marine environment is open constantly to external influences throughout the entire protected area. Many of these external influences can not be easily mitigated or managed.

There are other differences in the management principles of marine reserves compared with their terrestrial counterparts. An important difference between marine and terrestrial management is common property rights. There is a notion that the ocean is a common property resource that can be used by everyone to do anything (Faye 1999). Management agencies are able to buy dedicated sites for terrestrial national parks to conserve the target ecosystem, habitat or species. In contrast, the purchase of the seabed and/or the water column remains almost impossible. Therefore management agencies are faced with significant difficulties in selection and subsequent management of areas dedicated for marine protection in light of existing uses.

In Australia the most frequently used approach to marine conservation has been to establish large multiple-use MPAs with a management framework where only small areas are protected in the most restrictive or highest level of protection (i.e. marine reserves) (Kelleher & Kenchington 1992; Kelleher 1999). The zoning of such areas has resulted in only a small percentage of ecosystems and habitats protected in marine reserves where extraction is generally not permitted. Typically other zones in multiple-use marine parks allow for a range of extractive uses (Day 2002). A number of multiple-use marine parks have been declared throughout Australia in an attempt to conserve important components of marine and coastal biodiversity. For example, in Queensland, Moreton

Bay Marine Park was declared in 1993 recognising the importance of the Bay to dugong, migratory birds, mangroves, seagrasses and coral communities (State of Queensland 1998). The Solitary Islands Marine Park was declared in 1998 in New South Wales, which through the zoning plan has attempted to protect a representative range of habitats in marine reserves. This particular Marine Park represents one of the first marine parks in Australia to be declared with the primary objective of conserving a representative example of the marine ecosystems and habitats in a biogeographic region. The outcome of the zoning process has seen the protection of 12% of the marine park in marine reserves (NSW Marine Parks Authority 2001).

Recognition of the biodiversity and conservation values of the Great Barrier Reef led to its declaration as a multiple-use marine park in 1975. The declaration of the Great Barrier Reef Marine Park was primarily related to specific issues including the need to protect the reef from limestone mining, oil drilling, and plagues of crown-of thorns starfish (Hopley 1989). Within the Great Barrier Reef Marine Region there are a large variety of reef and non-reef ecosystems and habitats including soft sediment ecosystems, continental islands, sponge gardens and extensive areas of shallow and deep water seagrass (State of Queensland 1999). The primary focus of marine reserves was on coral reefs and not on less-known and less-spectacular habitats (Maniwavie et al. 2000). In recognition of the inadequacies of the original zoning plan, the Great Barrier Reef Marine Park Authority completed a representative areas program that aimed to identify the range of reef and non-reef ecosystems and habitats in the region. The representative areas project led to rezoning of the marine park with a system of marine reserves that protects a representative range of reef and non-reef bioregions (Great Barrier Reef Marine Park Authority 1998; Fernandes et al. 2005).

1.3.3 Marine protected areas – a conceptual framework

A conceptual framework to illustrate the steps involved with the selection, planning and management of multiple-use MPAs was suggested by Kelleher (1999) (Figure 1). This framework illustrates several important scientific, social and political considerations that contribute to successfully establishing a multiple-use MPA, which contains marine reserves representative of marine ecosystems, habitats and species. Throughout the world there is varying degrees of implementation of these steps to establish multiple-use MPAs, but historically selection of MPAs and marine reserves has been opportunistic and in response to a specific management problem or threat (McNeill 1994).

An important question for conservation practitioners, scientists and the community continues to be ‘How should the location of marine protected areas be chosen?’ This question raises some of the most controversial issues related to multiple-use MPAs and marine reserve design, and in particular the approaches to identifying and selecting sites for marine reserves (National Academy of Sciences 2001). Criteria relating to the scientific, social, economic and feasibility or practicality in the choice of any particular site have been described to guide the identification and selection of MPAs and marine reserves (Kelleher 1999). These guidelines generally provide little information on the relative importance of different criteria nor the development of priorities for declaration. The collection of information related to the scientific importance of a site is a key step that provides a basis for identification of sites, although this aspect has often not been adequately considered during the site selection steps (Figure 1).

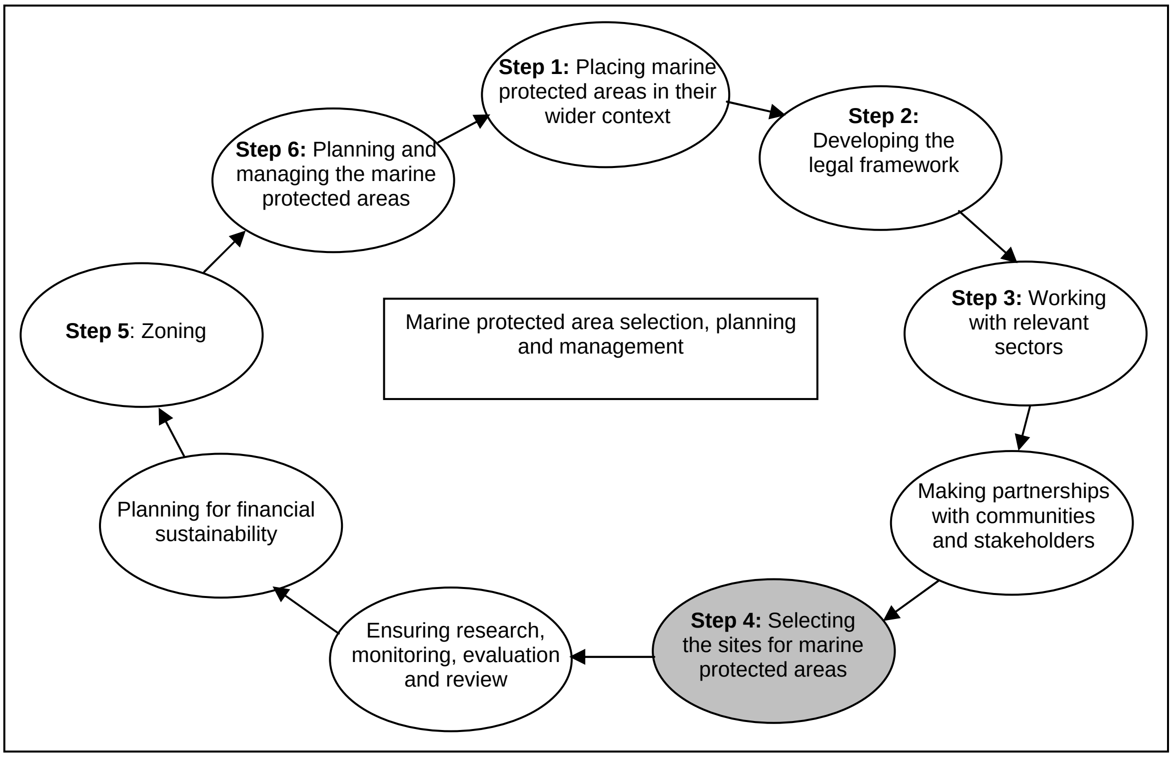


Figure 1: Steps involved in the identification, planning and management of marine protected areas (Kelleher 1999; National Academy of Sciences 2001).

Management of marine reserves is complicated by the nature of the boundaries that surround such areas as they are potentially open to a range of external impacts including the transport of sediment, pollutants and organisms via currents. The effectiveness of marine reserves can be dependent on an integrated coastal zone management framework, which requires wider management of pollution and human activities that threaten biodiversity (Pullen 1997; Boersma & Parrish 1999). Therefore an integrated approach to marine conservation and management must be adopted, which involves consideration of potential impacts both within, and external to candidate marine reserves. The final selection of marine reserves in the context of threats is crucial to their long-term effectiveness.

The effectiveness of measures to protect marine biodiversity has primarily been assessed in relation to commercially important species (see for example Childress 1997; Kelly et al. 2000). Broader issues, such as the conservation of different levels of biodiversity (e.g. species, genetic) including ecosystem function, have rarely been assessed. A conceptual framework has been developed to assess the effectiveness of terrestrial protected area management and whether the site or system achieves broader conservation goals and objectives (Hockings et al. 2000, 2006). Information needed to measure effectiveness of protected areas includes biological and cultural significance of the protected area, threats, and the vulnerability of the protected area to threats. This information is often absent or costly and difficult to obtain for the marine environment.

1.3.4 Recent developments in marine reserve theory – system planning

Establishing a system of reserves is viewed as a priority to sustain biological diversity (Pressey et al. 1993). In an attempt to establish marine reserves that will comprehensively and adequately achieve conservation objectives, marine conservation policy has increasingly focused on establishment of a system of marine reserves, analogous to the terrestrial reserve system. During the last 20 years, a global initiative to protect representative examples of marine ecosystems, communities and species has developed (Kelleher et al. 1995). This has resulted from recognition of the importance of biodiversity and concerns about its increasing exploitation (Kamppinen & Walls 1999). Developing systematic approaches to marine biodiversity conservation follows an historically *ad hoc* approach to declaration of marine reserves. The *ad hoc* approach is considered to be inefficient in achieving conservation goals (Pressey 1994, 1997). The representative system approach to marine conservation and management, links more closely with a broader integrated coastal management framework that aims to manage and conserve marine and coastal habitats and processes.

While the development of a representative system of marine reserves is potentially analogous to the development of a representative terrestrial reserve system, the marine environment tends to be viewed as a common property. Given the common property nature of the marine environment the most controversial issue in designing a system of marine reserves is deciding where to put them, particularly since the acceptance of the concepts of terrestrial national parks can not be transferred directly to the marine environment (National Academy of Sciences 2001). This has resulted in a greater need to justify, to the community, the need for inclusion of areas where activities will be banned or regulated. Increasingly the community expects a clear explanation and rationale for determining sites suitable for protection.

Increasing emphasis is directed towards the rationale, criteria and guidelines for the identification and selection of areas for marine reserve status (Kenchington 1990; Kelleher et al. 1995; Done & Reichelt 1998). This has resulted from recognition by scientists and conservation practitioners that the current approach fails to achieve conservation goals. There is also an increasing interest in conservation by the community leading to the need for greater justification of sites that are high priority for conservation measures where there will be restrictions on use for some stakeholders. Criteria have been described that can be used to design a system of marine reserves (Kelleher 1999), although defining them in a scientific context for marine biodiversity remains a difficult task.

The theoretical approach to the identification and selection of sites to be included in a system of marine reserves includes steps that consider the scientific basis for identifying sites that subsequently could be selected based on consideration of social and economic values (Figure 2). There is an assumption that conservation objectives will be achieved both comprehensively and adequately by following this approach to marine reserve identification and selection. There is however, little evidence that conservation objectives will be achieved using the theoretical model shown in Figure 2. There is a need to narrow the gap between the theoretical classification and scientific representation of the marine realm, their use for identifying sites that contain high conservation values and the use of tools for reserve system selection and design.

1.3.5 Developing a system of marine reserves

To facilitate the development of a system of marine reserves it is widely accepted that the model used to develop the terrestrial reserve system provides a sound basis for identifying, selecting and planning a representative system to protect marine biodiversity. The approach used in the terrestrial realm was a biogeographic approach, which has been developed and refined over 40 years (Udvary 1975). A hierarchically derived bioregional framework attempts to order the mosaic of various kinds of ecosystems and habitats. In particular the approach attempts to consider the hierarchically scaled spatial and temporal nature of physical and ecological processes that define ecosystem function and structure (Edyvane 1996). It involves the subdivision of the marine system into smaller spatial units, which is required due to the difficulties associated with mapping broad-scale biological distributions (Llewellyn et al. 2000).

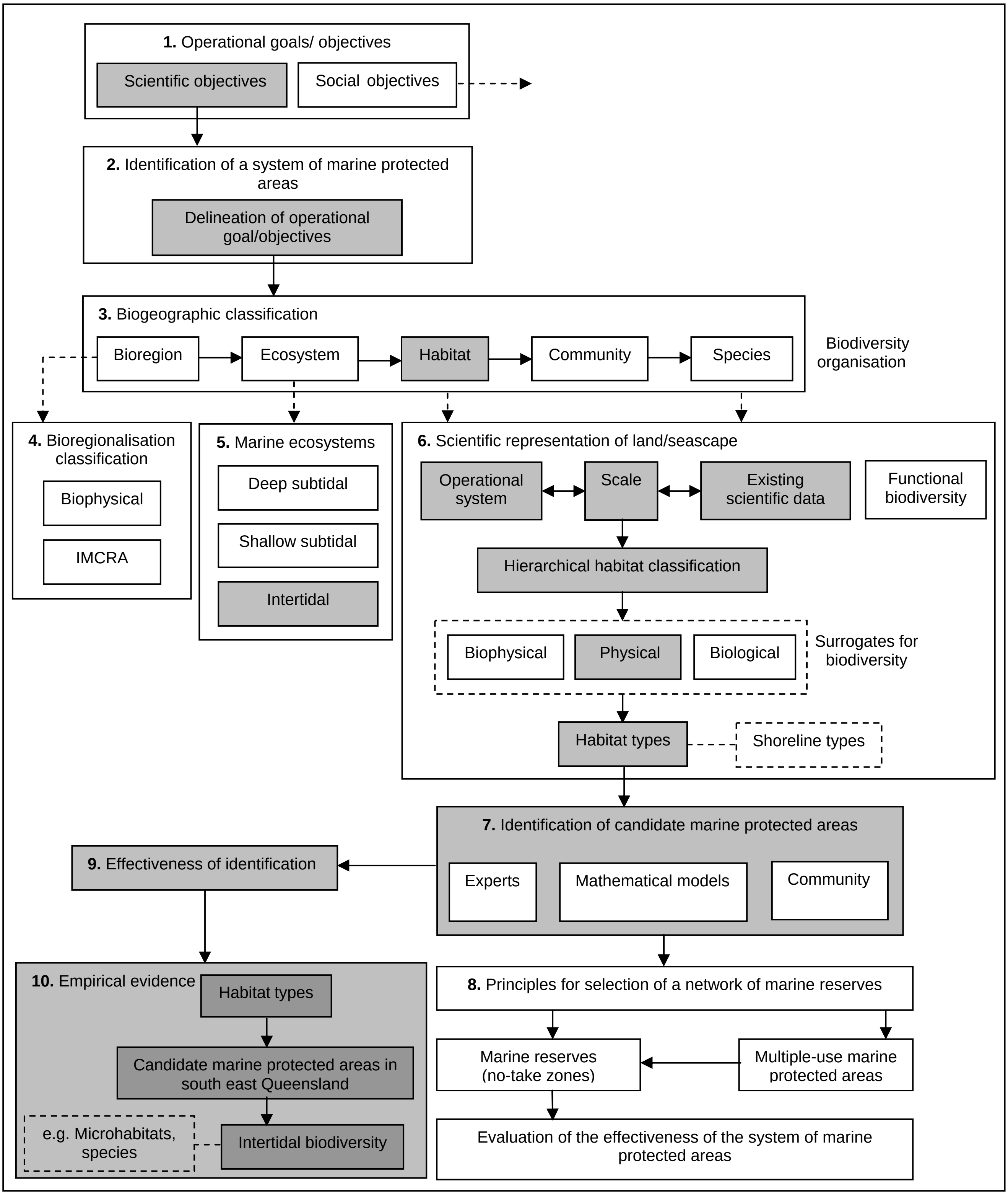


Figure 2: Theoretical model to illustrate the process of identifying and selecting sites to be considered as part of a representative system of MPAs (Kelleher 1999; National Academy of Sciences 2001).

Many government agencies have embraced an ecosystem-orientated approach to conservation (Grumbine 1994; Anderson et al. 1999). Ecosystem management is based on a broad-scale approach to conservation and planning. In Australia, the first attempts to use a broad-scale ecosystem based approach to marine conservation were based on the CONCOM classification (CONCOM 1985). This classification has been further refined into bioregions as part of the Interim Marine and Coastal Regionalisation for Australia providing the broad-scale (100s to 1000s of kilometres) scientific basis for MPA establishment (IMCRA Technical Group 1998).

1.3.6 Objectives and principles of a system of marine reserves

The current approach to biodiversity conservation has evolved to focus on the conservation and protection of representative ecosystems, habitat and species in the terrestrial environment (Margules & Usher 1981; Margules et al. 1988; Margules & Stein 1989; Pressey & Logan 1998). It is only in the last 20 years that similar approaches have developed in the marine environment (Alder 1996; Australian and New Zealand Environment and Conservation Council 1999; Ballantine 1999). The choice of sites for protection has historically been based on the opportunistic selection of sites worthy of protection rather than based on a systematic effort to conserve high quality and enduring examples of ecological communities, which will protect the majority of biodiversity (Anderson et al. 1999). The systematic identification of sites for protection commenced as part of an approach to developing a system of marine reserves. Defining scientific and social objectives is an important step in development of a system of marine reserves.

1.3.6.1 Scientific objectives

To guide MPA system establishment to achieve biodiversity conservation goals, defining scientific objectives and principles is an important initial step (Figure 2 – Box 1). It has been recognised that defined criteria or principles are needed to assist with determining, and prioritising sites to be included in a system of marine reserves. Of high importance in relation to the goal of biodiversity conservation are ecological criteria, which have been defined to include:

- ecological processes or life-support systems;
- integrity, or the degree to which an area, either alone or in association with other protected areas, encompasses a complete ecosystem;
- the variety of habitats; presence of habitat for rare or endangered species;
- presence of nursery or juvenile areas;
- presence of feeding, breeding or rest areas;

- existence of rare or unique habitat for any species; and
- degree of genetic diversity within species (Kelleher 1999).

The application, and identification of areas that meet these criteria remains a challenge to conservation practitioners and scientists.

In an attempt to simplify the identification of areas to conserve biodiversity in Australia these criteria have been summarised, based on a biogeographic framework, into the following principles that have been used for developing a national representative system:

- **Comprehensiveness –** includes the full range ecosystems recognised at appropriate scale within and across each bioregion.
- **Adequacy –** the maintenance of the ecological viability and integrity of populations, species and communities.
- **Representativeness –** those marine areas that are selected for inclusion in reserves should reasonably reflect the biotic diversity of the marine ecosystems from which they derive (Australian and New Zealand Environment and Conservation Council 1999).

While these criteria provide a guide to establishing a system of marine reserves there is difficulty with interpreting the meaning of the principles and their application in reserve system design at a local level (10s to 100s of metres). Concepts such as irreplaceability, used in terrestrial reserve system design, are also increasingly becoming an important part of the decision-making process for the design of a system of marine reserves. Irreplaceability has been defined as: (i) the potential contribution of a site to a reservation goal; and (ii) the extent to which the options for reservation are lost if the site is lost (Pressey et al. 1993).

Defining scientific objectives for the establishment of a system of marine reserves remains a challenge to conservation practitioners. Generally, the goal is to establish a system that protects representative examples of biodiversity in marine reserves. The complexity of defining biodiversity has been generally discussed as including: (a) measures from a number of different individuals (genetic diversity), (b) different species (species diversity), or (c) places (habitats or ecosystem diversity) (Underwood & Chapman 1999b). The findings of Underwood and Chapman (1999b) suggested that biodiversity must be defined as the variability in the number of species and their relative abundances within a habitat, however, the application of such a definition in designing a system of marine reserves that conserves a representative example of biodiversity is potentially problematic over large biogeographic areas. The validity of using habitats as a surrogate for biodiversity, as defined above, has not been tested adequately at large spatial or temporal scales.

1.3.6.2 Social objectives

While there are ecological consequences associated with over-exploitation and limited management of a species or a site there are also likely to be social and economic values affected due to the establishment of marine reserves to prevent such adverse outcomes. Therefore, the process of selecting sites for protection attempts to recognise the economic, indigenous, social and scientific interests of an area (Australian and New Zealand Environment and Conservation Council 1999). In recognition of the importance of the marine environment to tourism, recreation and a variety of industries a number of criteria have been developed to consider the social and economic values of an area. These criteria are proposed to be considered during the prioritisation and selection of sites for inclusion in a system of marine reserves.

Criteria recommended for consideration during the selection stage include:

- **Economic importance –** existing or potential contribution due to protection.
- **Social importance –** existing or potential value to local, national and international communities because of its heritage, historical, cultural, traditional, aesthetic, educational or recreational qualities.
- **International and national significance –** existence of any national or international designation.
- **Scientific importance –** value for research and monitoring.

The approach to designing many marine reserves to cater for social and economic consideration has been primarily based on a multiple-use model (i.e. multiple-use marine parks), which separates conflicting uses (see for example Lynch et al. 2004) and requires that a core area is protected in marine reserves. The multiple-use approach for MPAs still requires that consideration be given to identifying and protecting sites that contain high conservation values.

It has been suggested that the identification of marine reserves could use commercial fishing data, depth and bioregions as a surrogate for an ecosystem classification (Manson & Die 2001). Selecting sites for candidate marine reserves based, in the first instance, on use by the fishing industry is unlikely to achieve a goal of conservation of biodiversity. It may be necessary to include some areas targeted by commercial fishing in marine reserves to protect crucial habitats that coincide with locations of commercially important species. The importance of the selection stage in designing a system of marine reserves is to maximise conservation efforts while attempting to minimise impacts on industry and other users of the marine environment.

The generally accepted approach for designating marine reserves is to identify candidate sites containing high conservation values based on scientific information or biodiversity surrogates before consideration of social and economic interests. Surrogates aim to predict species diversity, and distribution with minimum sampling effort and cost (Goldberg et al. 2006). Information such as commercial fishing data would then be incorporated to assist with selecting and prioritising the candidate sites, which can minimise impacts on industry while achieving the goal of conserving biodiversity, for implementation.

For example, in recognition of the inadequacies of the 1975 zoning plan for the Great Barrier Reef Marine Park the Great Barrier Reef Marine Park Authority completed a major review of the zoning plan (Great Barrier Reef Marine Park Authority 1998; Day et al. 2002). The review, known as the representative areas program, involved: (i) the collation of more than 40 layers of biological and physical information in a geographic information system; (ii) classification and analysis of this information by reef and non-reef experts; and (iii) the development of bioregions, which were assumed to be typical of the surrounding habitats or ecosystems at a chosen scale (Great Barrier Reef Marine Park Authority 1998; Fernandes et al. 2005). At the same time additional information was collected relating to recreational and commercial use of the marine park area, which assisted with the identification of candidate sites for marine reserves. Following the identification of candidate sites a process to select the most appropriate sites was undertaken that involved formal public participation. The new zoning plan restricted use and entry to certain areas in order to protect representative examples of biodiversity (reef and non-reef bioregions) (Day et al. 2002; Fernandes et al. 2005).

1.3.7 Identification of priority conservation areas

Although it is clear that there is an urgent need to identify and select suitable sites for designation as marine reserves (Pullen 1997), there are few specific biogeographic approaches that are able to identify suitable candidate sites at local-scales (10s to 100s of metres). There are very general schemes such as the Interim Marine and Coastal Regionalisation for Australia (IMCRA Technical Group 1998) or, at the other end of the scale, local studies that examine the specific details of the ecology of particular species or habitats (e.g. Dawson & Slooten 1993; Davis 1995; Underwood & Chapman 1999a). The challenge for conservation practitioners is to collate information that enables decisions to be made at local levels (10s to 100s of metres) within the context of broader biogeographic (100s to 1000s of kilometres) patterns of biodiversity.

The focus of attention for selection of sites for protection as marine reserves has been on high profile ecosystems such as coral reefs, following the trend in terrestrial systems to historically focus conservation efforts on icon or high profile fauna, which has usually been strongly influenced by political and or community desire for the protection of charismatic mammals. For example, Hervey Bay Marine Park was declared in 1989 primarily to manage the whale watching industry of Hervey Bay, with little consideration of the need to protect other components within this large coastal embayment, including important fish nursery areas.

1.3.8 Scientific representation of the land/seascape

The availability of data relating to the spatial distribution of the different levels of biodiversity (e.g. genotypes, species, habitat types etc.) within a biogeographic region are limited at a spatial resolution suitable for marine reserve establishment. Field studies can directly survey only a small fraction of the total area within a region. The problems associated with the direct measurement of the spatial distribution of biodiversity within a region have been frequently addressed by using a surrogate as a measure of biodiversity (Pressey & Ferrier 1995; Ferrier 1997).

Inadequate descriptive knowledge of the marine realm has led to an increasing trend to develop biodiversity surrogates as a means to identify areas of high conservation value. Surrogates might be based on particular taxa, specific species assemblages or communities, or environmental variables (Faith et al. 2001). The development of classification systems to define surrogate measures of biodiversity provide a basis for describing or categorising the marine environment into spatial units that can support the identification, planning and management of marine reserves. Further research is required to determine whether the surrogate measures provide a reliable and effective approach to identifying biodiversity patterns to protect.

A hierarchically designed biogeographic framework has been used to develop surrogates for marine biodiversity (Cowardin et al. 1979; Dijkema 1991; Dethier 1992; Mumby & Harborne 1999). These schemes provide a basis for describing and mapping the marine system at a range of spatial scales. Many classification schemes advocate the use of physical and biological information to identify areas that are considered to represent particular habitat or community types (Dijkema 1991; Mumby

& Harborne 1999). The practical application of a surrogate measure to determine conservation priorities and achieve conservation goals in the marine environment has received little attention in the literature and therefore limited empirical evidence is available to evaluate the success of marine reserves to effectively conserve representative examples of marine biodiversity.

The most common approach to developing surrogates for the marine environment has been to use physical and oceanographic variables (Dethier 1992; Zacharias et al. 1998). The rationale for using these variables is that they are considered to provide a relatively stable and consistent basis for identifying areas exposed to certain physical and oceanographic features, which were described as constraining the distribution and interactions of marine plants and animals. These studies have been undertaken to assist with the development of a representative system of marine reserves and have tended to be over a large-scale (i.e. ecosystem). There are few empirical studies that have investigated the relationship between ecosystems, defined by physical variables, and diversity at finer scales such as habitats or community. There has however been a number of studies at fine-scales (<5 kilometres) that have investigated a suite of physical variables and their influence on the distribution of species (Coates 1995; Schoch & Dethier 1996). A gap in our knowledge relates to classification at intermediate scales (i.e. habitats), particularly over large expanses of the marine and coastal systems, and the relationship between habitat surrogates, and lower levels of biodiversity (e.g. microhabitat, and species diversity and abundance).

1.3.9 Designing marine reserve systems

Conservation decisions have been described as having three broad steps:

(i) defining the selection problem;

(ii) preparing the data that underpins the selection problem; and

(iii) running the reserve selection algorithm to find solutions to the problem (Possingham 2001).

There has been a new wave of theory concerned with designing marine reserves, which includes the development of computer algorithms (e.g. simulated annealing) to forecast the benefits of selecting one site over another (Causey 2000; Possingham et al. 2000; McDonnell et al. 2002; Leslie et al. 2003). These tools are becoming quite sophisticated as they consider a range of options and factors in determining sites to be included in a reserve system. Studies have demonstrated that there are many different combinations of systems that could meet habitat representation targets (Possingham et al. 2000; Leslie et al. 2003). The use of siting algorithms has a number of potential limitations in application to areas where there are limited data available on the distribution of species. This may be able to be solved through the use of suitable, and tested surrogate measures for biodiversity.

The outcomes of various reserve system scenarios forecast to achieve conservation targets have rarely been systematically surveyed to determine the representation of biodiversity and therefore the effectiveness of siting algorithms in prioritising areas that contain high conservation values to achieve conservation goals. The results of a habitat classification and computer modelling to evaluate the use of a habitat surrogate and siting algorithm as a means to cost-effectively identify a representative system of marine reserves remains largely untested.

While there is suspicion related to the effectiveness of using siting algorithms to identify sites for protection to achieve conservation goals, it is considered that such suspicion should lie with the dataset that underlies the reserve selection problem (Possingham 2001). Siting-algorithms are a tool to assist with decision-making in relation to designing a system of marine reserves and require sufficient information related to ecosystems, habitats or biological distributions over large areas. Alternatively, information related to surrogate measures of biodiversity such as biophysical factors, keystone species or indicator species is required (Gladstone 2002). There is often a lack or incompleteness of detailed biological diversity inventories whether it be for the distribution of species or habitats that provides the underlying dataset for the reserve selection problem (Gladstone 2002). There have been few empirical studies at appropriate spatial and temporal scales investigating the link between the theory of site selection and practical outcomes for biodiversity conservation.

1.3.10 Approaches to designing a system of intertidal marine reserves

There has been limited attention given to the conservation of intertidal habitats, which have historically represented a planning boundary between marine and terrestrial systems (Heyman & Kjerfve 1999). Intertidal organisms are the focus of collection activities for bait, human consumption and can be impacted as a result of development activities and trampling (McPhee & Skilleter 2002; McPhee et al. 2002). Intertidal habitats are often very accessible making them vulnerable to human impacts and collection activities, particularly adjacent to urban areas. Recognition of these threats has led to the development of management tools to remove or reduce human influences on impacted areas.

There are limited data available that could be used to assist a process for designing a system of intertidal marine reserves for extensive areas of coastline. The spatial and temporal variation of intertidal organisms on rocky shores has been well documented, although for only a few places throughout the world (e.g. Underwood & Petraitis 1993; Underwood & Chapman 1998a, 1998b). The value of experimental studies that focus on small sections of coast is that they provide a better understanding of the processes that affect intertidal species assemblages. However, increasing our knowledge of the landscape ecology of coastal habitats and how ‘intertidal landscapes’ interact is required to better understand the requirements of marine reserves for conserving intertidal biodiversity (Underwood & Chapman 1999b).

The effectiveness of intertidal marine reserves has been investigated in a number of locations from around the world with results indicating a variable response by intertidal assemblages to closures. There is evidence that indicates that marine reserves can result in recovery, and or an increase in the abundance of target species (Castilla & Duran 1985). The failure of intertidal marine reserves was demonstrated by an experimental study of intertidal protected areas near Sydney (Underwood & Chapman 1999a). This study found that the declaration of intertidal marine reserves failed to reduce human collection activities and therefore recorded limited differences in the species assemblages between sites declared as intertidal marine reserves (closed to harvesting) and sites open to harvesting activities. The study concluded that the failure for the intertidal marine reserves to conserve biodiversity was related to continuing use by humans and a lack of management or policing, which plays an important role in their effectiveness. The study provided evidence of the failure for intertidal marine reserves to change significantly species assemblages, in particular those species that were harvested by humans. The success or failure in this particular case should not be directed as a failure of the marine reserves concept, but was found to be associated with the failure of on-ground management and policing.

1.4 THESIS OVERVIEW

There are many areas of interest to the scientific community related to the identification and selection of marine reserves. It is unrealistic to validate each component of the theoretical framework for developing a system of marine reserves (Figure 1), which would need to include the identification and selection processes across a range of ecosystems and habitats or species. The need for scientific information and empirical evidence to support the identification and establishment of marine reserves is well known. To progress our understanding of the role of science in the identification of marine reserves, I reviewed current theory concerning the identification and selection of sites and examine whether there is evidence to support site selection in achieving conservation goals.

The purpose of this study is to develop a local-scale habitat surrogate to represent intertidal biodiversity that can be used in a mathematical reserve design algorithm to identify a representative system of intertidal marine reserves throughout Queensland. Intertidal habitats are very accessible and are threatened or vulnerable to a wide variety of impacts. The development of tools to progress their conservation is a priority. Chapter 2 of this dissertation outlines an intertidal classification that will be used to describe the intertidal habitats for the Queensland coast. The objectives are: (1) to describe the physical properties of Queensland's intertidal shoreline and use them to classify the coast into habitats; and (2) to evaluate the current protective status of Queensland's intertidal habitats as defined by the physical properties.

Chapter 3 describes an improvement over the previous attempts to apply a habitat surrogacy approach to reserve selection, as it is based on a fine-scale (10s to 100s of metres) intertidal habitat classification that has been applied consistently to 24,216 kilometres of the Queensland coastline. I describe a process for the systematic identification of sites for inclusion in a system of marine reserves that would protect representative examples of the full range of mainland intertidal habitats in Queensland. I evaluate the success of different reserve system scenarios in achieving conservation targets and the potential influences of reserve boundary compactness and the relative cost of each solution in identifying sites to be included in a representative system.

Chapter 4 aims to examine whether the presence/absence of microhabitats, or the distribution and areal extent of different intertidal habitats varied between and within rocky shores in south east Queensland, and how inclusion of this information affected marine reserve selection. I also examined different scenarios for a marine reserve system where additional information on the spatial extent of habitats or the presence/absence of microhabitats was not included. These scenarios were compared with scenarios when information on these features was included to identify how the inclusion of habitats or microhabitats will influence reserve system solutions.

Chapter 5 discusses how to move from scientific and theoretical approaches for establishing a system of marine reserves to a practical plan for forming a network of marine reserves. The chapter discusses: 1) the role of reserve network goals and criteria for identifying sites for marine reserves; 2) the scale (i.e. fine- and large-scale) at which surrogate measure of biodiversity can be applied and the relative importance of identification criteria in decision-making; and 3) provides guidance on the pragmatic implementation of marine reserve networks. I discuss these factors based on approaches to marine reserve network implementation in New South Wales (Australia) and New Zealand.

CHAPTER 2 — MAPPING INTERTIDAL HABITATS AND AN EVALUATION OF THEIR CONSERVATION STATUS

2.1 INTRODUCTION

Marine and coastal ecosystems are under increasing threat from human pressures (Gray 1997) and it is recognised that representative examples of marine biodiversity require protection (Kelleher et al. 1995). The ultimate goal of biodiversity conservation is to conserve diversity at three levels: ecosystem, species and genetic diversity (World Resources Institute 1992). Increasingly biodiversity conservation has focused on the protection of representative ecosystems, habitats and species in the terrestrial environment (Margules & Usher 1981; Margules et al. 1988; Margules & Stein 1989; Pressey & Logan 1998). Recently, similar approaches have developed in the marine environment (Alder 1996; Australian and New Zealand Environment and Conservation Council 1999; Ballantine 1999) with global and national initiatives to develop a representative system of MPAs (Kelleher et al. 1995).

A current aim of coastal zone management in Queensland, and elsewhere in Australia, is to determine the most appropriate way of designing and declaring a series of MPAs with the intention of protecting and conserving representative examples of Australia's marine biodiversity. This includes maximising the number and types of species conserved. One of the most controversial issues in designing a system of MPAs is deciding where to locate them (National Academy of Sciences 2001). Historically, an *ad hoc* approach has been taken to declare MPAs rather than selection based on any principles of reserve system design such as comprehensiveness, adequacy or representativeness (Australian and New Zealand Environment and Conservation Council 1999; McNeill 1994; Williams & Bax 2001). The *ad hoc* approach to selecting protected areas has been described as expensive and inefficient in protecting elements of biodiversity (Pressey 1994, 1997). It also appears to favour those habitats that have a lower demand for commercial extraction and therefore at least risk while habitats that are threatened continue to be over-exploited (Pressey et al. 2000). Although it is recognised that sites with high conservation values should be selected based on reserve system design principles, MPAs continue to be selected based on opportunistic or *ad hoc* means as indicated by recent claims that the current size and placement of MPAs falls far short of comprehensive or even adequate conservation objectives (Boersma & Parrish 1999).

The first stage in the selection of MPAs is to identify suitable sites, taking account of all available and relevant scientific information followed by selection based on the consideration of social, economic and cultural values. However, detailed information relating to biological distributions is largely unavailable and surveying them over large areas is costly and time consuming. As a result there has been an increasing use of biodiversity surrogates to determine MPA priorities as an alternative to detailed studies that document the biodiversity of each site. Surrogates attempt to define a biophysical or ecological unit that provides an understanding of natural ecosystems and patterns of biodiversity (Department of Primary Industries, Water and Environment 2000). There is, however, limited empirical evidence to evaluate the success of using surrogate measures to identify protected areas to achieve conservation goals in the marine and terrestrial environments.

A hierarchically designed biogeographic framework has been used to develop surrogates for biodiversity in the marine environment (Cowardin et al. 1979; Dijkema 1991; Dethier 1992; Mumby & Harborne 1999). In this approach bioregions are defined broadly and the objective is to conserve representative samples of each. This approach is better than *ad hoc* but I believe that the bioregions are usually at too coarse a scale. Protecting a sample of each bioregion will miss elements of biodiversity that respond to biophysical features at a finer scale (Pressey et al. 2000).

The selection of sites to be included in MPA systems has generally been undertaken using regional surrogate measures rather than using finer scale surrogate measures that define habitats. The development of surrogate measures (e.g. physical properties) at a finer scale (e.g. habitat) is likely to increase the possibility that a system of MPAs will adequately achieve biodiversity objectives by ensuring protection of a greater range of habitats and species. In Australia, the Interim Marine and Coastal Regionalisation for Australia scheme was developed to provide a national ecosystem-scale regionalisation for planning a representative system of MPAs and defines a surrogate ecosystem classification at a regional scale (100s to 1000s of kilometres) (see Figure 3) (IMCRA Technical Group 1998; Manson & Die 2001). The aim was to identify large units termed ‘bioregions’ that contain similar environmental attributes that can be used as ecosystem surrogates (Manson & Die 2001). The bioregions were defined based on physical (e.g. geology, coastal geomorphology), oceanographic (e.g. tides, currents, water temperature), climatic (e.g. wind) and biological (e.g. distribution of species) factors. While bioregions have been described as suitable for establishing national biodiversity and conservation planning priorities (Department of Primary Industries, Water and Environment 2000) they have also been described as having limited value at this scale as they cover large and heterogeneous areas of the landscape (Pressey et al. 2000). They are less suitable for determining specific sites that should be included in a representative system of MPAs, particularly since biological communities exist and are exploited at a much more local scale (William & Bax 2001).

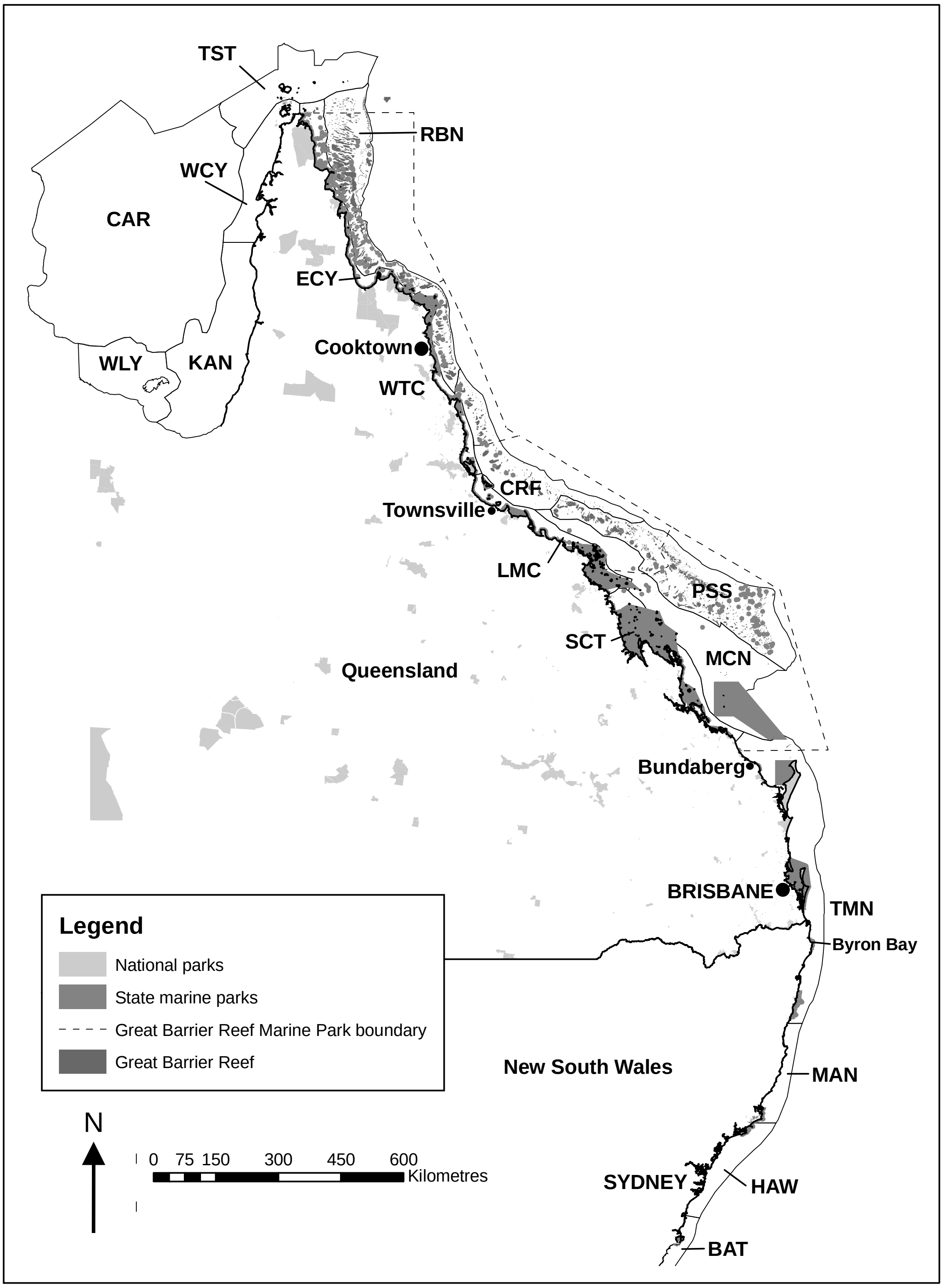


Figure 3: Queensland's bioregions defined by the Interim Marine and Coastal Regionalisation for Australia. (IMCRA bioregions – BAT: Batemans Shelf; CAR: Carpentaria; CRF: Central Reef; ECY: East Cape York; HAW: Hawkesbury Shelf; KAN: Karumba-Nassau; LMC: Lucinda-Mackay Coast; MAN: Manning Shelf; MCN: Mackay-Capricorn; PSS: Pompey-Swains; RBN: Ribbons; SCT: Shoalwater Coast; TST: Torres Strait; TWN: Tweed-Moreton; WCY: West Cape York; WLY: Wellesley; WTC: Wet Tropics Coast).

In Queensland the selection of sites to be included in the existing system of marine parks, declared under the *Marine Parks Act 1982* (Queensland), was not based on the ecosystem surrogate defined by the Interim Marine and Coastal Regionalisation for Australia (IMCRA Technical Group 1998). The focus was on the protection of commercial resources or unique and icon species and ecosystems (e.g. coral reefs (Wilkinson 2000)), and not representative habitats (State of Queensland 1999). Government commitments to the development of a national representative system of MPAs based on the principles of comprehensiveness, adequacy and representativeness (State of Queensland 2000) has led to a need to develop finer scale surrogate measures to assist with identifying suitable sites for MPAs. In addition, data are not available to assess the comprehensiveness, adequacy and representativeness of MPAs (State of Queensland 1999). There have been few studies that have attempted to assess the effectiveness of the marine parks programme in Queensland or elsewhere in Australia to meet the goal of protecting biodiversity and ecosystem processes (Edgar et al. 1997).

Under the Queensland *Marine Parks Act 1982* (Queensland), a marine park boundary generally extends to mean highest astronomical tide and therefore includes the intertidal shoreline, which is threatened by numerous human activities such as development. There has been no attempt to protect representative examples of Queensland shoreline due to the absence of suitable data. Therefore the development of a fine-scale surrogate measure of shorelines would contribute to the future selection of MPAs and the development of a representative system of MPAs that adequately protects intertidal biodiversity. The aims of this section are: (1) To describe the physical properties of Queensland's intertidal shoreline and use them to classify the coast into habitats; and (2) To evaluate the current protective status of Queensland's intertidal habitats as defined by the physical properties.

2.2 MATERIALS AND METHODOLOGY

2.2.1 Shoreline habitat classification

Queensland's intertidal shoreline habitats were classified using a scheme based on the British-Columbia shoreline mapping and classification system (Howes et al. 1994) and the refined Cowardin classification (Dethier 1992). These schemes provide a descriptive mapping system developed to systematically record shoreline morphology, shore-zone substrata and wave exposure at a variety of spatial scales (Table 1).

Table 1: Shoreline habitat types resulting from the classification.

SYSTEM				
Marine Estuarine	Estuaries are usually more sheltered than the open ocean influencing tide movements, currents and wind patterns (Morrisey 1995). The shoreline was segmented into marine and estuarine to reflect the physical and oceanographic differences. In addition, there are numerous management needs specific to estuaries associated with their sheltered nature increasing the use by humans. Estuarine systems along the Queensland coast were identified and classified by Digby et al. (1999).			
	SUB-SYSTEM			
Mainland Island	There is evidence of differences in the intertidal communities between the mainland and islands in Queensland (Coates 1998). These sub-systems also face different pressures from human use and therefore there are different requirements for coastal planning and management.			
CLASS	Substrata has been found to affect the distribution of species and provide the microhabitat requirements of many species (Underwood & Chapman 1995). Substrate may reflect current movements and the sources of sediment, whether it is from rivers or the sea providing different habitat conditions for species (Morrisey 1995). The approach to management of each type of substrata is different as they attract different users. While there are some over-riding shoreline management principles there are also some special management needs for each substrata type associated with their use as a natural resource.			
	SUB CLASS	**Width**	**Slope**	**Shoreline habitat type**
Consolidated	Bedrock	Wide	Steep	n/a
			Inclined	Rock ramp, wide
			Flat	Rock platform, wide
		Narrow	Steep	Rock cliff
			Inclined	Rock ramp, narrow
			Flat	Rock platform, narrow
	Beach Rock	Wide	Steep	n/a
			Inclined	Beach Rock ramp, wide
			Flat	Beach Rock platform, wide
		Narrow	Steep	Beach Rock cliff
			Inclined	Beach Rock ramp, narrow
			Flat	Beach Rock platform, narrow
	Boulder (>1m)	Wide	Steep	n/a
			Inclined	Inclined boulder field, wide
			Flat	Flat boulder field, wide
		Narrow	Steep	Boulder cliff
			Inclined	Inclined boulder field, narrow
			Flat	Flat boulder field, narrow
Unconsolidated	Cobbles	Wide	Steep	n/a
			Inclined	Inclined cobble beach, wide
			Flat	Flat cobble beach, wide
		Narrow	Steep	Steep cobble beach
			Inclined	Inclined cobble beach, narrow
			Flat	Flat cobble beach, narrow
	Gravel	Wide	Steep	n/a
			Inclined	Inclined gravel beach, wide
			Flat	Gravel flat, wide
		Narrow	Steep	Steep gravel beach
			Inclined	Inclined gravel beach, narrow
			Flat	Gravel flat, narrow
	Sand	Wide	Steep	n/a
			Inclined	Inclined sand beach, wide
			Flat	Sand flat, wide
		Narrow	Steep	Steep sand beach
			Inclined	Inclined sand beach, narrow
			Flat	Sand flat, narrow
	Mixed fines	Wide	Steep	n/a

Table 1: continued

			Inclined	Inclined mixed fines flat, wide
			Flat	Mixed fines flat, wide
		Narrow	Steep	Steep mixed fines flat
			Inclined	Inclined mixed fines flat, narrow
			Flat	Mixed fines flat, narrow
Artificial				Piles (jetty)
				Marina
				Rock Wall
Reef		Wide		Fringing reef, wide
		Narrow		Fringing reef, narrow
Additional modifiers				
Wave exposure	Very protected	The importance of wave action as a process that influences the distribution of intertidal species has been well documented (McQuaid & Branch 1984; Underwood & Jernakoff 1984; Bustamante & Branch 1996; Underwood & Chapman 1998b; Gaylord 1999). Due to the lack of broad-scale data for wave energy, relative exposure was calculated. The approach described by Howes et al. (1994) was used to determine relative wave exposure that has been found to agree with the distribution of species assemblages along British Columbian shorelines. The wave fetch window, which is the open water area offshore from the shoreline unit over which waves can be generated by winds, is used to determine relative exposure. Relative exposure is calculated based on the assumption that the larger the fetch window, the greater the wave exposure (Howes et al. 1994). The system was adapted to reflect prevailing swell conditions in Queensland waters.		
	Protected			
	Semi-protected			
	Semi-exposed			
	Exposed			
Tidal range	High	The regular rise and fall of the tides has been documented as influencing the distribution intertidal organisms (Underwood & Chapman 1995). Tidal range was based on the extreme tidal range described by Digby et al. (1999).		
	Medium			
	Low			

The shoreline was subdivided into alongshore units (see Figure 4) and across-shore components displayed as line segments or arcs in a geographic information system (GIS). Attributes used to define the shoreline habitats were the type of substrata present, in combination with variables that are considered as modifiers of the broad relationship between substrata and the ecosystem supported (e.g. the slope and width of the shoreline in particular areas) (Table 1). Each shoreline unit was further segmented based on changes in the across-shore components to provide, for example, an indication of the naturalness of the area, potential threats to the shoreline habitat or to assist with assessing the mosaic of habitats along the coastline. The across-shore components related to the seaward (i.e. subtidal), or landward features (i.e. foreshore and backshore) of the coastline adjacent to each shoreline habitat. Across-shore components were included to provide further detail about the geomorphology of the system and potentially the ecosystems or communities present along a section of coast. These across-shore components were described in relation to their substratum type (e.g. cobbles, boulders and bedrock), vegetation association (e.g. mangroves and saltmarsh) or artificial feature (e.g. rock wall, residential, industry). Thirty-four categories were specified for the across-shore components of Queensland's shoreline (Table 2).

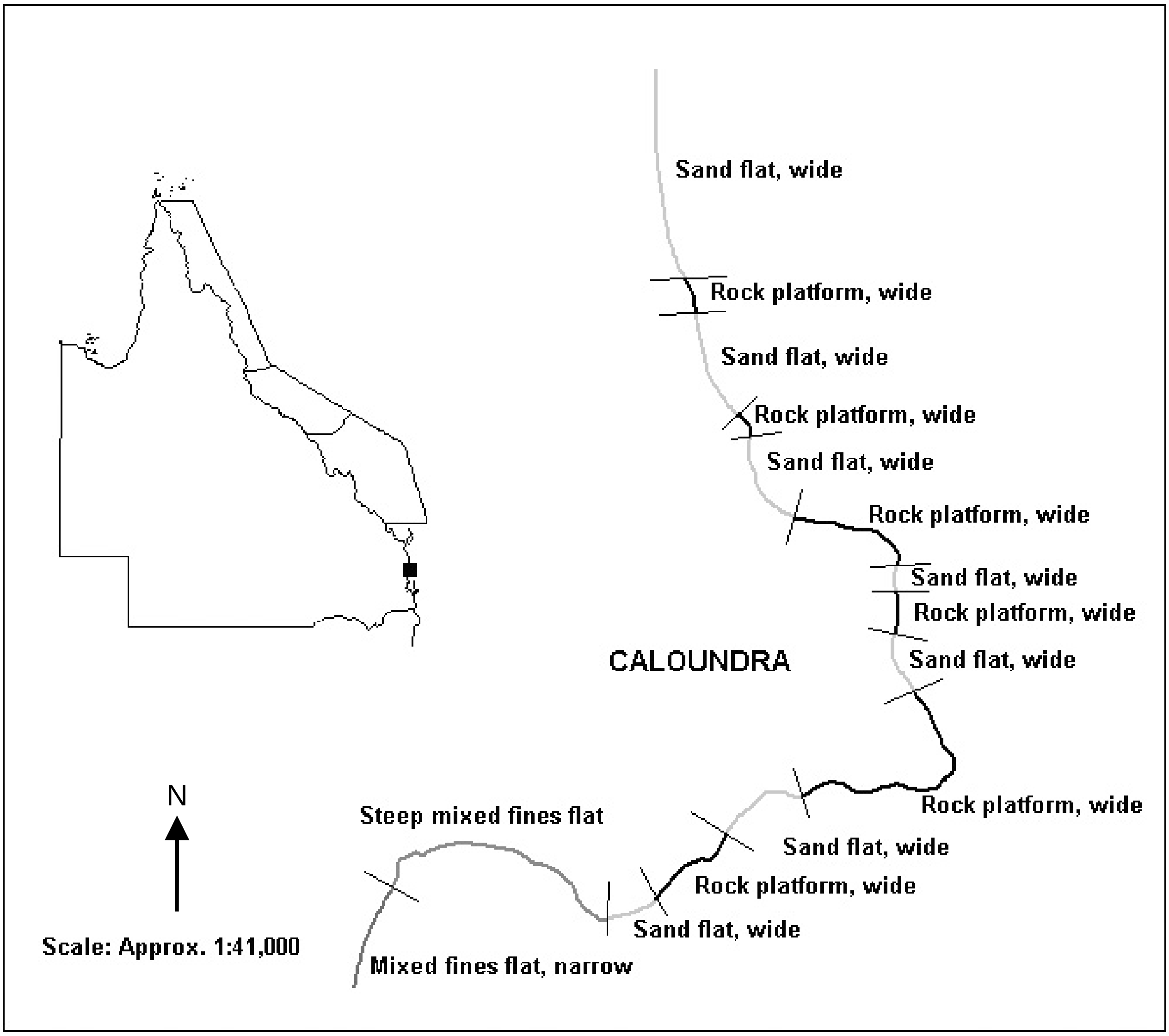


Figure 4: Example of the coastline segmented into shoreline types – Caloundra, Queensland.

2.2.2 Shoreline habitat protective status

Details about the boundaries of marine parks and their respective zoning plans were provided for each shoreline unit in the GIS. The *Marine Parks Act 1982* (Queensland) provides the statutory basis for developing a zoning plan, which may consist of one or more zones. Each type of marine park zone has a purpose for which the zone may be entered or used. For example, a person may enter a Protection Zone (other zones with similar restrictions include National Park B Zone, Scientific, Preservation, or no-take zones) without permission for recreational, educational, cultural or spiritual purposes, however they are prohibited from taking or disturbing marine plants and animals.

Table 2: Categories used as a basis for defining across-shore components of each shoreline type.

Across-shore categories	
Bedrock (platform)	Terrestrial vegetation
Beach rock	Road
Boulder	Residential
Cobble	Industry
Gravel	Parkland
Sand	Amusement park/theme park
Mud	Wetland
Mixed fines	Rocky reef
Piles (jetty)	Agriculture/silviculture
Marina	Coral communities
Rock wall	Fish traps – traditional
Wreck	Resort
Fringing coral reef	Aquaculture
Sand dunes	Runway
Cliff	Research facility
Mangroves	Beach rock
Saline coastal flat	Reef lagoon

2.2.3 Aerial photograph and video interpretation

The characterisation of Queensland's coast was based primarily on vertical aerial photograph interpretation and oblique aerial videography, supplemented by ground-truthing along selected sections of the coast. The attributes assigned to each shoreline unit were derived from aerial photographs with a scale of 1:12,000, which were viewed in their stereo pairs to identify the substrate type (e.g. boulders, cobbles and sand) and physical features (e.g. slope and width). The availability of 1:12,000 scale aerial photographs for shoreline interpretation was restricted to the area between the Qld/NSW border and Cooktown.

Between Cooktown and the Qld/NT border only 1:50,000 scale aerial photographs were available so interpretation from photographs at this scale was limited to the first level of the classification (i.e. consolidated, unconsolidated, artificial and reef). Consequently, this area was flown and videoed at an altitude of between 200 to 400 metres above sea level to improve the resolution of the classification of the different types of intertidal habitat present. Two observers were on the aircraft noting shoreline features. Observer one noted the substrate type for the classification of alongshore segments and recorded details onto hard copy maps at a scale of 1:100,000. Observer two used a 3CCD Digital Video Camera to record the shoreline, which assisted with collection of additional information and capture of images following completion of the flights.

Data relating to each shoreline segment were compiled, analysed and checked from aerial photograph and videography interpretation using ArcInfo and ArcView GIS software. The base map for the classification was a 1:100,000 scale coastline (AUSLIG 1997). The attribute table associated with the GIS was developed based on the hierarchical shoreline habitat classification, Interim Marine and Coastal Regionalisation for Australia bioregions and marine park boundaries. The development of the GIS and shoreline classification represents the first time such an approach has been used to assist with coastal planning and management for the entire length of Queensland's coast.

2.3 RESULTS

The total length of shoreline mapped and classified was 24,216 kilometres. This included shoreline habitats associated with estuaries, and the mainland and islands along the Queensland coast (see Figure 5).

2.3.1 Distribution of shoreline habitat types

Queensland's coastline is a complex mosaic of habitat types, which includes sandy beaches, rocky shores, mangrove communities, fringing reefs and coastal flats (Table 3). The coastline has many rocky habitats interspersed with long sandy beaches and boulder fields. There are rock ramps and a diversity of habitat types associated with the numerous continental islands located offshore from Queensland's coast.

As would be expected, estuarine areas were dominated by mudflats that support extensive areas of mangrove and estuarine beaches. The marine mainland sub-system was dominated by wide flat beaches (45.9%) followed by wide sand flats (28.6%), associated with several large embayments along the coast. The marine island sub-system was also dominated by wide flat beaches (22.2%) followed by fringing reefs (12.7%) (Table 3), which were recorded in the Torres Strait and islands in the Great Barrier Reef Region.

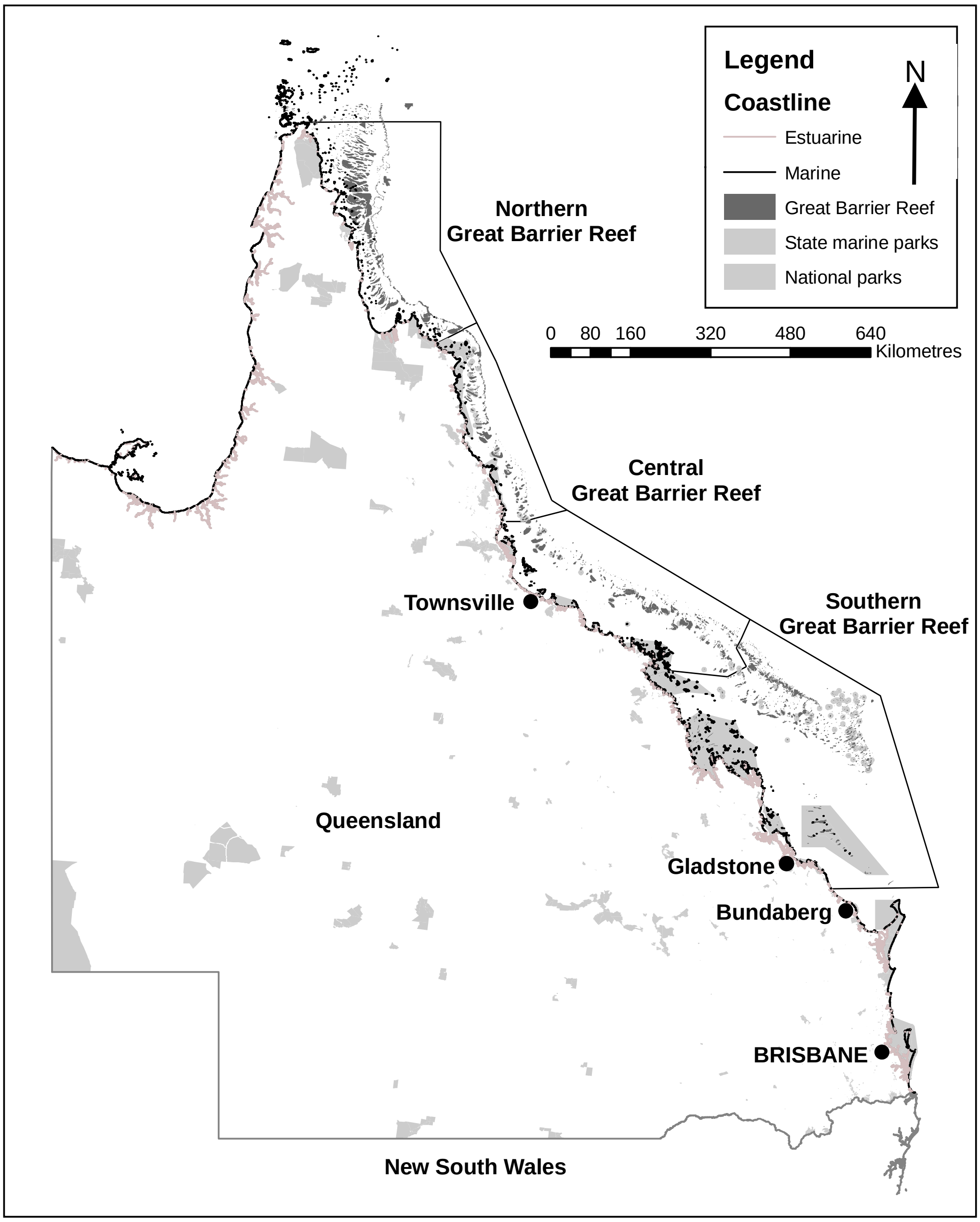


Figure 5: Queensland's shoreline classified and mapped as marine and estuarine systems.

Table 3: Percentage of the coastline mapped for Queensland and the percentage of each habitat type mapped for the marine and estuarine subsystem.

	Marine		**Estuarine**		
Shoreline habitat type	**Mainland** (%)	**Island** (%)	**Mainland** (%)	**Island** (%)	**Total** (%)
Rock ramp, wide	0.42	2.87	0.03	0.10	0.58
Rock platform, wide	2.74	6.18	0.15	0.73	1.69
Rock cliff	1.66	5.89	0.02	0.01	1.28
Rock ramp, narrow	1.85	9.54	0.03	0.10	1.94
Rock platform, narrow	0.05	-	-	0.03	0.01
Beach rock ramp, wide	-	-	-	-	<0.01
Beach rock platform, wide	0.37	0.09	0.01	-	0.08
Beach rock ramp, narrow	0.09	0.01	-	-	0.02
Beach rock platform, narrow	-	0.02	-	-	<0.01
Inclined boulder field, wide	0.54	1.44	-	-	0.34
Flat boulder field, wide	0.98	0.61	0.03	0.03	0.30
Boulder cliff	0.05	0.48	-	-	0.09
Inclined boulder field, narrow	4.02	6.54	0.01	0.07	1.82
Flat boulder field, narrow	-	0.01	-	-	<0.01
Inclined cobble beach, wide	0.33	1.35	-	-	0.28
Flat cobble beach, wide	0.59	0.65	0.02	0.01	0.22
Steep cobble beach	0.01	0.03	-	-	0.01
Inclined cobble beach, narrow	1.00	3.46	0.04	0.03	0.78
Flat cobble beach, narrow	-	-	-	-	<0.01
Inclined gravel beach, wide	0.17	0.74	-	-	0.15
Gravel flat, wide	0.44	0.99	0.01	0.20	0.28
Steep gravel beach	-	0.11	-	-	0.02
Inclined gravel beach, narrow	0.38	1.36	0.03	0.04	0.31
Inclined sand beach, wide	1.42	3.43	0.14	0.04	0.90
Sand flat, wide	45.87	22.16	5.58	4.19	15.30
Steep sand beach	-	0.15	1.04	0.14	0.58
Inclined sand beach, narrow	4.53	4.61	1.55	2.12	2.67
Sand flat, narrow	0.12	-	0.01	0.02	0.03
Inclined mixed fines flat, wide	1.01	1.07	18.89	10.04	11.48
Mixed fines flat, wide	28.62	9.54	47.70	43.23	37.35
Steep mixed fines flat	0.14	0.86	9.94	15.57	7.51
Inclined mixed fines flat, narrow	0.76	1.21	13.28	20.70	10.12
Mixed fines flat, narrow	-	-	0.35	0.94	0.32
Piles	0.36	0.03	0.12	0.31	0.18
Marina	0.07	0.04	0.42	0.09	0.25
Rock wall	0.40	0.10	0.57	1.24	0.56
Fringing coral reef, narrow	0.01	1.78	-	-	0.30
Fringing coral reef, wide	1.01	12.66	0.01	-	2.28

2.3.2 Shoreline habitat types in the IMCRA bioregions

There was little relationship between the fine-scale habitat classification described in this study and the broader scale IMCRA bioregions, which were determined through a combination of biological and physical data (IMCRA Technical Group 1998). There was no apparent relationship between the bioregions and the relative diversity of shoreline habitat types or the relative lengths of each shoreline habitat type recorded in each bioregion (Table 4).

The combination of continental islands and mainland coast resulted in a high diversity of shoreline habitat types associated with the Lucinda-Mackay bioregion. The presence of fringing reef as a shoreline habitat type was associated with bioregions in the Great Barrier Reef Region and Torres Strait. Fringing reefs were the dominant shoreline habitat types for Torres Strait (49.0%), the Ribbon Reefs (39.0%) and the Mackay-Capricorn (26.5%) bioregions (Table 5).

The Tweed-Moreton bioregion was dominated by vast stretches of wide flat beaches. The coastline in this area is typically exposed to higher wave action relative to the shoreline of the bioregions protected by the Great Barrier Reef. The across-shore component associated with the majority of beaches exposed to oceanic swell was sand dunes.

Artificial substrate was mapped for all coastal bioregions (Table 5). Less than one per cent[1] of the shoreline mapped was modified by rock walls, piles (jetties) and marinas. The highest level of modification has occurred in the Shoalwater Coast bioregion (0.25% of the mapped shoreline), which was associated with several major port developments. The length of shoreline modified is likely to be underestimated due to the absence of aerial photography in some key areas of the estuarine mainland sub-system (e.g. the mouth of the Brisbane River) where substantial modification has occurred.

[1] This does not include modifications of the coastline mapped for the across-shore components of the shoreline.

Table 4: Percentage of shoreline habitat types within each IMCRA bioregion based on the total length (8,316 kilometres) of Queensland's shoreline classified as marine system.

	IMCRA bioregion (%)											
Shoreline habitat type	**TMN**	**SCT**	**MCN**	**LMC**	**CRF**	**WTC**	**ECY**	**RBN**	**TST**	**WCY**	**KAN**	**WLY**
Rock ramp, wide	-	1.06	0.07	0.18	-	-	-	-	-	-	-	0.28
Rock platform, wide	0.21	2.01	-	0.29	-	0.04	0.09	-	0.03	0.16	-	1.55
Rock cliff	0.05	0.71	0.01	2.20	-	0.27	0.20	0.15	-	0.03	-	0.06
Rock ramp, narrow	0.08	1.75	0.93	1.76	-	0.51	0.33	0.02	0.03	0.01	-	0.13
Rock platform, narrow	0.03	-	-	-	-	-	-	-	-	-	-	-
Beach rock platform, wide	0.04	-	-	0.04	-	-	0.13	-	-	-	-	0.02
Beach rock ramp, narrow	-	-	-	0.03	-	-	0.02	-	-	-	-	-
Beach rock platform, narrow	0.01	-	-	-	-	-	-	-	-	-	-	-
Inclined boulder field, wide	-	0.37	0.06	0.45	-	-	0.05	-	-	-	-	0.05
Flat boulder field, wide	0.26	0.04	-	0.09	-	0.04	0.19	0.02	0.08	0.07	-	0.02
Boulder cliff	-	0.01	-	0.06	-	0.01	0.03	0.04	-	-	-	0.10
Inclined boulder field, narrow	0.02	0.35	0.08	1.71	-	1.03	1.36	0.09	0.23	0.28	-	0.08
Flat boulder field, narrow	0.01	-	-	-	-	-	-	-	-	-	-	-
Inclined cobble beach, wide	-	0.31	0.04	0.43	-	-	-	-	-	-	-	0.05
Flat cobble beach, wide	-	0.05	-	0.20	-	0.05	0.13	0.01	0.04	0.10	-	0.04
Steep cobble beach	-	-	-	0.01	-	-	-	-	-	-	-	-
Inclined cobble beach, narrow	-	0.07	0.02	1.42	-	0.36	0.11	-	0.10	0.08	-	0.02
Inclined gravel beach, wide	-	0.05	-	0.36	-	-	-	-	-	-	-	0.02
Gravel flat, wide	-	0.07	-	0.45	-	0.10	0.02	-	0.01	0.05	-	-
Steep gravel beach	-	-	-	0.01	-	-	-	-	-	-	-	0.04
Inclined gravel beach, narrow	-	0.01	-	0.64	-	0.13	0.01	-	0.01	0.04	-	-
Inclined sand beach, wide	0.03	0.37	0.09	0.35	-	0.01	0.10	-	-	-	0.63	0.81
Sand flat, wide	7.89	3.63	-	2.63	0.01	1.81	7.50	0.02	0.59	4.27	3.02	3.09
Steep sand beach	-	-	-	-	-	-	-	-	-	-	-	0.07
Inclined sand beach, narrow	0.08	0.10	0.03	1.01	-	1.31	0.62	0.06	0.27	0.44	0.04	0.59
Sand flat, narrow	0.01	-	-	-	-	0.02	0.03	-	-	-	-	-
Inclined mixed fines flat, wide	0.02	0.41	0.01	0.12	-	0.01	-	-	-	-	0.42	0.05
Mixed fines flat, wide	0.16	4.70	-	5.07	-	1.13	1.86	-	0.32	0.27	4.06	1.87
Steep mixed fines flat	-	0.38	-	0.08	-	-	0.03	-	-	-	-	-
Inclined mixed fines flat, narrow	0.02	0.28	-	0.29	-	0.10	0.10	-	0.06	0.10	0.01	0.03
Piles	0.01	0.16	-	0.02	-	-	-	-	-	-	-	-
Marina	-	-	-	0.04	-	0.01	-	-	-	-	-	-
Rock wall	0.04	0.09	-	0.10	-	-	-	-	0.01	0.01	-	-
Fringing coral reef, narrow	-	-	0.05	0.53	-	0.01	0.10	0.05	0.03	0.10	-	-
Fringing coral reef, wide	-	0.06	0.50	1.55	-	0.47	0.85	0.30	1.76	1.14	-	-
Percentage of the marine system	9.0	17.0	2.0	22.0	0.01	7.0	14.0	1.0	4.0	7.0	8.0	9.0

IMCRA bioregions: TMN – Tweed-Moreton; SCT – Shoalwater Coast; MCN – Mackay-Capricorn; LMC – Lucinda-Mackay Coast; CRF – Central Reef; WTC – Wet Tropic Coast; ECY – East Cape York; RBN – Ribbons; TST – Torres Strait; WCY – West Cape York; KAN – Karumba-Nassau; WLY – Wellesley (see Figure 3).

Table 5: Percentage of habitat type in each bioregion (based on Queensland's shoreline classified as marine system).

	IMCRA bioregion (%)											
Shoreline habitat type	**TMN**	**SCT**	**MCN**	**LMC**	**CRF**	**WTC**	**ECY**	**RBN**	**TST**	**WCY**	**KAN**	**WLY**
Rock ramp, wide	-	6.23	3.71	0.80	-	0.04	0.01	-	-	-	-	3.13
Rock platform, wide	2.37	11.77	-	1.31	-	0.54	0.68	-	0.97	2.27	-	17.29
Rock cliff	0.51	4.18	0.66	9.96	-	3.70	1.43	19.52	-	0.46	-	0.71
Rock ramp, narrow	0.86	10.24	49.04	7.97	-	6.82	2.39	3.18	0.77	0.15	-	1.48
Rock platform, narrow	0.29	-	-	-	-	-	-	-	-	-	-	-
Beach rock platform, wide	0.49	-	-	0.19	-	-	0.94	-	-	-	-	0.20
Beach rock ramp, narrow	-	-	-	0.12	-	-	0.14	-	-	-	-	-
Beach rock platform, narrow	0.08	-	-	-	-	-	-	-	-	-	-	-
Inclined boulder field, wide	-	2.17	3.14	2.03	-	-	0.34	-	-	-	-	0.50
Flat boulder field, wide	2.86	0.25	-	0.41	-	0.56	1.34	2.04	2.29	0.99	-	0.19
Boulder cliff	0.01	0.05	-	0.27	-	0.12	0.24	5.59	-	-	-	1.11
Inclined boulder field, narrow	0.26	2.08	4.24	7.73	-	13.93	9.78	11.35	6.48	3.96	-	0.85
Flat boulder field, narrow	0.06	-	-	-	-	-	-	-	-	-	-	-
Inclined cobble beach, wide	-	1.79	2.04	1.92	-	0.05	-	-	-	-	-	0.56
Flat cobble beach, wide	0.01	0.32	-	0.88	-	0.71	0.92	0.70	1.02	1.44	-	0.45
Steep cobble beach	-	0.02	-	0.04	-	0.04	0.01	-	-	-	-	-
Inclined cobble beach, narrow	-	0.42	1.03	6.44	-	4.79	0.80	-	2.93	1.09	-	0.26
Inclined gravel beach, wide	-	0.31	-	1.64	-	-	-	-	-	-	-	0.28
Gravel flat, wide	-	0.43	-	2.01	23.76	1.33	0.14	0.44	0.40	0.64	-	-
Steep gravel beach	-	-	-	0.03	-	0.03	-	-	-	-	-	0.46
Inclined gravel beach, narrow	-	0.08	0.09	2.91	13.03	1.71	0.07	-	0.17	0.58	-	0.03
Inclined sand beach, wide	0.32	2.14	4.89	1.57	-	0.10	0.72	-	0.09	-	7.67	9.05
Sand flat, wide	88.07	21.28	0.10	11.91	48.94	24.32	54.13	3.14	16.44	59.71	36.92	34.38
Steep sand beach	-	0.01	-	0.01	-	-	-	0.06	-	-	-	0.77
Inclined sand beach, narrow	0.94	0.61	1.50	4.55	14.23	17.69	4.48	8.49	7.59	6.11	0.53	6.57
Sand flat, narrow	0.10	-	-	-	-	0.26	0.25	-	-	-	-	-
Inclined mixed fines flat, wide	0.19	2.42	0.65	0.53	-	0.12	-	-	-	-	5.11	0.58
Mixed fines flat, wide	1.75	27.53	-	22.91	-	15.17	13.46	-	8.90	3.79	49.63	20.79
Steep mixed fines flat	-	2.23	-	0.36	-	-	0.19	-	-	-	0.02	-
Inclined mixed fines flat, narrow	0.19	1.63	-	1.30	-	1.32	0.70	-	1.74	1.35	0.12	0.35
Piles	0.13	0.95	-	0.11	-	0.06	0.01	-	-	0.01	-	-
Marina	-	-	-	0.20	-	0.18	-	-	-	-	-	-
Rock wall	0.50	0.53	-	0.46	-	0.04	0.01	-	0.17	0.13	-	0.01
Fringing coral reef, narrow	-	-	2.39	2.40	-	0.07	0.69	6.44	0.96	1.43	-	-
Fringing coral reef, wide	-	0.32	26.52	7.02	-	6.31	6.13	39.06	49.0	15.89	-	-
Shoreline habitat diversity types in each bioregion (excluding artificial shoreline)	18	24	14	28	4	23	24	12	15	15	7	22

IMCRA bioregions: TMN – Tweed-Moreton; SCT – Shoalwater Coast; MCN – Mackay-Capricorn; LMC – Lucinda-Mackay Coast; CRF – Central Reef; WTC – Wet Tropic Coast; ECY – East Cape York; RBN – Ribbons; TST – Torres Strait; WCY – West Cape York; KAN – Karumba-Nassau; WLY – Wellesley (see Figure 3).

2.3.3 Intertidal shoreline habitat protection

Approximately 7,786 kilometres (32.5%) of intertidal shoreline habitat in Queensland is protected in the current system of State marine parks. However, only a small fraction of this is protected at levels equivalent to IUCN Category Ia (Kelleher et al. 1995) (1.0%) or IUCN Category II (2.2%) (Table 6). The majority of the different types of habitat protected in zones equivalent to IUCN Category Ia are at levels of less than five per cent. Three shoreline habitat types are not protected under any sort of protection in State marine parks (Table 6).

Approximately 45% of the marine system (excluding artificial shoreline types) was protected in marine parks (Table 7). However, the highest level of protection (i.e. IUCN Category Ia) constituted less two precent of the mapped marine system. Similarly while a reasonable level of the mapped estuarine system was protected by marine parks less than one per cent of the mapped estuarine system was protected at levels equivalent to IUCN Category Ia. This evidence further suggests that there have been no systematic attempts to protect marine or estuarine intertidal habitats.

2.4 DISCUSSION

The primary goal of Australia’s national representative system of MPAs 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” (Australian and New Zealand Environment and Conservation Council 1999). In Queensland marine parks generally provide the means to protect marine biological diversity while allowing for a wide range of extractive and non-extractive uses. To date there has been no assessment of the effectiveness of the current system of marine parks in protecting marine biological diversity due to the lack of ecological information relating to large areas of the marine system.

The IMCRA bioregions were designed to provide the broad scale (100s to 1000s of kilometres) planning framework for designing a representative system of MPAs in Queensland and Australia (IMCRA Technical Group 1998). In Queensland more detailed information, relating to the distribution of habitats and species, is required for the coastline to assist with the design of a representative system of MPAs. Given the cost, time and resources required to survey biological distributions over large areas biodiversity surrogates are needed that support the identification of sites that contain high conservation values. Broad-scale classifications such as bioregions are not suitable for designing a system of MPAs because it is less likely to result in the protection of representative examples of marine biological diversity.

Table 6: Percentage of total shoreline habitat classified and mapped for each level of IUCN protection.

Shoreline habitat type	IUCN Ia (%)	IUCN II (%)	IUCN IV (%)	Total protected (%)
Rock ramp, wide	2.88	1.42	69.39	73.69
Rock platform, wide	1.96	1.86	36.47	40.28
Rock cliff	6.96	19.28	52.88	79.11
Rock ramp, narrow	3.71	6.15	74.85	84.72
Rock platform, narrow	0.00	0.00	0.00	0.00
Beach rock ramp, wide	0.00	0.00	100.00	100.00
Beach rock platform, wide	0.00	1.39	0.00	1.39
Beach rock ramp, narrow	0.00	9.58	0.00	9.58
Beach rock platform, narrow	0.00	0.00	0.00	0.00
Inclined boulder field, wide	2.41	2.37	83.88	88.65
Flat boulder field, wide	0.00	2.84	43.78	46.62
Boulder cliff	2.51	14.35	23.86	40.72
Inclined boulder field, narrow	4.32	10.79	44.65	59.76
Flat boulder field, narrow	0.00	0.00	0.00	0.00
Inclined cobble beach, wide	3.51	9.33	79.23	92.07
Flat cobble beach, wide	0.48	5.57	39.83	45.88
Steep cobble beach	0.00	0.00	77.55	77.55
Inclined cobble beach, narrow	3.10	24.59	50.17	77.85
Flat cobble beach, narrow	0.00	0.00	100.00	100.00
Inclined gravel beach, wide	1.67	32.88	58.53	93.09
Gravel flat, wide	4.10	11.84	66.27	82.21
Steep gravel beach	7.19	2.31	5.70	15.20
Inclined gravel beach, narrow	3.31	24.91	58.64	86.86
Inclined sand beach, wide	1.00	5.04	30.69	36.73
Sand flat, wide	0.33	2.21	22.63	25.16
Steep sand beach	0.00	0.11	5.78	5.89
Inclined sand beach, narrow	1.19	5.41	33.98	40.59
Sand flat, narrow	0.00	0.00	24.14	24.14
Inclined mixed fines flat, wide	0.00	0.12	9.10	9.22
Mixed fines flat, wide	0.91	0.28	24.89	26.08
Steep mixed fines flat	0.01	0.71	50.67	51.39
Inclined mixed fines flat, narrow	0.63	0.20	39.61	40.44
Mixed fines flat, narrow	0.00	0.00	15.75	15.75
Fringing coral reef, narrow	10.69	34.69	24.45	69.82
Fringing coral reef, wide	5.34	13.72	20.84	39.90
Total	1.02	2.18	29.28	32.48

Note: IUCN Category Ia - Protection, Scientific Research, Preservation, Marine National Park B Zones; **IUCN Category II** - Marine National Park A, National Park and Buffer Zones; **IUCN Category IV** - General Use, General Use A, Habitat Protection, Habitat, Conservation Park, Conservation and Estuarine Conservation, Conservation and Mineral Resource Zones.

Table 7: Length and percentage of Queensland's marine and estuarine shoreline habitat types protected in Marine Parks.

	Marine system				Estuarine system			
IUCN Category	**Mainland** (km)	**Island** (km)	**Total system** (km)	**Total protected** (%)	**Mainland** (km)	**Island** (km)	**Total system** (km)	**Total protected** (%)
Ia	7.4	172.7	180.1	2.2	7.6	56.8	64.5	0.4
II	87.3	401.7	489.0	5.9	20.5	13.2	33.7	0.2
III	-	-	-	-	-	-	-	-
IV	1273.7	1825.1	3098.8	37.5	2555.3	1365.5	3920.9	25.0
V	-	-	-	-	-	-	-	-
VI	-	-	-	-	-	-	-	-
Not protected	2908.8	1596.3	4505.1	54.5	9683.0	2001.3	11684.3	74.4
Total	4277.1	3995.9	8273.0		12266.4	3436.8	15703.4	

Note: IUCN Category Ia - Protection, Scientific Research, Preservation, Marine National Park B Zones; **IUCN Category II** - Marine National Park A, National Park and Buffer Zones; **IUCN Category IV** - General Use, General Use A, Habitat Protection, Habitat, Conservation Park, Conservation and Estuarine Conservation, Conservation and Mineral Resource Zones.

There was limited evidence that suggested there was a relationship between the IMCRA bioregions and the shoreline habitats described. This is most likely associated with the data used to derive the IMCRA bioregions, which only included two datasets that related to the intertidal shoreline, namely littoral crab diversity and mangrove and saltmarsh biogeography. The habitat types described increase the level of descriptive detail of coastal habitats than was provided by the IMCRA bioregions or any other study for the coastline of Queensland. Attempts to estimate the extent of coastline types around Australia have been reported as varying enormously (Fairweather & Quinn 1995). The development of finer habitat scale descriptions of the coastline will continue to provide increasing accuracy of the description of the coast. The value of such a description of the coast is that there is a consistent fine-scale habitat surrogate for each bioregion, which can be used to assist with determining MPA priorities at a statewide level.

There is increasing recognition of the need for criteria to identify and prioritise critical ecosystems, habitat or species for enhanced environmental protection (Daily 2000; Mysz et al. 2000). Methods to identify areas of high biological diversity for inclusion in a system of MPAs have been developing in response to inadequate broad-scale ecological information to support site planning and management. Coastal classification has been described as providing useful information to coastal management (Cooper & McLaughlin 1998). The classification of the intertidal habitats and the development of a GIS that allows easy manipulation of data enhances the potential for management agencies to integrate marine and adjacent terrestrial systems into decision-making. The management of data in a GIS provides a means to review the MPA priorities for the coastline from a statewide perspective but also would assist with management decisions at a local level.

2.4.1 Intertidal conservation

It has been recommended that intertidal biodiversity must be defined as ‘the variability in the numbers of species and their relative abundances within a habitat’ (Underwood & Chapman 1999b). The level of detail required to assess the number of species and their relative abundances is currently not available over vast stretches of the coastline. It has been acknowledged that appropriate methods to describe or understand patterns of biodiversity are required (Underwood & Chapman 1999b). To increase the likelihood that a system of MPAs conserves representative examples of intertidal biodiversity there is a need to increase our knowledge of the spatial and temporal variation of intertidal organisms over large stretches of the coast. Alternatively habitat surrogates could be used, although there is limited evidence currently available in the marine environment where the validity of such approaches support their use. The development of appropriate methods needs to consider the relationship between small scale (10s to 100s of metres) variation in the distribution of organisms and broad-scale (100s to 1000s of kilometres) approaches to designing a representative system of MPAs.

There are limited data on the types of habitats present along Queensland's intertidal shoreline to support coastal planning and management, which amongst other things is likely to be a result of the traditional view that the coastline represents a planning boundary between marine and terrestrial systems (Heyman & Kjerfve 1999). The results of this study represent the first detailed assessment of the protective status of the intertidal shoreline habitats of Queensland. The majority of shoreline habitat types protected in no-take zones are at levels of less than five per cent, which illustrates the absence of a systematic approach to the conservation of representative examples of shoreline habitats in Queensland.

Based on the description of the different types of habitats along the Queensland coast the existing system of MPAs fails to achieve adequate conservation of intertidal marine biodiversity. The focus of the existing system of MPAs is on habitats dominated by fine substrata, most likely a result of protecting large areas of mangroves along the coastline. There has been no systematic process of identifying the range of different habitats present along the coast to ensure the majority of habitats and hence species are protected in the existing system of MPAs. The identification of specific habitat types represents an approach that can be used to identify sites for inclusion in a representative system of intertidal MPAs increasing the likelihood that a greater number of species is protected.

At present there is not sufficient information to determine a target percentage of protection required for each shoreline habitat type to ensure adequate protection of intertidal biodiversity. This is particularly important given that the shoreline habitat classification is based on shoreline morphology, substrate type and oceanographic characteristics and not biological communities and species. The validity of using intertidal habitats described in this paper is the subject of ongoing research into the design of a system of intertidal MPAs that meets the goals of biodiversity conservation.

The differences in shoreline habitat type and morphology, climate and oceanographic processes along the coastline provide conditions suitable for a wide range of communities and species. There is however, relatively little broad-scale information available concerning the distribution of biological communities and species over such a range of environmental conditions, so it is difficult, if not impossible, to assess the extent of the variability in the composition of these communities along the coast. There have been only few studies that have attempted to describe the biological communities associated with the intertidal areas over a large section of the coast (see for example Endean et al. (1956a), Endean et al. (1956b)). The possible linkage between the specific types of habitat and the distribution of different biota requires detailed investigation. Without this information the value and application of habitat surrogates in the planning process cannot be evaluated.

2.5 CONCLUSION

Marine parks in Queensland suffer from the consequences of *ad hoc* reservation that is evident in other marine and terrestrial protected area systems (McNeill 1994; Pressey 1994). The current system of State marine parks, which have been declared over the last 15 years, fails to protect representative examples of intertidal habitats and is neither comprehensive nor adequate in protecting intertidal biodiversity. There is a need to determine conservation priorities based on finer scale surrogate measures in order to develop a representative system of MPAs. This would provide a sound basis for a systematic approach to site selection increasing the likelihood that conservation goals will be achieved.

CHAPTER 3 — INTERTIDAL HABITAT CONSERVATION: IDENTIFYING CONSERVATION TARGETS IN THE ABSENCE OF DETAILED BIOLOGICAL INFORMATION

3.1 INTRODUCTION

The designation of a system of marine reserves is seen as a mechanism for protecting representative examples of marine biodiversity in the coastal zone (Kelleher et al. 1995; Allison et al. 1998; Day & Roff 2000; Thompson et al. 2002). In Australia, and elsewhere in the world, there are commitments to the development of representative systems of marine reserves (Australian and New Zealand Environment and Conservation Council 1999; Day & Roff 2000), modelled on the principles of reserve design used for terrestrial systems that aim to maximise the protection of representative examples of ecosystems, habitats and species (Margules et al. 1988; Bedward et al. 1992; Davey 1998; Carr et al. 2003). Marine reserves are closed to all forms of extraction (e.g. fishing, mining, etc.) and may be a single reserve or be core areas within a larger multiple-use MPA (Agardy et al. 2003). A representative system of marine reserves would include the complete range of environmental gradients or habitat types at a given scale to maximise the protection of marine biodiversity (Kelleher et al. 1995; Australian and New Zealand Environment and Conservation Council 1999; Day & Roff 2000). The benefits of protecting representative examples of habitats using a systematic, science-based framework are expected to include the protection of the associated biological communities and species, a better understanding of marine systems through the establishment of long-term experimental and monitoring programmes, improved non-consumptive opportunities (e.g. enhanced educational opportunities) and potential fisheries benefits (e.g. increased abundance of overfished stocks) (Sobel 1993; Day & Roff 2000).

One of the most controversial issues associated with establishing marine reserves is determining where best to locate these areas (National Academy of Sciences 2001). This has often led to the opportunistic or *ad hoc* declaration of marine reserves, an approach considered expensive, inefficient and generally favouring only charismatic fauna (e.g. whales) or habitats that are at least risk because there is a lower demand for their use by extractive industries (Pressey 1994, 1997; Boersma & Parrish 1999; Pressey et al. 2000). The failure of such historical approaches to the establishment of marine reserves results in the inadequate protection of a truly representative range of habitats (e.g. intertidal areas in Queensland multiple-use marine parks; Banks & Skilleter 2002; Stewart et al. 2003). One approach to identifying marine reserves to protect representative examples of biodiversity involves the systematic collection of data on the distribution and ecology of the entire biodiversity within a region (Schoch & Dethier 1996; Rodriguez & Young 2000; Beck & Odaya 2001). However, in Australia, and elsewhere in the world, there is often a lack of detailed biological information on the distribution of species over large areas, even for intertidal habitats that have been traditionally well studied (Thompson et al. 1996; Ward et al. 1999; Roff & Taylor 2000; Underwood & Chapman 2001). In response to the lack of knowledge about the distribution of the biota, ecosystem-based 'coarse filter' approaches that use surrogate measures of biodiversity have been developed to support marine reserve identification (Ward et al. 1999; Beck & Odaya 2001; Ardron et al. 2002; Ardron 2003).

Surrogate measures of biodiversity include, for example, the use of biophysical properties, keystone species or indicator species (Banks & Skilleter 2002; Gladstone 2002; Warwick & Light 2002). Biophysical factors have most often been used to define and map habitat types (e.g. Ward et al. 1999; Zacharias & Roff 2000). These habitat types are mapped on the assumption that environments that have similar biophysical properties and environmental conditions predict, or at least correlate with, patterns of biological distributions (Araujo & Costa de Azevedo 2001; Stevens & Connolly 2004). The mapping of surrogates to define habitat types and delineate their boundaries requires the development of a consistent classification system, often based on enduring (e.g. abiotic) features of the marine environment (Roff & Taylor 2000; Zacharias & Roff 2000; Roff & Evans 2002). An ability to predict the distribution of biodiversity using surrogate measures would enable decisions about where to locate marine reserves to be made more reliably in the absence of detailed data on the distribution of species (Roberts et al. 2003b). There is increasing knowledge about the distribution of species in relation to species and habitats defined using physical properties (e.g. Williams & Bax 2001; Curley et al. 2002; Valesini et al. 2004), improving the likelihood that marine reserve systems designed on this basis would benefit the protection of representative biodiversity. Schoch & Dethier (1996) partitioned the coastline of San Juan Island (USA) into relatively distinct segments based generally on abiotic characteristics and were able to predict the composition of intertidal communities in the area. This approach has not been applied to other regions though, so there is no information on its general applicability. There is a clear need for further investigation of the relationship between physical properties of habitats and their ability to predict biological distributions (Stevens & Connolly 2004) for a wide range of species and habitat types. Banks and Skilleter (2002) described the intertidal habitats along 24,216 kilometres of Queensland's mainland and island coastline as a complex mosaic of intertidal habitat types, which included sandy beaches, rocky shores, mangrove communities, fringing coral reefs and coastal sand flats. The effectiveness of existing protection of multiple-use reserves was assessed and this demonstrated that the existing system of reserves failed to protect the full range of different intertidal habitats, with potential implications for biodiversity conservation if these different habitats support different communities or species (Schoch & Dethier 1996; Roff & Taylor 2000).

Several reserve design scenarios, are explored, which incorporate information about cost, reserve boundary length and existing protection of an intertidal habitat surrogate to identify the range of areas that would need to be included in a reserve system. Social, economic and management constraints, including associated costs (e.g. political or management-related costs) are important factors that will influence the choice of sites for inclusion into a system of marine reserves, particularly if large areas are required to protect representative examples of biodiversity. This study is an improvement over the previous attempts to apply a habitat surrogacy approach to reserve selection, as it is based on a fine-scale (10s to 100s of metres) intertidal habitat classification that has been applied consistently to 24,216 kilometres of the Queensland coastline. I describe a process for the systematic identification of sites for inclusion in a system of marine reserves that would protect representative examples of the full range of mainland intertidal habitats in Queensland. I evaluate the success of different reserve system scenarios in achieving conservation targets and the potential influences of reserve boundary compactness and the relative cost of each solution in identifying sites to be included in a representative system.

3.2 MATERIALS AND METHODOLOGY

The first detailed classification of Queensland’s intertidal shoreline was recently completed (Figure 5) (Banks & Skilleter 2002). The information in this classification was used to map the Queensland coast and provide the descriptive information for planning units that could be considered for inclusion in a system of ‘candidate’ intertidal marine reserves. The shoreline was subdivided into alongshore units that described the physical characteristics of the intertidal habitats at low tide. To consider the mosaic of intertidal habitats along a particular section of the shoreline, alongshore units were partitioned based on changes in the across-shore components, which reflected changes in substratum (e.g. bedrock, gravel), or features of the landscape landward (e.g. sand dunes, road, residential) of the low-tide habitat type.

The shoreline was classified into four major categories: marine (mainland and island) and estuarine (mainland and island). These major categories were then further partitioned into intertidal habitat types (e.g. wide sand flat; narrow rock ramp). A total of 63 intertidal habitat types was included in the initial reserve identification problem for the mainland (marine and estuarine) coast (i.e. 17,463 kilometres) of Queensland to identify priority areas that would contribute to representation of intertidal habitats in a system of ‘candidate’ marine reserves. There were three artificial habitats (i.e. piles, marina and rock wall) that were not considered part of the marine reserve identification problem because they are highly modified areas and are often associated with locations used for industrial or commercial purposes.

The across-shore component (habitat type) described by Banks and Skilleter (2002) for the littoral zone higher on the shore and adjacent to each intertidal habitat type was included as additional conservation features in the reserve identification problem. This enabled variation across the shoreline to be considered as part of the reserve system identification process, increasing the likelihood that the complete range of biodiversity in an area could be included in a marine reserve system solution. The number of conservation features recorded in the adjacent littoral zone was 30, taking the total number of conservation features to 93 for the mainland coast of Queensland.

3.2.1 Reserve selection algorithm

MARXAN (v1.8.3) was used to identify areas of coastline that are representative of the range of intertidal habitats in Queensland that could potentially be included in a representative system of marine reserves (Ball & Possingham 2000). One formulation of the reserve system identification problem is to minimise its cost, whilst ensuring that the specific level of representation for each conservation feature is met (Pressey et al. 1993; Leslie et al. 2003). Given reasonably uniform data on species, habitats and/or other relevant biodiversity features and surrogates for a number of planning units, MARXAN was designed to minimise the cost (a weighted sum of area and boundary length while meeting user-defined biodiversity targets) of reserve systems while meeting all targets (Possingham et al. 2000).

Simulated annealing was the optimisation method used to find appropriate solutions to the marine reserve identification problem. The simulated annealing algorithm in MARXAN identifies a range of potential solutions to the problem of representing all the habitat types to a predefined percentage, while minimising total cost (a weighted sum of area and boundary length). The summed irreplaceability of a site is the percentage of times each planning unit is chosen amongst the various solutions. Summed irreplaceability produces a value between zero and one for each planning unit. A unit that was allocated a value close to one is necessary for inclusion to meeting conservation goals, whereas a unit allocated a low value would be one that is unlikely to be required (Ball & Possingham 2000).

3.2.2 Scenarios explored

Scenarios 1–24 were explored for identifying possible state-wide priorities for the conservation of a representative example (i.e. minimum one) of the intertidal habitat types in Queensland (Table 8). The choice of these 24 scenarios was based on consideration of several features that were varied, including: (i) boundary length modifier (three levels), which puts more or less weight on the cost of the reserve boundaries (free ends) compared with reserve length; (ii) the number of conservation features (63 intertidal habitats; or 93 i.e. 63 intertidal habitats plus 30 adjacent littoral zone habitats); (iii) the conservation feature target (5, 10, 20% of each habitat type); and (iv) an occurrence target (minimum of one or three). I compared the 'best' of 100 runs for each scenario explored using planning units 10 kilometres in length.

3.2.2.1 Planning units

To determine state-wide conservation priorities, I grouped the mainland, intertidal habitats into 10 kilometre lengths (i.e. 1,746 planning units) to delineate the spatial location of potential ('candidate') sites to be included in a marine reserve system. Each planning unit consisted of one or more intertidal habitat types. The length of each habitat type in each planning unit was the basic data matrix for input into the reserve identification problem. Twenty-six planning units were locked into the marine reserve system as they were coastline areas adjacent to or within terrestrial national parks, or they were already protected in a 'no-take' zone within a state multiple-use marine park.

3.2.2.2 Conservation targets

Conservation targets were set at 5, 10, or 20% representation of each intertidal habitat for the mainland (marine and estuarine) coastline of Queensland (Table 9). Establishing conservation targets remains an area of reserve selection that leads to much controversy, particularly as there is currently limited empirical evidence to support the selection of one target over another. There has recently been support for establishing a conservation target of between 20 and 30% related to the reproductive attributes of some commercially exploited species (Bohnsack et al. 2002).

Table 8: Scenarios explored to identify state-wide conservation priorities for the conservation of intertidal habitats. The table includes details of the features used for each scenario to identify state-wide conservation priorities, including boundary length modifier, number of conservation features, pre-defined conservation targets and minimum occurrence of each conservation feature.

Scenario no.	No. of conservation features	Target (%)	Boundary Length Modifier	Replication	Separation distance
1	63	5	0	minimum 1	
2	63	5	0.5	minimum 1	
3	63	5	1	minimum 1	
4	63	5	0.5	3	
5	63	5	0.5	3	50 km
6	93	5	0	minimum 1	
7	93	5	0.5	minimum 1	
8	93	5	1	minimum 1	
9	63	10	0	minimum 1	
10	63	10	0.5	minimum 1	
11	63	10	1	minimum 1	
12	63	10	0.5	3	
13	63	10	0.5	3	50 km
14	93	10	0	minimum 1	
15	93	10	0.5	minimum 1	
16	93	10	1	minimum 1	
17	63	20	0	minimum 1	
18	63	20	0.5	minimum 1	
19	63	20	1	minimum 1	
20	63	20	0.5	3	
21	63	20	0.5	3	50 km
22	93	20	0	minimum 1	
23	93	20	0.5	minimum 1	
24	93	20	1	minimum 1	
25	63	5	0	Minimum cost assigned to each planning unit	
26	63	5	0.5		
27	63	5	1		
28	63	10	0		
29	63	10	0.5		
30	63	10	1		
31	63	20	0		
32	63	20	0.5		
33	63	20	1		

Table 9: 5%, 10% and 20% conservation targets (in kilometres) for each of the 63 intertidal habitat types.

Intertidal habitat type	Total length mapped	5% target	10% target	20% target
Estuarine				
Boulder cliff	0.04	<0.01	<0.01	0.01
Beach rock platform, wide	0.82	0.04	0.08	0.16
Flat boulder field, wide	4.02	0.20	0.40	0.80
Flat cobble beach, narrow	0.23	0.01	0.02	0.05
Flat cobble beach, wide	1.91	0.10	0.19	0.38
Fringing coral reef, wide	0.88	0.04	0.09	0.18
Gravel flat, wide	1.37	0.07	0.14	0.27
Inclined boulder field, narrow	1.80	0.09	0.18	0.36
Inclined boulder field, wide	0.45	0.02	0.04	0.09
Inclined cobble beach, narrow	5.67	0.28	0.57	1.13
Inclined cobble beach, wide	0.28	0.01	0.03	0.06
Inclined gravel beach, narrow	4.08	0.20	0.41	0.82
Inclined gravel beach, wide	0.08	<0.01	0.01	0.02
Inclined mixed fines flat, narrow	1657.18	82.86	165.72	331.44
Inclined mixed fines flat, wide	2373.30	118.67	237.33	474.66
Inclined sand beach, narrow	197.17	9.86	19.72	39.43
Inclined sand beach, wide	17.32	0.87	1.73	3.46
Marina	50.88	-	-	-
Mixed fines flat, narrow	44.42	2.22	4.44	8.88
Mixed fines flat, wide	5947.02	297.35	594.70	1189.40
Piles	20.15	-	-	-
Rock cliff	1.89	0.09	0.19	0.38
Rock platform, wide	19.67	0.98	1.97	3.93
Rock ramp, narrow	4.34	0.22	0.43	0.87
Rock ramp, wide	3.43	0.17	0.34	0.69
Rock wall	93.97	-	-	-
Sand flat, narrow	1.60	0.08	0.16	0.32
Sand flat, wide	730.76	36.54	73.08	146.15
Steep mixed fines flat	1266.91	63.35	126.69	253.38
Steep sand beach	132.68	6.63	13.27	26.54
Unclassified		-	-	-
Estuarine total	**13014.01**	**620.95**	**1241.93**	**2483.86**
Marine				
Boulder cliff	2.08	0.10	0.21	0.42
Beach rock platform, wide	16.79	0.84	1.68	3.36
Beach rock ramp, narrow	3.84	0.19	0.38	0.77
Beach rock ramp, wide	0.07	<0.01	0.01	0.01
Flat boulder field, wide	43.55	2.18	4.35	8.71
Flat cobble beach, wide	26.40	1.32	2.64	5.28
Fringing coral reef, narrow	0.48	0.02	0.05	0.10
Fringing coral reef, wide	45.47	2.27	4.55	9.09
Gravel flat, wide	19.50	0.97	1.95	3.90
Inclined boulder field, narrow	181.62	9.08	18.16	36.32
Inclined boulder field, wide	23.84	1.19	2.38	4.77
Inclined cobble beach, narrow	44.85	2.24	4.49	8.97
Inclined cobble beach, wide	14.72	0.74	1.47	2.94
Inclined gravel beach, narrow	16.46	0.82	1.65	3.29
Inclined gravel beach, wide	7.46	0.37	0.75	1.49
Inclined mixed fines flat, narrow	33.78	1.69	3.38	6.76
Inclined mixed fines flat, wide	45.20	2.26	4.52	9.04
Inclined sand beach, narrow	201.11	10.06	20.11	40.22

Table 9: continued

Intertidal habitat type	Total length mapped	5% target	10% target	20% target
Inclined sand beach, wide	63.74	3.19	6.37	12.75
Marina	3.10	-	-	-
Mixed fines flat, wide	1254.87	62.74	125.49	250.97
Piles	16.07	-	-	-
Rock cliff	73.30	3.66	7.33	14.66
Rock platform, narrow	2.05	0.10	0.21	0.41
Rock platform, wide	121.21	6.06	12.12	24.24
Rock ramp, narrow	82.43	4.12	8.24	16.49
Rock ramp, wide	18.29	0.91	1.83	3.66
Rock wall	16.85	-	-	-
Sand flat, narrow	5.44	0.27	0.54	1.09
Sand flat, wide	2058.03	102.90	205.80	411.61
Steep cobble beach	0.33	0.02	0.03	0.07
Steep mixed fines flat	6.33	0.32	0.63	1.27
Steep sand beach	0.18	0.01	0.02	0.04
Marine total	**4449.44**	**220.64**	**441.34**	**882.70**
GRAND TOTAL	**17463.45**	**841.59**	**1683.27**	**3366.67**

The different scenarios also explored a target for the inclusion of a minimum of three examples of each conservation feature to protect against the loss of associated biodiversity arising from any major catastrophes that may affect an area (i.e. oil spills or adjacent land-use development). Buffering a reserve system against such catastrophes involves spreading the risk along the coast or determining an appropriate ‘insurance factor’ (Allison et al. 2003). By building into the reserve-design criteria a minimum separation distance, impacts of a catastrophic event on a reserve locally would not destroy the integrity of the entire reserve system (Ball & Possingham 2000). The inclusion of these areas was based on a separation distance of 50 kilometres between three or more conservation features protected.

3.2.2.3 Boundary length modifier

The boundary length modifier was varied to determine the relative importance of system compactness. The algorithm ignores the boundary length of planning units when the boundary length modifier is set at zero and the compactness becomes increasingly important as the boundary length modifier is increased. To determine the influence of the boundary length modifier on the identification of sites for state-wide conservation priorities, boundary length modifiers of 0, 0.5 and 1 were used.

3.2.2.4 Cost

In this exercise, the objective was to minimise the total cost of the system in terms of total length, while ensuring that at least 5, 10 or 20% of every one of the conservation features (i.e. intertidal habitat types) were represented across the entire marine reserve system. The overall cost of the system is a combination of purchase (or compensation), dedication and ongoing management. Although direct measures of cost are hard to obtain, area and boundary length are useful surrogates and were used in this analysis. A relative cost was measured in terms of overall conservation rather than a real cost associated with the exclusion of commercial extraction. The length of each across-shore component of a planning unit that contained road, industry or residential areas or other artificial feature determined the relative cost and all other planning units had a relative cost of zero.

In all scenarios I assume that sites adjacent to residential areas are likely to have greater social or economic cost associated with the planning and management of a multiple-use marine park. Developing a zoning plan would normally involve the closure of certain areas to all forms of extraction (e.g. recreational and commercial fishing). I assume that the process of removing or ceasing exploitation may have a higher cost (i.e. enforcement and compliance or political costs) associated with the implementation of a management regime in areas adjacent to residential or tourist development where use may be higher.

Nine additional scenarios (scenarios 25–33) were explored to determine the influence of relative cost on the intertidal marine reserve system solutions (Table 8). These scenarios were completed with relative cost determined as described above; however, planning units adjacent to terrestrial protected areas were given a zero cost, with a minimal cost provided to all others.

3.3 RESULTS

The length of coast identified to be included in a ‘candidate’ system of intertidal marine reserves ranged from 3,476 kilometres (20% of the mainland coastline) to 7,500 kilometres (43% of the mainland coastline) for the 24 scenarios analysed (Table 8) where relative cost was only included for those planning units that contained modified across-shore components (e.g. roads, residential areas) (Table 10).

Table 10: Results of scenarios explored to identify state-wide conservation priorities for the conservation of intertidal habitats (see table 8 for scenario details).

Scenario no.	Conservation feature targets met	Length of coast protected (km)	Number of planning units
1	Yes	7126.7	713
2	Yes	3476.4	348
3	Yes	5785.6	579
4	33 met	6045.8	605
5	33 met	6375.2	638
6	Yes	7464.8	746
7	Yes	6798.6	680
8	Yes	5878.7	588
9	Yes	6895.2	689
10	Yes	6197.6	620
11	Yes	6885.2	689
12	32 met	5104.1	511
13	33 met	5455.4	546
14	Yes	7216.5	723
15	Yes	7574.9	758
16	Yes	5505.5	551
17	Yes	7226.9	723
18	Yes	6817.3	682
19	Yes	5428.5	543
20	35 met	6795.2	680
21	33 met	6702.9	671
22	62 met	7529.5	752
23	Yes	7204.7	721
24	Yes	6914.9	692
25	Yes	920	92
26	Yes	940	94
27	Yes	910	91
28	Yes	1760	176
29	Yes	1760	176
30	Yes	1760	176
31	Yes	3430	343
32	Yes	3430	343
33	Yes	3440	344

3.3.1 Habitat representation and conservation targets

The full range of intertidal habitats described for the coastline of Queensland were included in the marine reserve system solutions (for scenarios 1–24). A larger reserve system was required to achieve conservation feature targets as these targets increased from five per cent (Figure 6), to 10% (Figure 7) and 20% (Figure 8) of each habitat type. The average proportion of mainland shoreline required to be included in a system of intertidal marine reserves to achieve the conservation feature targets was 35% (for 5% target), 37% (for 10% target) and 38% (for 20% target) of the total shoreline analysed.

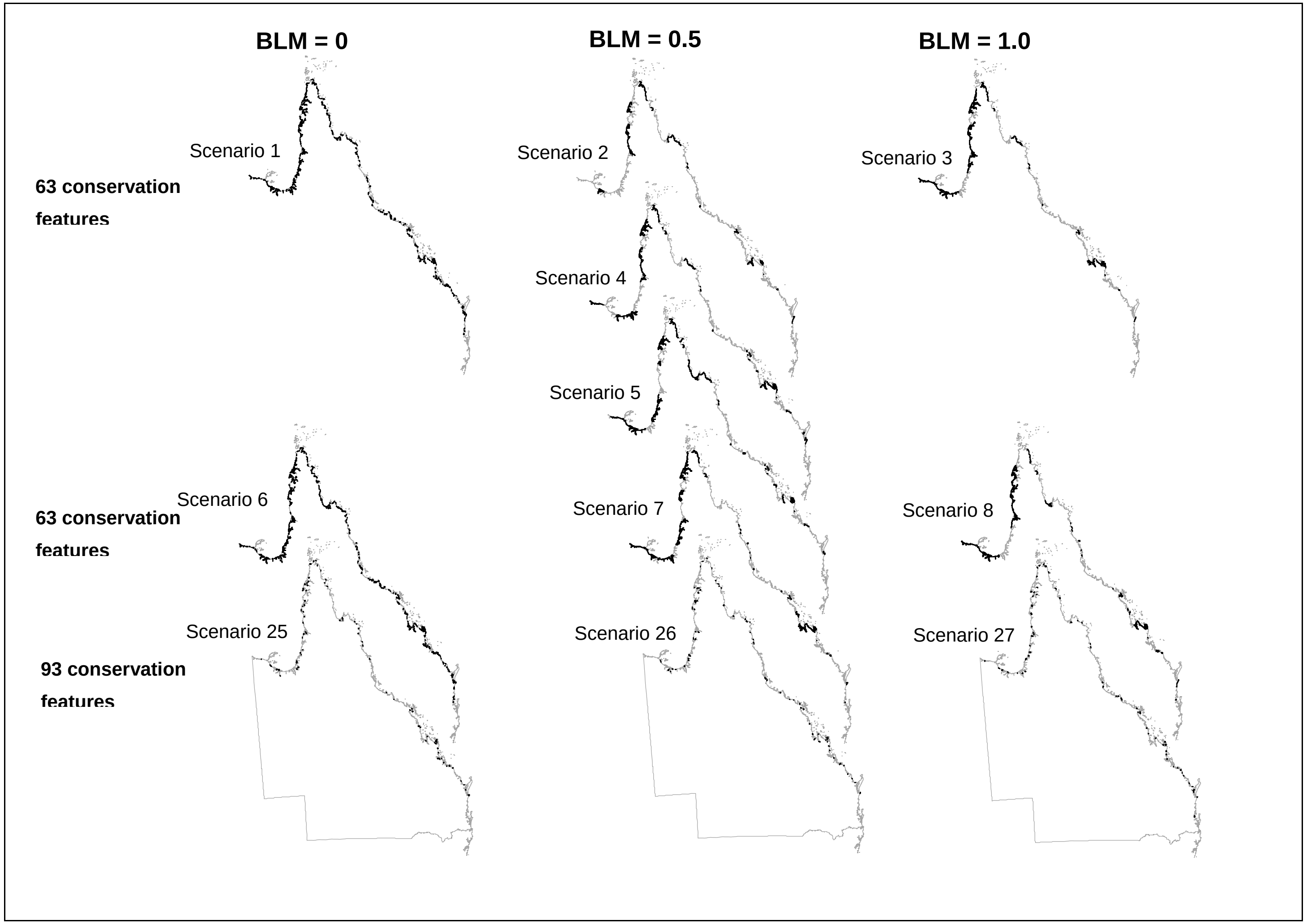


Figure 6: Scenarios that used five per cent conservation feature targets (see table 8 for scenario details).

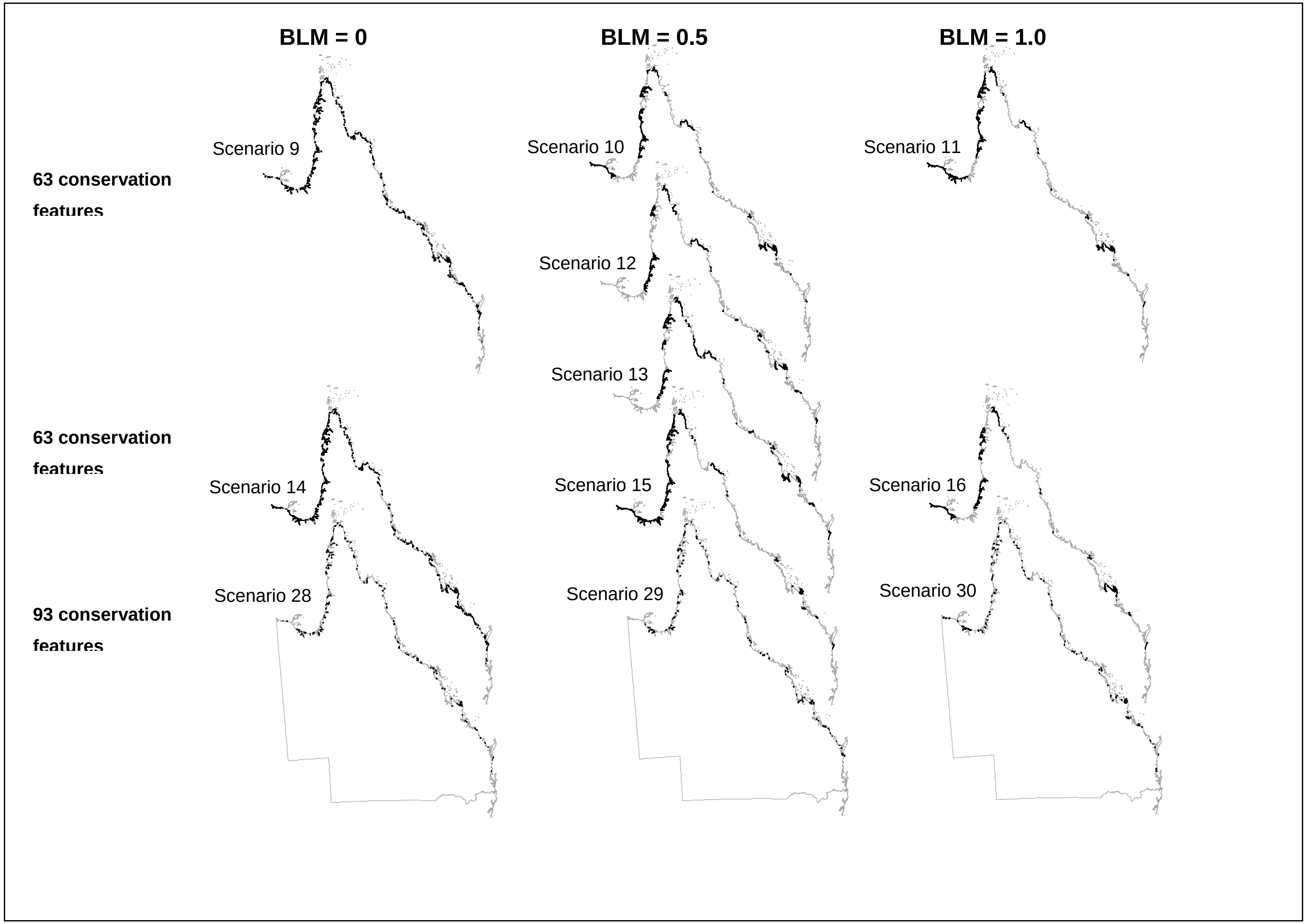


Figure 7: Scenarios that used 10% conservation feature targets (see table 8 for scenario details).

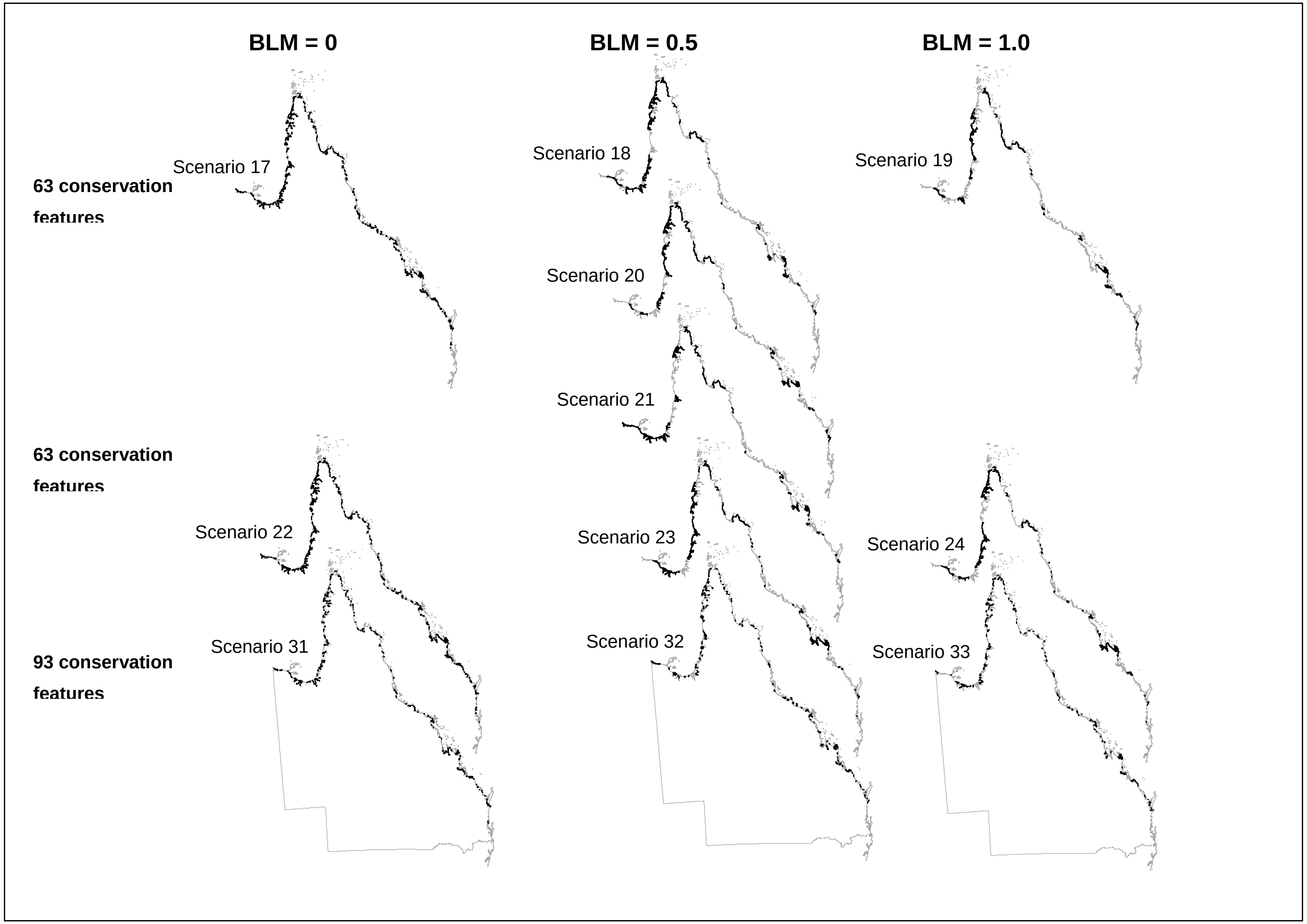


Figure 8: Scenarios that used 20% conservation feature targets (see table 8 for scenario details).

Similar lengths of coast were identified for inclusion in a system of intertidal marine reserves whether there were 63 conservation features or 93 conservation features included in the reserve identification problem (i.e. when the adjacent upper shore habitats were included). Increasing the number of conservation features by incorporating this vertical dimension in a shoreline should increase the likelihood that representative examples of intertidal communities and species will be conserved by protecting a mosaic of intertidal habitat types in the reserve system.

The conservation-feature targets were generally exceeded for each intertidal habitat type in all the solutions. For a target of five per cent, the proportion of all habitat types selected to be included in a system of marine reserves was >10% of the available habitat. Similarly, >50% (for a 10% target) and >30% (for a 20% target) of each habitat type was selected to be included in the reserve solutions.

In solutions where the targets were not achieved, this outcome generally related to scenarios where the requirement of at least three examples of each conservation feature could not be met.

3.3.2 Boundary length modifier

As the boundary length modifier increased above zero the compactness of the ‘candidate’ marine reserve systems increases (Figures 6 – 8). Increasing the boundary length modifier from zero to one resulted in a reduction in the overall length of shoreline selected to be included in the reserve solution. Based on targets of five per cent, 10% and 20% and 63 conservation features, the reduction in shoreline required to achieve these targets was 19%, 0.1% and 24% respectively when the boundary length modifier was increased from zero to one. Similarly, when 93 conservation features were included in the analysis, the overall length of shoreline required for protection reduced by 21%, 24% and 8% respectively as the boundary length modifier increased.

3.3.3 Cost

Taking relative cost into account resulted in the preferential inclusion of areas that were adjacent to areas that contained natural features in the adjacent across-shore components rather than modified sections of the coastline. Areas close to roads or residential areas attracted a higher relative cost and were, therefore, less likely to be included in a reserve solution. Additional planning units were included in the system, as they were considered to have no relative cost and, therefore, were free to be selected.

For scenarios where a minimum cost was given to all planning units (except those adjacent to terrestrial reserves) the reservation targets were achieved. The average proportion of the mainland shoreline required to be included in the system of intertidal marine reserves to achieve conservation targets was five per cent (for a 5% target; Figure 6), 10% (for a 10% target; Figure 7) and 20% (for a 20% target; Figure 8).

3.4 DISCUSSION

The design of a system of intertidal marine reserves that meet pre-specified goals can be made more efficient through the use of siting algorithms and a consistent fine-scale classification of intertidal habitats. This approach enables marine reserve managers to negotiate a conservation outcome that is more likely to protect the range of intertidal assemblages and species than the historical *ad hoc* approach. The ability to 'factor in' other features, important to marine reserve selection (i.e. cost and boundary length), into the decision-making process for such a large section of coast represents an important step forward in marine reserve identification and selection. To demonstrate the influence of relative boundary length (i.e. compactness), relative cost and variations in the conservation feature targets would greatly assist the negotiation process with decision makers and stakeholders (Badalamenti et al. 2000; Scholz et al. 2004). These factors will also provide a sound basis to ensure that stakeholders, marine reserve managers and politicians understand the influence their decisions may have in achieving conservation goals while minimising costs (management or political) associated with implementing different reserve system solutions.

3.4.1 Conservation feature targets

Determining the area or amount of each habitat to be included in a system of marine reserves remains a challenging problem (Bohnsack 1998; Pressey et al. 2003). Protecting a minimum of 10–50% of the total area of all representative habitats has been recommended based on cultural traditions, social acceptability and the precautionary principle (Ballantine 1991; Bohnsack 1993; Dayton et al. 1995; Bohnsack et al. 2002; Airame et al. 2003; Roberts et al. 2003b). Targets of 20–30% of total area have been recommended in relation to fisheries management and maintaining fish stocks (Bohnsack 1998). There has been a trend for targets of between 10–30% to be used in reserve-system planning. For example, targets of 20% of reef and non-reef bioregions were used in the rezoning of the Great Barrier Reef Marine Park (Great Barrier Reef Marine Park Authority 2003), and targets of 10–30% of the area of 26 habitats were used as one of a number of scenarios to design a system of marine reserves in the Florida Keys (Leslie et al. 2003). Studies have shown that there are many different combinations of reserve systems that could meet conservation representation targets (Possingham et al. 2000; Leslie et al. 2003; Stewart et al. 2003).

Although there is limited biological/ecological evidence to support the establishment of a particular target for representation of ecosystems, habitats or species in a reserve system, examination of a range of conservation feature targets should be used to assist in a negotiation process with stakeholders. These different scenarios could include varying targets for overall representation (of all habitat types) or for specific habitat types (e.g. rare or unique habitats). A range of reserve solutions using different scenarios provides a sound basis for understanding the implications of conservation outcomes with consideration of cost, planning unit length and reserve system compactness.

3.4.2 Reserve compactness and cost

There are social and political implications for a system of marine reserves where there is an absence of information related to the cost of implementing or managing the system. It is important that an appropriate assessment of cost for each planning unit be made prior to the identification and planning process commencing. The solutions presented in this paper demonstrate the importance of defining cost for each planning unit, which, when not considered, resulted in a much higher average length of mainland coastline required to achieve a conservation feature target of five per cent (approximately 6,000 kilometres or 35% of the mainland coast). A greater proportion of the planning units, compared with those scenarios when all planning units had a minimum cost, were considered cost free and, therefore, were potentially available for selection in a reserve system solution, resulting in the selection of a larger number of planning units than was necessary to achieve reserve system targets. Closing some areas to harvesting would potentially draw attention from recreational and industrial fishers who generally oppose such closures, particularly if it involves extensive areas of the coastline, thus resulting in higher social and/or political costs (Scholz et al. 2004). Therefore, it is imperative that realistic costs are used for each planning unit.

Some areas of coastline, including areas in the southern Gulf of Carpentaria, Shoalwater Bay, Princess Charlotte Bay, east and west Cape York and Double Island Point in south east Queensland, were consistently identified as a 'key' part of the candidate system of marine reserves, but this may have been at the cost of other areas because of the influence of 'relative cost'. The identification of these areas, many of which are in northern Queensland, were influenced by the relative cost assigned to planning units that contained modified intertidal habitat types or that were adjacent to residential or industrial areas, such as those in south east Queensland. The cost of including the latter in a representative marine reserve scheme is likely to be greater. Greater relative costs associated with some otherwise similar planning units are likely to have led to the selection of planning units that, for example, contain wide sand flats (i.e. beaches) in the Gulf of Carpentaria, where there is little modification of the shoreline and adjacent terrestrial habitats, rather than similar habitats in south east Queensland.

The data used by the siting algorithm did not incorporate latitudinal variation when assigning a planning unit to be included in the reserve system. To overcome this problem, analysis of the mainland coastline could be examined in relation to the IMCRA marine bioregions (IMCRA Technical Group 1998), with the objective of representative coverage of all biogeographic regions in reserves (Roberts et al. 2003a, 2003b). This would provide a basis for representation of intertidal habitat types in a latitudinal context and would assist with differentiating intertidal habitats between climatic and oceanographically different areas along the coast of Queensland.

The ability to consider compactness of reserve system solutions and costs associated with either the establishment or management of a system of marine reserves strengthens the ability of decision makers to make an informed decision about where to locate the reserves and provides a sound basis for negotiation with stakeholders. Compactness of the reserve system is important in relation to overall social and political acceptability of a system of protected areas, because people affected by closures to harvesting activities are generally likely to seek minimal protection over small areas and are, therefore, more likely to support a highly compact system.

3.4.3 Planning unit size

It is likely that the reserve system outcomes shown here have been strongly influenced by the length of the planning unit (i.e. 10 kilometres). To achieve the conservation targets, planning units were selected and/or substituted by MARXAN to determine the ‘best’ reserve system solution based on 100 runs for a scenario. Where, for example, a particular type of unique or rare habitat (e.g. steep cobble beach) occurs as a unit less than the 10 kilometres length, it will be selected, along with the surrounding, more extensive habitat type (e.g. wide sand flats). Although this ensures the rare habitat type is included in the reserve system, it may lead to over-representation (i.e. above conservation targets) of these other habitats. Thus, planning units that contain habitats considered rare or unique should probably be ‘locked into’ the system during the stakeholder negotiations. Increased representation of habitats that may be desirable from a biodiversity conservation perspective, however, is likely to lead to greater social or political costs in establishing the system. Use of a smaller length of planning unit (e.g. one kilometre) would reduce the potential to over-represent common or extensive habitat types.

3.4.4 Using habitat surrogates in reserve design

The application of reserve solutions derived through the use of decision-support tools and based on a physical habitat surrogate can assist the decision-making process. It has been suggested that the best strategy for conserving intertidal habitats is to conserve patches of habitats of different sizes and shapes and at different distances apart (Underwood & Chapman 2001). The systematic classification of the coast of Queensland provided a consistent description of the physical intertidal habitat types (Banks & Skilleter 2002). The problem with such physical classifications of the coastline is that it is not yet possible to predict reliably the biological communities that are associated with each type of habitat (Zacharias et al. 1999; but see Valesini et al. (2003) for an alternative).

At present, it is assumed that the combination of physical factors used to derive the habitats adequately predicts differences in the diversity and abundance of intertidal organisms between one habitat type and another. This is probably the case when considering differences in the assemblages supported by extremes in habitat types (Valesini et al. 2003), such as sandy beaches, rocky shores or biologically derived reefs. However, it is more problematic when considering the variation within broad habitat types (e.g. different types of rocky shore, such as wide rock platforms, narrow rock platforms and rock ramps). There are no detailed empirical data that allow predictions to be made about the relationship between physical attributes of a specific habitat type and the biota found there. If there is sufficient replication within the reserve system then this may not matter; however, there is now the need to test the relationship between the habitat surrogate and the distribution and abundance of intertidal organisms.

Clearly, in the absence of detailed information related to the distribution and abundance of intertidal organisms, conservation priorities must be identified using physical surrogates as the input data for a siting algorithm. The strength of this pragmatic approach is that it can be applied to large sections of the intertidal coastline. It will also assist with negotiation processes with stakeholders who will be better able to understand the implications of selecting one site over another for inclusion in a system of marine reserves.

3.4.5 Marine reserve planning in Queensland

The results represent a small subset of those solutions available to protect a representative range of intertidal habitats in Queensland. Use of 10 kilometre planning units enabled conservation priorities to be broadly identified over a vast length (i.e. 17,463 kilometres) of the Queensland coast. Although the outcomes of the present analysis provided a range of solutions for the identification of a set of priority areas that could be protected in intertidal marine reserves, the marine conservation framework used in Queensland involves the establishment of large multiple-use marine parks. These parks usually have ‘core’ areas (i.e. zones) that are closed to all extractive activities. As discussed by Agardy et al. (2003), there is debate about the value of establishing large multiple-use marine parks that contain ‘core’ areas as no-take zones versus the establishment of a marine reserve over a specific area that is entirely closed to extraction.

This analysis identified a representative suite of areas of conservation interest in Queensland. From here, marine-park planners and decision makers need to consider a range of other factors at a variety of spatial scales in the context of social and economic impacts of the different options to select the location of reserves (i.e. areas zoned to prohibit extractive activities) within multiple-use marine parks. Following the declaration of a marine park, a zoning plan is required that establishes the spatial management regime that defines marine reserves (areas of ‘no-take’) to areas where extractive activities can continue. During this process, finer scale (e.g. one kilometre) planning units should be used to determine local priorities for zoning areas as ‘no-take’. The identification of these ‘no-take’ zones using a systematic science-based framework will be fundamental to achieving conservation goals by protecting representative examples of intertidal habitats in fully protected ‘no-take’ zones.

The findings support the need for Queensland to review zoning arrangements for the current system of multiple-use State marine parks (Banks & Skilleter 2002). In particular, we need a system of intertidal reserves or ‘no-take’ zones that include a representative range of intertidal habitats to provide protection from damaging activities. There is also a need for consideration of the establishment of additional multiple-use marine parks in bioregions where no marine park protection currently exists (e.g. Torres Strait and the bioregions in the Gulf of Carpentaria).

The differences between the physical and biological processes operating in the marine and terrestrial environments have been documented and recognised as important for determining the planning response by management agencies (Avery 2003). This, however, does not necessarily mean that reserve systems to protect representative examples of the range of ecosystems, habitats and species in these environments need to develop independently. Avery (2003) describes the planning approaches to marine and terrestrial reserves systems as being largely the same; this enables the systems to be developed to complement each other, reducing the risk that important habitats in the coastal zone are not protected. In Queensland, there has been little progress in the protection of intertidal habitats in ‘no-take’ zones in marine parks. The future design and planning of protected area systems should ensure that marine and terrestrial system identification and selection is integrated to optimise the likelihood that the range of intertidal habitats is protected. If marine reserve systems continue to be developed without consideration of the adjacent terrestrial reserve system (and the role that system may play in protecting intertidal habitats) then there will continue to be poor representation of intertidal habitats in reserves.

CHAPTER 4 — THE IMPORTANCE OF INCORPORATING FINE-SCALE HABITAT DATA INTO THE DESIGN OF AN INTERTIDAL RESERVE SYSTEM

4.1 INTRODUCTION

Conservation programmes, including the establishment of marine reserve systems, are preferably developed based on detailed knowledge of each species' distribution, abundance, life history and their interactions with other species and the biophysical environment in which they exist (Brooks et al. 2004). There are, however, serious gaps in the knowledge of the distribution and abundance of species, especially in the marine environment (Thompson et al. 2002; Gladstone & Alexander 2005). As a result there is an urgent need to develop and test alternative approaches such as the use of biodiversity surrogates to support ongoing management and conservation programmes (Ward et al. 1999; Brooks et al. 2004; Cowling et al. 2004; Pressey 2004).

Surrogate approaches include, for example, the use of habitats (Ward et al. 1999; Banks and Skilleter 2002), species (e.g. individual or focal), species assemblages or higher taxonomic levels (Ward et al. 1999; Gladstone 2002; Hitt & Frissell 2004; Gladstone & Alexander 2005; King & Beazley 2005; Smith 2005), or environmental diversity (Araujo et al. 2001). The use of habitat surrogates in marine reserve design programmes generally assumes that protection of particular habitat types will lead to the protection of a larger suite of species whose conservation needs, distribution and abundance remains unknown (Banks & Skilleter 2002; Pressey 2004; Stevens & Connolly 2004).

Regional conservation assessments have generally used ecosystem-based 'coarse filter' approaches that include broad-scale (10s to 100s of kilometres) surrogate measures of biotic diversity to support marine reserve establishment (Ward et al. 1999; Beck & Odaya 2001; Ardron et al. 2002; Ardron 2003). These coarse filter approaches typically classify intertidal systems into broad categories (i.e. rocky shore, sandy beach). Such approaches do not consider finer-scale (10s to 100s of metres) variation in the composition of intertidal assemblages within and among intertidal habitats or microhabitats at smaller scales. Within each intertidal category there are often different types of habitats or microhabitats, and it is often at this scale that there is considerable variation in the patterns of biota in different communities.

For example, biota on rocky shores is often greater in boulder fields (McGuinness 1986, 1987a; Underwood & Chapman 2001; Chapman 2002; Le Hir & Hily 2005) and rock pools (Bennett & Griffiths 1984; Huggett & Griffiths 1986; Prochazka & Griffiths 1992; Mahon & Mahon 1994) than on surrounding rock platforms, yet typically these microhabitats are not specifically included in classification schemes used to identify reserves (Ardron et al. 2002; Banks & Skilleter 2002; Ardron 2003; Breen et al. 2003). Some species, including rare ones, may only be associated with specific habitats or microhabitats such as boulder fields (Chapman 2002) or rock pools (Bennett & Griffiths 1984; Huggett & Griffiths 1986) so unless the occurrence of these intertidal features is included in the selection process there is a risk that the associated biota will not be conserved.

The composition of intertidal assemblages is the result of complex interacting biological and physical processes operating at a variety of spatial and temporal scales (Underwood 1994; Underwood & Chapman 2001; Thompson et al. 2002). Physical and biological processes considered to influence the structure of intertidal assemblages include wave exposure (Menge 1976; McQuaid & Branch 1984, 1985a, 1985b; Hartnoll & Hawkins 1985; Bustamante & Branch 1996), substratum type (Menge et al. 1985; Cruz Motta et al. 2003), slope and vertical and horizontal surfaces (Garrity 1984), rock pools (Garrity 1984), cracks and crevices (Seapy & Littler 1978; Garrity 1984; Menge et al. 1985; McGuinness & Underwood 1986; Menconi et al. 1999), presence of biologically derived habitat (Seed 1996; Thompson et al. 1996; Monteiro et al. 2002), topographic complexity (Underwood & Chapman 1989) and species interactions (predation, grazing, competition) (Underwood et al. 1983; Rivadeneira et al. 2002). Many of these factors and processes operate at the scale of habitats or microhabitats within rocky shores (McGuinness & Underwood 1986; Beck 2000; Bell 2005). Despite detailed information being available on the factors structuring intertidal assemblages and the contribution rocky intertidal research has made to general ecological theory (Underwood 2000), this understanding of the linkages between biota and habitats or microhabitats has generally not been incorporated into the design of marine reserves.

Banks and Skilleter (2002) used an intertidal classification to subdivide the coastline of Queensland into alongshore (i.e. lineal) units that described the physical characteristics of all intertidal habitats at low tide. These shoreline types provided a finer-scale (10s to 100s of metres) description of the coastline than the coarse filter approaches (used by Ward et al. 1999; Beck & Odaya 2001; Ardron et al. 2002; Ardron 2003), which are only likely to predict reliably intertidal assemblages at extremes in habitat such as sandy beach, rocky shore or biologically derived habitat. There have been few attempts to include information on the variation within these shoreline types, for example, different types of rocky shore, such as wide rock platforms, narrow rock platforms, rock ramps and rock cliffs in marine reserve selection (but see Banks et al. 2005). In the absence of information on the distribution and abundance of intertidal species, it may not be valid to assume that the different types of rocky shore are similar in the assemblages and communities they support. Indeed, available information suggests that the presence/absence and/or coverage of habitats and microhabitats such as rock pools, boulder fields and sandy patches could have a marked effect on the patterns of biodiversity on a rocky shore (see references above). This information was not used in the analysis of shoreline types done by Banks et al. (2005). Different types of rocky shore were included in different reserve systems on the basis that each shoreline type needed to be represented. However, variation in the types and extent of habitats or microhabitats within those shoreline types was not incorporated. As a result there is the potential for biota not to be protected if the distribution and abundance of these habitats or microhabitats is not uniform across the different types of rocky shores.

There are insufficient data on the distribution and abundance of intertidal species in Queensland to determine whether broad shoreline types are reliable surrogate measures for different levels (habitat, microhabitat, species) of biodiversity. Future marine reserve planning based only on this information may be inadequate to protect all levels of biodiversity. For example, if a reserve system based only on shoreline types happened to select rocky shores that do not include examples of rock pools and other important microhabitats (because these criteria were not included in any analysis), then the associated biota would not be protected. Similarly the patterns of microhabitats and habitats on rocky shores may also vary between, and within different types of rocky shore, further increasing the risk that features influencing the distribution and abundance of biota may be over- or under-represented in a reserve system.

The aims of this research were to examine whether the presence/absence of microhabitats, or the distribution and areal extent of different intertidal habitats varied between and within rocky shores in south east Queensland, and how inclusion of this information affected marine reserve selection. I approached this in two ways: 1) by mapping the spatial extent of the different habitats present on rocky shores; and 2) by recording the presence/absence of microhabitats on rocky shores. I then examined different scenarios for a marine reserve system where additional information on the spatial extent of habitats or the presence/absence of microhabitats was not included. These scenarios were compared with scenarios when information on these features was included to identify how the inclusion of habitats or microhabitats will influence reserve system solutions.

4.2 MATERIALS AND METHODOLOGY

4.2.1 Study area

This study focused on the Queensland section of the Tweed-Moreton bioregion, which has a mapped shoreline length of approximately 3,246 kilometres extending from Bundaberg to the Queensland/New South Wales border (Figure 9). The mapped shoreline includes 1,625 kilometres classified as mainland coastline (marine or estuarine) (Banks & Skilleter 2002). Approximately 321 kilometres of the mainland coastline were classified as marine and 1,304 kilometres classified as estuarine. The mainland coastline of the Tweed-Moreton bioregion is dominated by vast stretches of wide sandy beaches backed by sand dunes (Endean et al. 1956a; Banks & Skilleter 2002). The long beaches are interspersed with rocky shores of varying substratum composition and stability. The bioregion (Queensland and New South Wales sections) covers 42,713 km^2 and contains a variety of biota associated with continental shelf waters and several large estuaries (e.g. Moreton Bay and Hervey Bay). Oceanographic circulation is dominated by the East Australian Current that flows southwards along the continental shelf from the Great Barrier Reef (Endean et al. 1956a; Hamon 1962, 1965).

4.2.2 Description of habitats and microhabitats on rocky shores

Rocky shores in south east Queensland are variable in size and composition of habitats, comprising a mixture of flat pavement and patches of sand, coarse rubble and boulder fields. The extent of each of these types of habitat appears to vary among the different shores, potentially providing different resources to communities of plants and animals.

All mainland rocky shores (41 sites) in south east Queensland were mapped to determine the distribution and extent of the different intertidal habitats present between low and high water on each shore. Mapping was done for shoreline types classified by Banks and Skilleter (2002) as either ‘wide rock platform’, ‘narrow rock platform’, ‘narrow rock ramp’ or ‘rock cliff’ (Figure 9; Table 11). Digital video and aerial photographs (Scale 1:12 000) of the shoreline were used to map the spatial extent of each of the habitats. Digital videography was shot from an altitude of approximately 70–100 metres above sea level to enable identification of habitats. A 3CCD Digital Video Camera was used to record the shoreline during low tide. Other attributes recorded for each site included tidal range, length of shoreline segment, adjacent (i.e. alongshore segments) shoreline types (e.g. wide sand beach – see Banks and Skilleter (2002) for further details) and adjacent subtidal substratum (i.e. sand or rocky reef).

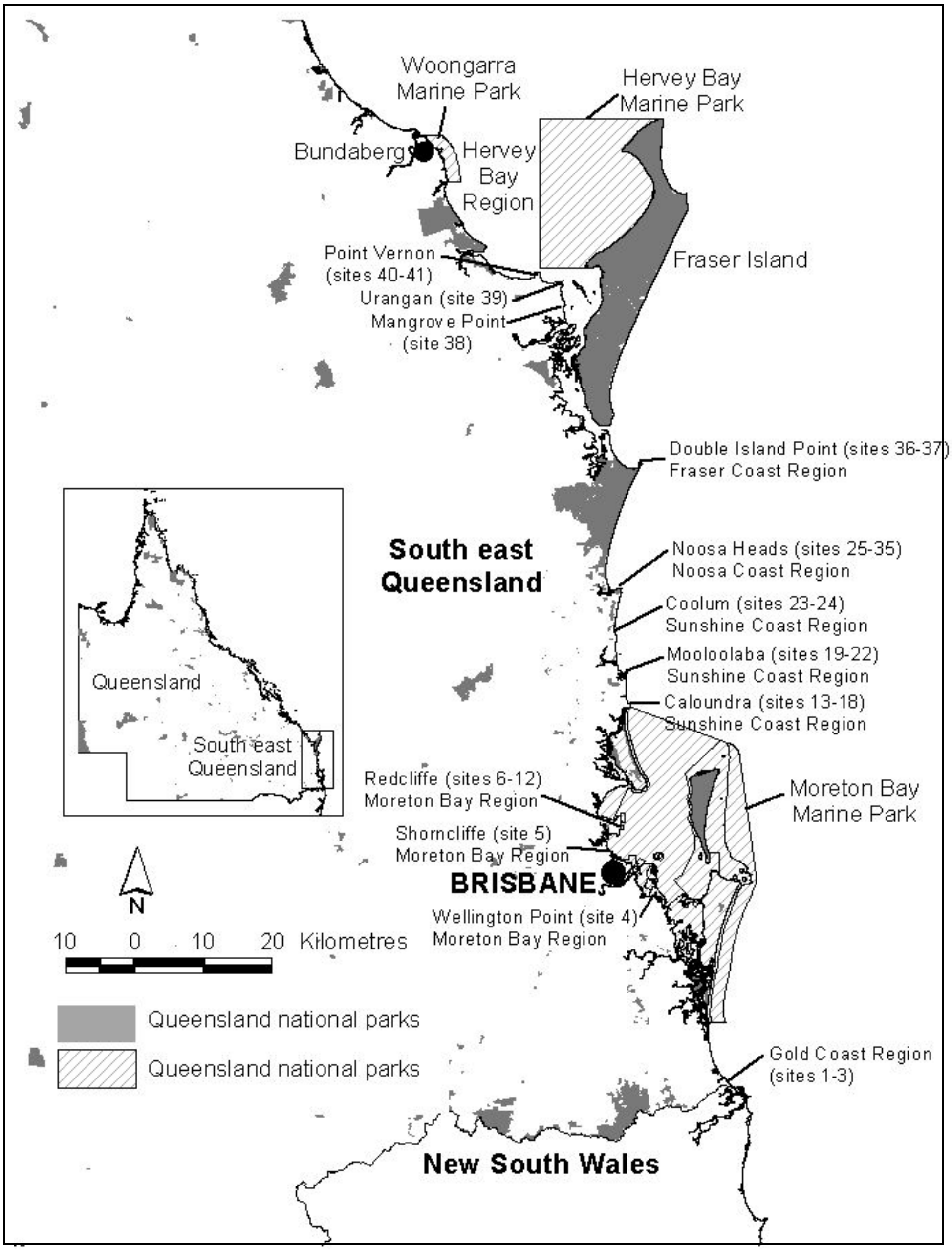


Figure 9: Rocky shore sites mapped for the spatial extent of intertidal habitats and the presence/absence of microhabitats in south east Queensland.

Table 11: Shoreline type, the spatial extent of intertidal habitats and number of microhabitat features for 41 rocky shore sites in south east Queensland. (WRP: wide rock platform; NRP: narrow rock platform; NRR: narrow rock ramp; RC: rock cliff).

Site no.	Site name	Shoreline type	Area (ha)	Length (m)	Spatial extent of habitats mapped on rocky shores (ha)										Presence/Absence of microhabitats (no. recorded)
					Rock platform	Boulder fields	Cobble beach	Sand patches	Mixed fines sediment	Shallow rock pool	Deep rock pool	Lagoon	Artificial substrate	Mangroves	
1	Point Lookout	WRP	0.87	236.0	0.78	–	–	0.06	–	0.01	0.01	–	0.01	–	11
2	Elephant Rock	WRP	0.53	219.0	0.21	–	–	0.04	–	–	–	–	0.28	–	10
3	Currumbin	WRP	0.86	241.0	0.69	–	–	0.15	–	0.02	–	–	–	–	13
4	Wellington Point	WRP	1.33	44.9	0.52	–	–	0.03	0.69	–	–	–	0.07	–	6
5	Shorncliffe	WRP	1.88	236.5	1.19	–	–	0.15	0.54	–	–	–	–	–	8
6	Redcliffe	WRP	1.10	221.0	0.77	–	–	–	0.33	–	–	–	–	–	8
7	Redcliffe	WRP	3.08	311.0	2.48	–	–	–	0.60	–	–	–	0.01	–	8
8	Redcliffe	WRP	2.46	562.2	1.59	–	–	0.55	0.32	–	–	–	–	–	10
9	Redcliffe	NRR	0.61	78.8	0.46	–	–	0.15	–	–	–	–	–	–	5
10	Redcliffe	NRR	0.26	179.4	0.24	–	–	–	–	–	–	–	0.02	–	9
11	Redcliffe	WRP	1.28	272.0	1.03	–	–	0.20	0.20	–	–	–	0.05	–	13
12	Redcliffe	WRP	4.82	1020.0	4.20	–	–	–	0.54	–	–	–	0.08	–	7
13	Caloundra	WRP	1.00	513.0	0.80	0.12	–	0.01	–	–	–	–	0.08	–	10
14	Caloundra	WRP	9.75	1259.0	7.37	0.51	–	1.09	–	0.20	–	0.50	0.08	–	18
15	Caloundra	WRP	1.05	199.0	0.71	–	–	0.34	–	–	–	–	–	–	8
16	Caloundra	WRP	3.93	616.0	2.12	0.38	–	0.49	–	0.17	–	0.77	–	–	16
17	Caloundra	WRP	2.15	143.0	1.56	–	–	0.59	–	–	–	–	–	–	8
18	Caloundra	WRP	0.90	163.0	0.54	–	–	0.36	–	–	–	–	–	–	8
19	Mooloolaba	WRP	5.53	729.0	4.37	0.34	–	0.55	–	–	–	0.27	–	–	18
20	Mooloolaba	WRP	8.71	1037.0	5.58	–	–	1.78	–	–	–	1.35	–	–	20
21	Mooloolaba	WRP	4.56	307.0	2.52	–	–	2.04	–	–	–	–	–	–	14
22	Mooloolaba	WRP	0.84	151.0	0.78	–	–	0.07	–	–	–	–	–	–	10
23	Coolum	WRP	4.71	1343.0	3.54	0.10	–	0.70	–	0.01	–	0.37	–	–	21
24	Coolum	WRP	1.55	708.0	1.55	–	–	–	–	0.01	–	–	–	–	17
25	Noosa Heads	NRR	0.29	125.4	0.29	–	–	–	–	–	–	–	–	–	12
26	Noosa Heads	RC	0.48	307.9	0.44	–	0.04	–	–	–	–	–	–	–	9
27	Noosa Heads	RC	0.19	97.5	0.16	–	0.03	–	–	–	–	–	–	–	8
28	Noosa Heads	WRP	0.42	124.9	0.41	–	–	–	–	0.01	–	–	–	–	11
29	Noosa Heads	RC	0.62	348.0	0.59	0.02	–	0.01	–	–	–	–	–	–	8
30	Noosa Heads	NRR	1.58	790.0	1.54	–	0.02	–	–	–	0.02	–	–	–	15
31	Noosa Heads	NRR	0.45	151.0	0.34	–	0.11	–	–	–	–	–	–	–	16

Table 11: continued

32	Noosa Heads	NRR	0.58	354.0	0.58	–	–	–	–	–	–	–	–	–	11
33	Noosa Heads	NRP	1.76	262.0	1.65	0.05	0.08	–	–	–	–	–	–	–	13
34	Noosa Heads	NRP	0.41	328.0	0.32	0.09	–	–	–	–	–	–	–	–	13
35	Noosa Heads	NRP	0.41	311.0	0.41	–	–	–	–	–	–	–	–	–	12
36	Double Island Point	RC	2.17	1552.9	1.96	0.06	–	0.15	–	–	–	–	–	–	9
37	Double Island Point	NRP	1.19	1121.1	1.01	0.14	0.01	0.03	–	–	–	–	–	–	23
38	Mangrove Point	WRP	16.02	546.0	5.92	–	–	–	1.25	–	–	–	–	8.85	7
39	Urangan	WRP	0.43	100.0	0.31	–	–	0.02	0.03	–	–	–	0.08	–	10
40	Point Vernon	WRP	69.54	4775.0	62.86	–	0.32	5.66	0.38	–	–	–	–	0.31	23
41	Point Vernon	WRP	15.42	748.0	12.38	–	0.56	1.48	0.78	–	–	–	0.01	0.21	23

The availability of different microhabitats was qualitatively assessed for each of the 41 rocky shores. The assessment was based on ground truthing each site supplemented by oblique low level aerial videography. Each site was evaluated for the presence/absence of microhabitats that are considered to affect the distribution of biota on rocky shores (Table 12).

Table 12: Habitats and microhabitats and other factors known to influence the distribution and abundance of intertidal organisms.

Microhabitat	Source	Applied in Banks and Skilleter (2002) classification
Rock pools/ gutters/ lagoons	Endean et al. (1956a) Garrity (1984) Underwood & Jernakoff (1984) Griffiths (2003)	No
Pits/cracks/crevice	Endean et al. (1956a) Garrity (1984) Seapy & Littler (1978) Menge et al. (1985) McGuinness & Underwood (1986) Menconi et al. (1999)	No
Topography and substratum	Menge et al. (1985) McGuinness (1986) Underwood & Chapman (1989) Cruz Motta et al. (2003)	No
Biologically generated habitat	Endean et al. (1956a) Seed (1996) Thompson et al. (1996) Monteiro et al. (2002)	No
Boulders, cobbles	McGuinness (1986) McGuinness (1990) Endean et al. (1956a)	Yes – recorded for alongshore and across-shore components
Gravel and sand	McGuinness (1987a, 1987b)	Yes – recorded for alongshore and across-shore components
Overhangs	Endean et al. (1956a)	No
Bare rock	Thompson et al. (1996)	No
Physical stress (e.g. desiccation)	McGuinness (1990)	No
Mechanical stress (e.g. exposure)	Endean et al. (1956a) McGuinness (1987a, 1987b) Underwood & Chapman (1998b)	Yes – relative exposure for shoreline segments
Horizontal/vertical rock faces	Endean et al. (1956a) Garrity (1984)	No

4.2.3 Data analyses

The purpose of these analyses was to determine if there are any natural groupings in the patterns of intertidal habitats or microhabitats on rocky shores in south east Queensland. The Euclidean distance measure was used to construct separate similarity matrices based on the percentage cover of the various intertidal habitats or presence/absence of microhabitats on each of the 41 rocky shore sites. Euclidean distance has been recommended for environmental variables, which in this study were the habitats and microhabitats present at the 41 rocky shore sites (Clarke 1993; Clarke & Warwick 1994). All multivariate analyses were carried out using PRIMER 5.2 (Clarke & Gorley 2001). A Euclidean distance matrix was calculated for percentage cover of intertidal habitats using fourth root transformed data to allow greater emphasis on rarer habitats on the rocky shores. The presence/absence of microhabitats used non-transformed data. Ordination of data was done using non-metric Multidimensional Scaling (nMDS), which provided graphic representation of the patterns of structure of intertidal habitats and microhabitats on rocky shores.

One-way Analysis of Similarities (ANOSIM) was used to determine whether the abundance of intertidal habitats or the presence/absence of microhabitats differed significantly among: 1) the pre-defined rocky shoreline types (classified by Banks & Skilleter 2002); 2) based on the influence of marine or estuarine conditions on the rocky shores; and 3) based on the regional location of the 41 rocky shore sites (i.e. Gold Coast, Moreton Bay, Sunshine Coast (i.e. Caloundra, Mooloolaba, Coolum), Noosa Coast, Fraser Coast, Hervey Bay). The latter analysis would assist with determining other important factors (i.e. regional location of sites) to be considered in planning a reserve system to protect representative examples of intertidal habitats and species using surrogate approaches for site identification. For each of the one-way ANOSIM tests, the null hypothesis that there were no significant differences in the composition of intertidal habitats or presence/absence of microhabitats among groups was rejected when the significance level (P) was $<5\%$. The R-statistic value was used to determine the extent of any significant differences produced (Clarke & Green 1988; Clarke 1993).

Where an ANOSIM test detected a significant difference in the patterns of distribution of habitats or microhabitats among a priori groups (i.e. rocky shoreline types or regional location of sites), SIMPER (SIMilarities PERcentages) (Clarke 1993) was performed to identify which habitats and microhabitats contributed the most to the average dissimilarity between groups.

4.2.4 Reserve selection algorithm

MARXAN (v1.8.6) was used to identify potential combinations of shoreline types that should be included in a representative system of marine reserves (Possingham et al. 2000). This was done firstly by using simple shoreline types (i.e. lineal segments of coastline), as the conservation features, to identify optimal reserve design solutions representative of the range of these features.

Additional conservation features (i.e. the spatial extent of different habitats on the rocky shores or the presence/absence of microhabitats) were then included, separately, in the dataset to determine whether the system of marine reserves changed considerably when these intertidal features were incorporated.

Simulated annealing was the optimisation method used to find good solutions to the problem of representing all the shoreline types, with and without inclusion of the data on the spatial extent of the intertidal habitats or the presence/absence of microhabitats on the rocky shores to a pre-defined percentage target, while minimising total cost (a weighted sum of area and boundary length). The summed irreplaceability of a site is the percentage of times each planning unit is chosen amongst the various solutions (Pressey et al. 1993). Summed irreplaceability produces a value between 0 and 1 for each planning unit. A unit that was allocated a value close to 1 is necessary for inclusion to meeting conservation goals, whereas a unit allocated a low value would be one that is unlikely to be required (Possingham et al. 2000; Leslie et al. 2003; Stewart et al. 2003).

4.2.4.1 Scenarios explored

Thirty-six scenarios were explored for identifying possible conservation areas in the Tweed-Moreton bioregion for the protection of at least one representative example of the conservation features. The ‘best’ of 100 runs for each scenario was compared.

Planning unit: to determine conservation priorities for the Tweed-Moreton bioregion, mainland intertidal habitats were grouped into one kilometre lengths (i.e. 1,608 planning units) to delineate the spatial location of potential (candidate) sites to be included in a marine reserve system. The basic data matrix for input into the reserve system identification problem consisted of the following for each planning unit: (i) the lengths of each shoreline type; (ii) the spatial extent of habitats on rocky shores; and (iii) presence/absence of microhabitats on rocky shores.

Conservation target: the conservation target was set at 20% of the mapped length of each shoreline type, 20% of the mapped area of each rocky shore intertidal habitat, and 20% of sites that contained each type of microhabitat (Table 13). Targets of 20–30% of each habitat type have been recommended (United Nations 2002a; IUCN 2004). Six artificial shoreline types that were mapped for the south east Queensland coast were not included as conservation features.

Table 13: Results of scenarios explored to identify priority sites for the conservation of intertidal habitats in south east Queensland. The conservation feature target or all scenarios was 20%.

Scenarios	Target	BLM	Status	Overall conservation feature target met	Number of planning units
1	20% of shoreline	0	Free	Yes	319
2	types	0.1	Free	Yes	321
3		0.5	Free	Yes	321
4		1	Free	Yes	324
5	20% of shoreline	0	Marine parks locked in	Yes	317
6	types	0.1	Marine parks locked in	Yes	321
7		0.5	Marine parks locked in	Yes	318
8		1	Marine parks locked in	Yes	323
9	20% of shoreline	0	National parks locked in	Yes	377
10	types	0.1	National parks locked in	Yes	383
11		0.5	National parks locked in	Yes	415
12		1	National parks locked in	Yes	426
13	20% of shoreline	0	Free	Yes	319
14	types plus spatial	0.1	Free	Yes	320
15	extent of habitats	0.5	Free	Yes	323
16		1	Free	Yes	325
17	20% of shoreline	0	Marine parks locked in	Yes	317
18	types plus spatial	0.1	Marine parks locked in	Yes	320
19	extent of habitats	0.5	Marine parks locked in	Yes	320
20		1	Marine parks locked in	Yes	319
21	20% of shoreline	0	National parks locked in	Yes	378
22	types plus spatial	0.1	National parks locked in	Yes	383
23	extent of habitats	0.5	National parks locked in	Yes	413
24		1	National parks locked in	Yes	433
25	20% of shoreline	0	Free	Yes	312
26	types plus	0.1	Free	Yes	312
27	presence/absence of	0.5	Free	Yes	313
28	microhabitats	1	Free	Yes	314
29	20% of shoreline	0	Marine parks locked in	Yes	307
30	types plus	0.1	Marine parks locked in	Yes	311
31	presence/absence of	0.5	Marine parks locked in	Yes	311
32	microhabitats	1	Marine parks locked in	Yes	318
33	20% of shoreline	0	National parks locked in	Yes	377
34	types plus	0.1	National parks locked in	Yes	379
35	presence/absence of	0.5	National parks locked in	Yes	424
36	microhabitats	1	National parks locked in	Yes	421

The choice of the 36 scenarios was based on consideration of several features that were varied, including:

i. Conservation features: 1) 32 shoreline types (Banks and Skilleter (2002), (2) 32 shoreline types plus the spatial extent of 10 intertidal habitats mapped on 41 rocky shore sites, or (3) 32 shoreline types plus the presence of up to 39 microhabitats on 41 rocky shore sites (Table 13);

ii. Boundary length modifier: varied to determine the relative importance of reserve system compactness. The algorithm ignores the boundary length of planning units when the boundary length modifier is set at zero and compactness becomes increasingly important as the boundary length modifier is increased. To determine the influence of the boundary length modifier on the identification of sites for conservation priorities in south east Queensland a boundary length modifier of 0, 0.1, 0.5 and 1 was used; and

iii. Current protective status: 1) scenarios 1–4; 13–16; 25–28: No reserve scenarios (free) ignores the current status of existing protection provided by marine parks and national parks. No planning units were locked into the reserve system, 2) scenarios 5–8; 17–20; 29–32: No-take marine park zones fixed scenarios generates a system based on locking into the reserve system solution all ‘no-take’ zones (equivalent to IUCN Categories Ia and II (IUCN/WCMC 1994; Wells & Day 2004; Dudley 2008)) within the existing system of marine parks. Nine planning units containing existing no-take zones were locked into the reserve system solution, and 3) scenarios 9–12; 21–24; 33–36: National park fixed scenarios generates a system based on locking into the reserve system solution all planning units adjacent to terrestrial national parks or nature reserves. A total of 180 planning units were locked into the reserve system solution in these scenarios. National parks and nature reserves that border the marine environment would prohibit development in the terrestrial environment. They do not however prohibit collection of invertebrates or fishing in the intertidal zone.

4.2.4.2 Cost

In this exercise, the objective was to minimise the total cost of the system in terms of total length of reserves, while ensuring that at least 20% of every one of the conservation features (i.e. proportion of shoreline types, habitats or microhabitats) were represented across the marine reserve system. The overall cost of a system is a combination of purchase (or compensation), dedication and ongoing management. While direct measures of cost can be hard to obtain, area and boundary length are useful surrogates, which were used in this analysis.

A relative cost was measured rather than a real cost associated with the exclusion of commercial extraction. The length of each across-shore component of a planning unit that contained road, industry or residential areas or other artificial features determined the relative cost. Where these features were absent in the across-shore components planning units had a relative cost of zero.

In all scenarios, sites adjacent to residential areas are assumed to have greater social or economic cost associated with the planning and management of a multiple-use marine park than those more distant. Developing a zoning plan would normally involve the closure of certain areas to all forms of extraction (e.g. recreational and commercial fishing). It is assumed that the process of removing or ceasing exploitation may have a higher cost (i.e. enforcement and compliance or political costs) associated with the implementation of a management regime in areas adjacent to residential or tourist development where use may be higher.

4.2.5 Habitat surrogate efficacy

The efficacy of the shoreline type as a surrogate was assessed by comparing those planning units selected based on: (1) the spatial extent of habitats on rocky shores; and (2) the presence of intertidal microhabitats on rocky shores, with the planning units selected using the shoreline type data only.

4.3 RESULTS

4.3.1 Intertidal habitats and microhabitats on rocky shores

A total of 176 hectares of intertidal habitats was mapped for the 41 rocky shore sites, which consisted of 78% consolidated rock platform, 9.5% sand patches, 5.3% mangrove (95% of these mangroves were from one site, occurring at the top of the shore at Mangrove Point, Hervey Bay) and 3.2% mixed fines beach. Other habitats (e.g. lagoons, cobble beaches) accounted for less than five per cent of the intertidal habitats mapped for the rocky shores.

The presence of microhabitats varied considerably among the 41 rocky shore sites, ranging from five types of microhabitat for a site at Redcliffe (Moreton Bay) to 23 types recorded at Double Island Point and Point Vernon. The average number of microhabitats recorded for rocky shores classified as marine and estuarine was 12.9 (± 0.81 SE) and 10.5 (± 1.63 SE) respectively (Table 11).

4.3.2 Multivariate analyses

Spatial extent of habitats: When the 41 rocky shore sites were grouped according to the categories of rocky shoreline types (i.e. wide rock platform, narrow rock platform, narrow rock ramp and rock cliff as defined by Banks & Skilleter 2002), there was no significant difference in the spatial extent of the different intertidal habitats among these groups (ANOSIM, Global $R = -0.151$, $p > 0.05$).

There were significant differences in the abundance of the different types of habitats on rocky shores when the sites were grouped according to their regional location (e.g. Gold Coast versus Moreton Bay) (ANOSIM, Global $R = 0.454$, $p < 0.05$). Pairwise comparisons in this ANOSIM test also showed that the most significant differences in the spatial extent of intertidal habitats occurred between groups of sites that are influenced by oceanic conditions (e.g. Sunshine Coast), and those sites that are influenced by estuarine conditions (e.g. Moreton Bay).

There was a significant difference between sites dominated by oceanic conditions (e.g. Gold Coast, Sunshine Coast, Noosa Coast) versus those sites (e.g. Moreton Bay, Hervey Bay) that are strongly influenced by estuarine conditions (ANOSIM, Global $R = 0.28$, $p < 0.05$). Regional groups, defined a priori, for the spatial extent of intertidal habitats on rocky shores, superimposed on the nMDS ordination (Figure 10a), shows an overlap amongst sites influenced by marine conditions. There is also an overlap amongst sites (e.g. Moreton Bay, Hervey Bay) influenced by estuarine conditions.

The spatial extent of patches of mixed fines sediment contributed to the average dissimilarity between groups of sites within estuaries (i.e. Hervey Bay and Moreton Bay) and groups of sites dominated by oceanic conditions (e.g. Sunshine Coast) (Figure 11). Habitats that were most important in separating the groups of sites dominated by oceanic conditions (e.g. Gold Coast versus Noosa Coast) into regional locations were the proportion of sand and artificial substrate at, for example, Gold Coast sites.

Presence/absence of microhabitats: One-way ANOSIM tests showed that there was no significant difference in the presence/absence of microhabitats among the different shoreline types classified by Banks and Skilleter (2002) (Global $R = -0.061$, $p > 0.05$). Pairwise comparisons did however show that there were significant differences between rock cliffs, and narrow rock ramps (0.391, $p < 0.05$) and narrow rock platforms (0.693, $p < 0.05$).

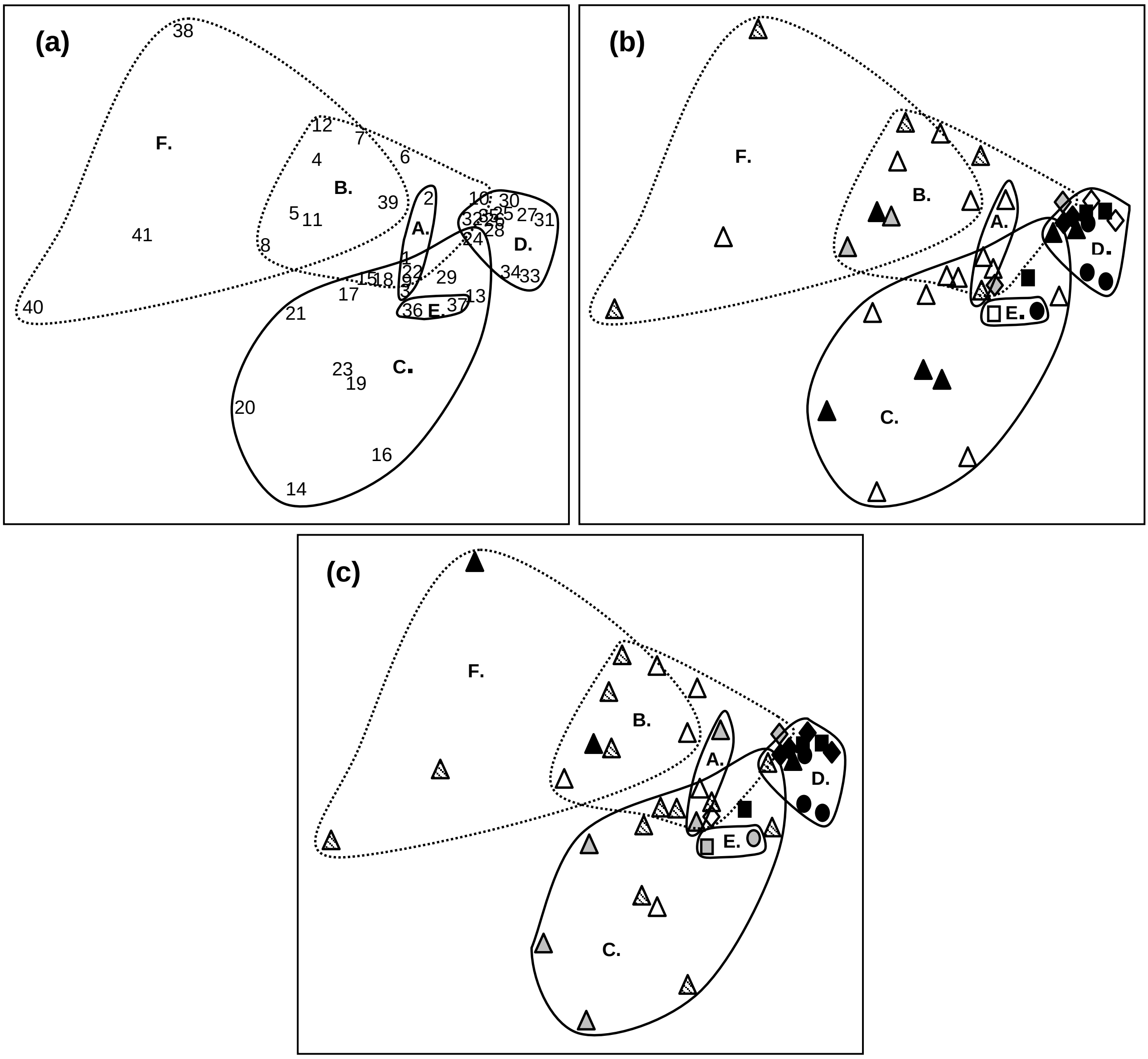


Figure 10: (a) nMDS ordination of the spatial extent of habitats mapped from 41 rocky shore sites in south east Queensland grouped into regional location (Kruskal's stress 0.14); **(b)** Irreplaceability of sites from MARXAN based on shoreline types only (Scenario 4); **(c)** Irreplaceability of sites from MARXAN based on shoreline types plus the spatial extent of habitats (Scenario 16). Key to symbols: A. Gold Coast; B. Moreton Bay; C. Sunshine Coast; D. Noosa Coast; E. Fraser Coast; F. Hervey Bay. Rocky shoreline types – wide rock platform (triangle), narrow rock platform (circle), narrow rock ramp (diamond), rock cliff (square). Irreplaceability score: solid black = high irreplaceability (0.91–1.0); solid grey =moderate irreplaceability (0.51–0.9); striped = low irreplaceability (0.1–0.5); open = not selected (irreplaceability = 0). Solid line – marine sites; dotted line – estuarine sites.

	Gold Coast	**Moreton Bay**	**Sunshine Coast**	**Noosa Coast**	**Fraser Coast**
Moreton Bay	Mixed fines sediment (35%) Artificial (23%)				
Sunshine Coast	Artificial (28%) Sand (19%) Lagoon (16%)	Mixed fines sediment (34%) Sand (25%)			
Noosa Coast	Sand (35%) Artificial (23%)	Mixed fines sediment (37%) Sand (21%)	Sand (41%) Cobble (15%)		
Fraser Coast	Boulder (34%) Artificial (27%)	Mixed fines sediment (28%) Boulder (34%)	Boulder (31%) Sand (29%)	Boulder (36%) Sand (35%)	
Hervey Bay	Sand (35%) Artificial (24%)	Mixed fines sediment (25%) Sand (23%) Mangroves (22%)	Mixed fines sediment (20%) Sand (19%) Mangroves (17%)	Mixed fines sediment (22%) Sand (21%) Mangroves (18%)	Boulder (27%) Mixed fines sediment (22%) Sand (18%)

Figure 11: Results of SIMPER analysis of the spatial extent of habitats mapped from 41 rocky shore sites in south east Queensland, showing habitats and their percentage contribution to the separation of the regional groups of sites from each other.

The analysis of sites based on their regional location, as grouped on the nMDS (Figure 12a), showed that the presence/absence of microhabitats on rocky shores from different regions were significantly different (Global R = 0.584, $p < 0.05$). Pairwise comparisons showed that there was significant difference between groups of sites influenced by oceanic conditions (e.g. Gold Coast) versus estuarine conditions (e.g. Moreton Bay) (R = 0.99, $p < 0.01$).

Microhabitat features that were most important in separating the estuarine groups of sites (Moreton Bay and Hervey Bay) were the presence of mixed fines sediment and gravel/rubble (Figure 13). The presence of vertical rock faces distinguished the Gold Coast group from other regional locations. The group of Mooloolaba sites were distinguished from other regional locations based on the presence of sand or boulders in gutters, and the feature that was an important contributor to distinguishing the Caloundra group of sites was the presence of ascidian beds.

Microhabitat complexity was higher on wide rock platforms (average 12.4 (± 1.00 SE)), narrow rock platforms (average 15.3 (± 2.59 SE)), narrow rock ramps (average 11.3 (± 1.65 SE)) when compared with rock cliffs (average 8.5 (± 0.29 SE)).

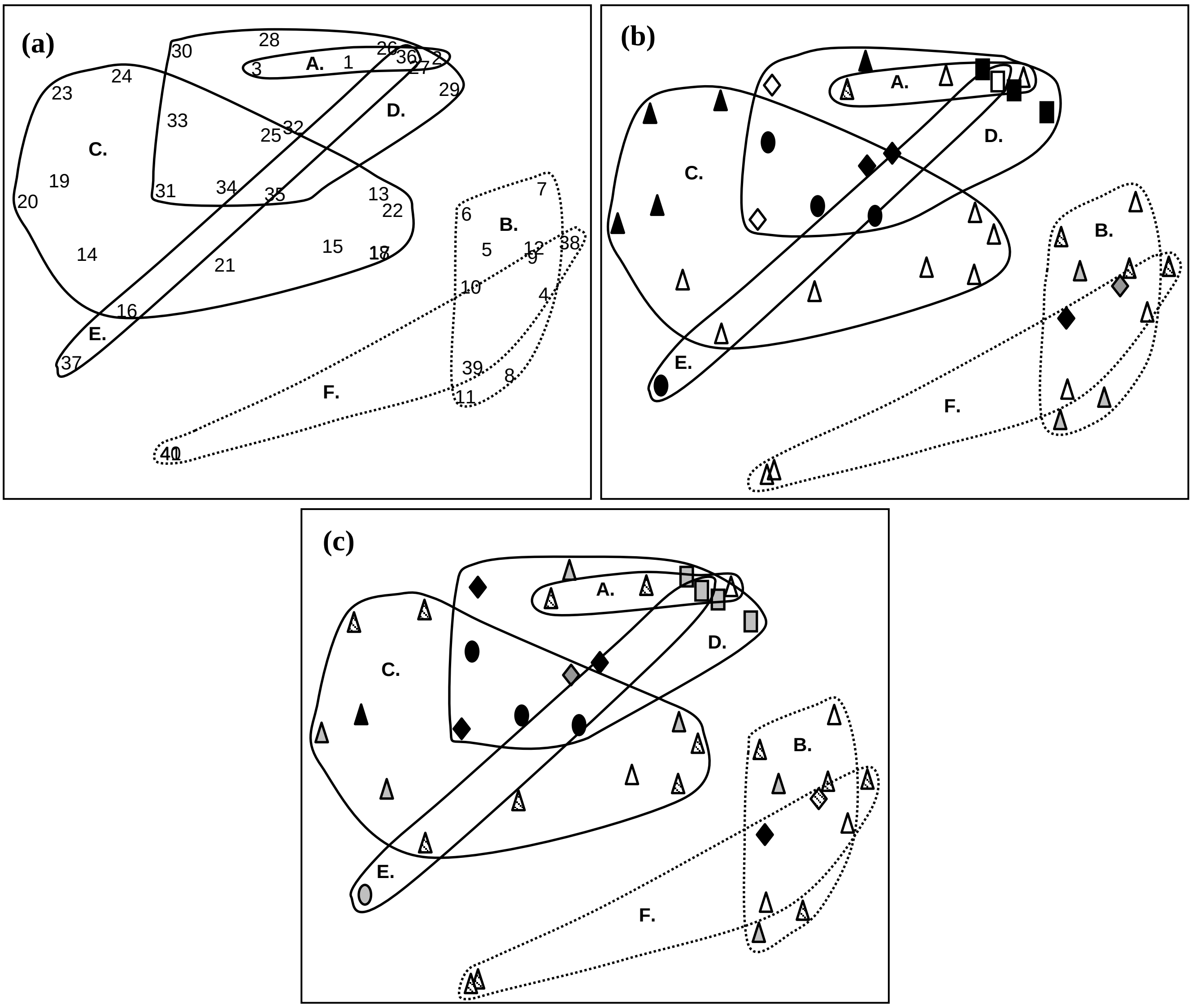


Figure 12: (a) nMDS ordination for presence/absence of microhabitats mapped from 41 rocky shore sites in south east Queensland (Kruskal's stress 0.13) showing the groups representing regional locations of the 41 sites; **(b)** Irreplaceability of sites from MARXAN based on shoreline types only (Scenario 4); **(c)** Irreplaceability of sites from MARXAN based on shoreline types plus the presence/absence of microhabitats (Scenario 28). Key to symbols: A. Gold Coast; B. Moreton Bay; C. Sunshine Coast; D. Noosa Coast; E. Fraser Coast; F. Hervey Bay. Rocky shoreline types: wide rock platform (triangle), narrow rock platform (circle), narrow rock ramp (diamond), rock cliff (square). Irreplaceability score: solid black = high irreplaceability (0.91–1.0); solid grey = moderate irreplaceability (0.51–0.9); striped = low irreplaceability (0.1–0.5); open = not selected (irreplaceability = 0). Solid line – marine sites; dotted line – estuarine sites.

Gold Coast					
Flat platform (8.0%)					
Parallel landward facing platform (8.0%)	**Moreton Bay**				
Dry crevices (8.0%)					
Wide gutters (8.0%)					
Flat platform (8.3%)	Gravel/rubble (7.0%)				
Parallel landward facing platform (8.3%)	Water present in gutters (7.0%)	**Sunshine Coast**			
Vertical rock faces (7.2%)	Narrow gutter (6.1%)				
Water present in crevices (5.7%)	Mixed fines sediment (6.0%)				
Parallel landward facing platform (8.0%)	Dry crevices (8.2%)	Sand patches (7.8%)			
Sand patches (9.9%)	Gravel/rubble (7.3%)	Dry crevices (6.1%)	**Noosa Coast**		
Vertical rock faces (7.6%)	Mixed fines sediment (6.3%)	Adjacent subtidal rocky reef (5.6%)			
Flat platform (7.6%)	Narrow gutters (5.9%)	Adjacent subtidal sand (5.4%)			
Boulders (11.3%)	Boulders (7.5%)	Adjacent subtidal rocky reef (5.4%)	Sand patches (8.6%)		
Water present in gutters (7.8%)	Dry crevices (7.5%)	Dry crevices (5.4%)	Boulders (7.0%)	**Fraser Coast**	
Parallel landward facing platform (7.1%)	Wide gutters (7.5%)	Flat platform (5.1%)	Narrow gutters (5.8%)		
Narrow gutters (7.1%)	Water present in gutters (6.7%)	Water present in crevices (5.1%)	Vertical rock faces (5.2%)		
Mixed fines sediment (6.7%)	Mangroves (7.2%)	Mixed fines sediment (6.9%)	Mixed fines sediment (7.4%)	Boulders (7.1%)	
Vertical rock faces (6.7%)	Platform perpendicular to shoreline (7.2%)	Gravel/rubble (4.9%)	Gravel/rubble (5.2%)	Mixed fines sediment (7.1%)	**Hervey Bay**
Flat platform ((6.7%)	Shallow rock pools - sand present (6.3%)	Mangroves (4.9%)	Mangroves (5.2%)	Mangroves (5.1%)	
Parallel landward facing platform (6.7%)	Boulder clumps (6.2%)	Platform perpendicular to shoreline (4.9%)	Sand present in crevices (4.9%)	Platform perpendicular to shoreline (5.1%)	

Figure 13: Results of SIMPER analysis for presence/absence of microhabitats mapped from 41 rocky shore sites in south east Queensland, showing microhabitats and their percentage contribution to the separation of the regional groups from each other.

4.3.3 Reserve system solutions

The proportion of planning units included in a reserve system to achieve 20% representation of each shoreline type resulted in the selection of 19–20% of the coastline in the bioregion. The overall length of mainland shoreline required for protection was between 307 to 325 kilometres in length. There was no increase in the length of coast required for protection when existing no-take zones of marine parks were locked into the reserve system solution. When national parks were locked into the reserve system solution the number of planning units selected increased by 3–6%. This resulted in protection to the mainland shoreline of between 377 to 433 kilometres.

4.3.4 Conservation feature representation

The average quantity of each shoreline type selected for inclusion in a reserve system ranged from 20–100% of their mapped length. For the best solution for scenarios 1–8, 19 of the 32 shoreline types were over-represented (i.e. more than 20% of each feature was included in the reserve system). This included eight shoreline types that had a mapped length of less than one kilometre and 11 that had mapped lengths of between one to five kilometres. Where conservation targets were met shoreline types generally had a larger mapped length available for inclusion in the reserve system. Where mapped lengths were small, all or most of the shoreline type may have been included in a single planning unit. If this planning unit was selected to be included in the reserve system the result would be over-representation.

Scenarios 8–12 had 22 of the 32 shoreline types over-represented including nine shoreline types that had a mapped length of less than one kilometre and 11 that had mapped lengths of between one to five kilometres. A further two had mapped lengths greater than 50 kilometres. Those shoreline types that had mapped lengths greater than 50 kilometres (i.e. sand beaches and estuarine sand flats) bordered national parks in the south east Queensland region and were therefore locked into the system for these scenarios.

The 20% target for the spatial extent of intertidal habitats on rocky shores was met for scenarios 13–24. Similarly the target to achieve representation of microhabitats on rocky shores in the reserve system was met for scenarios 25–36. There tended to be over-representation of the intertidal habitats and microhabitats in the best solution for each scenario.

The results of the best reserve solutions showed that planning units selected for inclusion into a system of marine reserves would result in the protection of the range of microhabitats present on rocky shores in south east Queensland and that in general conservation targets were met or exceeded (Table 14). In many instances the planning units selected included a mosaic of intertidal habitats and microhabitats.

4.3.5 Reserve system configuration

For all scenarios selection of planning units tended to focus on the Sunshine Coast and Hervey Bay areas, rather than similar shoreline types in Moreton Bay and the Gold Coast region, as the boundary length modifier was increased above zero.

The locking in of planning units adjacent to terrestrial national parks had the greatest impact on the compactness of the reserve system when the boundary length modifier was varied. Between a boundary length modifier of zero and a boundary length modifier of 1 there was a three per cent increase in the length of coastline selected to be included in a reserve system associated with locking the adjacent terrestrial national parks into the reserve solution.

4.3.6 Surrogate efficacy

Of the 41 mainland rocky shore sites mapped in south east Queensland only one site (Shorncliffe – Moreton Bay) had an irreplaceability of one for the best solution of 100 runs for all scenarios.

Biologically generated microhabitats, considered to be important for biodiversity (Monteiro et al. 2002) were not represented in the reserve system solutions. These microhabitats (for example ascidian beds (i.e. *Pyura stolonifera*)) were not extensive at the sites and small clumps tended to be located at the low tide interface of most rocky shores. Ascidian beds were most notable at Caloundra and Moffat Head, which were sites that were not highly irreplaceable for reserve selection.

Table 14: Conservation feature targets and indicating whether the target was met (representation of 20–25% of the mapped length) or exceeded (>25% of mapped length) based for the best run of each scenarios (M: Target met; E: Target exceeded).

			Scenario																																		
	Feature Name	**Target**	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**16**	**17**	**18**	**19**	**20**	**21**	**22**	**23**	**24**	**25**	**26**	**27**	**28**	**29**	**30**	**31**	**32**	**33**	**34**	**35**	**36**
	Adjacent subtidal sand	6.60	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	E	E	E	E	E	E	E	E	E	E	E	E
	Adjacent subtidal rocky reef	2.80	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	E	E	E	E	E	E	E	E	E	E	E	E
	Mangroves	0.60	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	E	E	E	E	E	E	E	E	E	E	E	E
	Ascidian beds	0.60	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	M	M	M	M	M	M	M	M	M	M	M	M
	Algal mats	0.60	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	E	E	M	M	E	E	E	M	E	M	M	M
	Patches of mixed fines sediment	2.20	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	M	M	M	E	M	M	M	M	M	M	M	M
	Sand patches	5.00	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	E	E	E	E	E	E	E	E	E	E	E	E
	Gravel/rubble	2.40	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	E	E	E	E	E	E	E	E	E	E	E	E
	Cobbles	1.60	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	E	E	E	E	E	E	E	E	E	E	E	E
	Boulders	2.40	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	E	E	E	E	E	E	E	E	E	E	E	E
	Isolated boulders	0.20	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	M	M	M	M	M	M	M	M	M	M	M	M
	Clumps of boulders	2.20	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	E	E	E	E	E	E	E	E	E	E	E	E
	Large pits	6.80	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	E	E	E	E	E	E	E	E	E	E	E	E
	Small pits	2.20	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	M	M	M	E	M	M	E	M	E	E	E	E
Microhabitats	Rock slabs	0.40	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	M	M	M	M	M	M	M	M	M	M	M	M
	Lagoons - algal mats present	0.20	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	M	M	M	M	M	M	M	M	M	M	M	E
	Lagoons - sand present	0.40	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	E	E	E	E	E	E	E	E	E	E	E	E
	Lagoons - boulders present	0.80	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	E	E	E	E	E	E	E	E	E	E	E	E
	Deep rock pools - boulders present	0.60	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	M	E	E	E	M	E	E	E	E	E	E	E
	Deep rock pools	0.80	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	M	E	E	E	M	E	E	E	E	E	E	E
	Shallow rock pools - sand present	1.40	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	E	E	E	E	E	E	E	E	E	E	E	E
	Shallow rock pools - boulders present	3.00	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	E	E	E	E	E	E	E	E	E	E	E	E
	Shallow rock pools	4.60	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	E	E	E	E	E	E	E	E	E	E	E	E
	Gutters - water present	5.20	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	E	E	E	E	E	E	E	E	E	E	E	E
	Gutters - boulders present	2.60	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	E	E	E	E	E	E	E	E	E	E	E	E
	Gutters - sand present	1.80	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	E	E	E	E	E	E	E	E	E	E	E	E
	Wide gutters	3.80	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	E	E	E	E	E	E	E	E	E	E	E	E
	Narrow gutters	6.00	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	E	E	E	E	E	E	E	E	E	E	E	E
	Crevices - sand present	1.60	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	E	E	E	E	E	E	E	E	E	E	E	E

Table 14: continued

	Dry crevices	4.60	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	E	E	E	E	E	E	E	E	E	E	E	E
	Crevices - water present	5.60	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	E	E	E	E	E	E	E	E	E	E	E	E
	Crevices	7.20	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	E	E	E	E	E	E	E	E	E	E	E	E
	Rock faces perpendicular to shoreline	1.00	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	E	E	M	E	E	E	M	E	E	E	E	E
	Rock face parallel and landward facing	0.80	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	M	M	M	M	M	M	M	M	E	E	E	E
	Rock faces parallel seaward facing	2.40	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	E	E	E	E	E	E	E	E	E	E	E	E
	Flat rock faces	6.80	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	E	E	E	E	E	E	E	E	E	E	E	E
	Vertical rock faces	2.00	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	M	M	M	M	M	M	E	M	E	E	E	E
Habitats	Mangroves	1.83	–	–	–	–	–	–	–	–	–	–	–	–	E	E	E	E	E	E	E	E	E	E	E	E	–	–	–	–	–	–	–	–	–	–	–	–
	Artificial substrate	0.14	–	–	–	–	–	–	–	–	–	–	–	–	E	E	M	E	E	E	E	E	E	E	E	E	–	–	–	–	–	–	–	–	–	–	–	–
	Lagoon	0.60	–	–	–	–	–	–	–	–	–	–	–	–	E	M	M	E	E	E	E	E	M	E	E	E	–	–	–	–	–	–	–	–	–	–	–	–
	Deep rock pool	0.01	–	–	–	–	–	–	–	–	–	–	–	–	E	E	E	E	E	E	E	E	E	E	E	E	–	–	–	–	–	–	–	–	–	–	–	–
	Shallow rock pool	0.35	–	–	–	–	–	–	–	–	–	–	–	–	E	M	M	M	M	M	M	M	M	M	M	E	–	–	–	–	–	–	–	–	–	–	–	–
	Mixed fines beach	0.87	–	–	–	–	–	–	–	–	–	–	–	–	E	E	E	E	E	E	E	E	E	E	E	E	–	–	–	–	–	–	–	–	–	–	–	–
	Sand patches	3.58	–	–	–	–	–	–	–	–	–	–	–	–	M	M	M	E	E	M	M	E	M	M	M	E	–	–	–	–	–	–	–	–	–	–	–	–
	Cobble beach	0.23	–	–	–	–	–	–	–	–	–	–	–	–	E	M	M	M	M	M	M	M	M	M	M	M	–	–	–	–	–	–	–	–	–	–	–	–
	Boulder fields	0.62	–	–	–	–	–	–	–	–	–	–	–	–	E	E	E	E	E	E	E	E	E	E	E	E	–	–	–	–	–	–	–	–	–	–	–	–
	Rock platform	27.04	–	–	–	–	–	–	–	–	–	–	–	–	M	E	M	M	E	M	M	M	M	M	M	M	–	–	–	–	–	–	–	–	–	–	–	–
Shoreline types	Marine Sand flat wide	58.84	M	M	M	M	M	M	M	M	E	E	E	E	M	M	M	M	M	M	M	M	E	E	E	E	M	M	M	M	M	M	M	M	E	E	E	E
	Marine Sand flat narrow	0.00	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E
	Marine Rock wall	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
	Marine Rock ramp narrow	0.32	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E
	Marine Rock platform wide	2.89	E	M	M	E	E	E	E	E	M	M	M	M	E	E	E	M	E	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M
	Marine Rock platform narrow	0.47	E	E	E	E	E	E	E	E	E	E	E	E	M	E	E	E	E	E	E	E	E	E	E	E	M	M	E	E	M	M	M	E	E	E	E	E
	Marine Rock cliff	0.58	M	M	E	E	M	M	M	E	E	E	E	E	M	E	M	E	M	E	E	E	E	E	E	E	M	M	E	M	M	M	M	E	M	M	M	M
	Marine Piles	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
	Marine Mixed fines flat wide	2.63	E	M	M	M	M	E	M	E	M	E	M	M	E	M	M	M	M	M	M	M	M	M	M	M	E	E	E	M	M	M	M	M	M	M	E	E
	Marine Marina	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
	Marine Inclined sand beach wide	0.45	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E
	Marine Inclined sand beach narrow	0.29	E	E	E	E	E	E	E	E	E	E	E	E	M	E	E	E	E	E	E	E	E	E	E	E	M	M	E	E	M	M	E	E	E	E	E	E
	Marine Inclined mixed fines flat wide	0.31	M	M	M	M	E	E	M	M	M	E	M	M	M	M	M	M	E	M	M	M	M	M	M	M	M	E	E	M	M	M	M	M	M	M	E	M

Table14: continued

Marine Inclined mixed fines flat narrow	0.26	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E
Marine Inclined boulder field narrow	0.12	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E
Marine Flat cobble beach wide	0.06	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E
Marine Flat boulder field wide	4.28	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M
Marine Coffee rock platform wide	0.37	E	E	E	E	E	E	E	E	E	E	E	E	E	M	M	M	M	M	E	M	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	M
Marine Boulder cliff	0.02	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E
Estuarine Steep sand beach	2.69	E	E	E	E	E	E	E	E	E	M	M	M	E	E	E	E	E	E	E	E	E	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M
Estuarine Steep mixed fines flat	8.33	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M
Estuarine Sand flat wide	10.59	M	M	M	M	M	M	M	M	E	E	E	E	M	M	M	M	M	M	M	M	E	E	E	E	M	M	M	M	M	M	M	M	E	E	E	E
Estuarine Sand flat narrow	0.32	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E
Estuarine Rock wall	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
Estuarine Rock ramp narrow	0.07	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E
Estuarine Rock platform wide	0.71	E	E	E	M	E	M	E	M	M	M	E	M	M	M	E	M	E	M	M	M	M	M	M	M	E	E	E	M	E	E	E	E	E	E	M	E
Estuarine Piles	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
Estuarine Mixed fines flat wide	118.77	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M
Estuarine Mixed fines flat narrow	7.55	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M
Estuarine Marina	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
Estuarine Inclined sand beach wide	0.02	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E
Estuarine Inclined sand beach narrow	16.24	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M
Estuarine Inclined mixed fines flat wide	0.94	E	M	M	M	M	M	M	M	E	E	E	E	E	M	E	E	E	M	M	M	E	E	E	E	E	M	M	M	E	M	M	E	E	E	E	E
Estuarine Inclined mixed fines flat narrow	68.22	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M
Estuarine Inclined boulder field narrow	0.06	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E
Estuarine Gravel flat wide	0.22	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E
Estuarine Flat cobble beach wide	0.06	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E
Estuarine Flat boulder field wide	0.22	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E	E

All rocky shoreline types were included in reserve system solutions. Representation of shoreline types, intertidal habitats and microhabitats was achieved. Sites in the Noosa Heads area could be considered a high priority for inclusion in a reserve system due to the diversity of shoreline types in this area and the presence of a national park that protects the adjacent terrestrial environment.

4.3.6.1 Spatial extent of habitats

Sites from the Noosa Heads area represented the focus for highly irreplaceable features of the coastline whether they were selected based on shoreline types alone (Figure 10b), or shoreline types plus the spatial extent of habitats mapped for the intertidal zone of rocky shores (Figure 10c). Sites from Caloundra and Mooloolaba were considered highly irreplaceable based on shoreline types. However, when spatial extent of intertidal habitats was included in the analysis of these sites, while still considered important for a representative reserve system, the sites were not highly irreplaceable.

Mangrove Point became highly irreplaceable (i.e. score of one) when the spatial extent of intertidal habitats were added to the reserve system identification problem (scenarios 13–24), which is due to the presence of the greatest extent of mangroves recorded on a mainland rocky shore in south east Queensland.

4.3.6.2 Presence/absence of microhabitats

Sites from the Sunshine Coast and Noosa Heads area represented the focus for highly irreplaceable features of the coastline based on shoreline types alone (Figure 11b). When shoreline types plus the presence/absence of microhabitats were included in the reserve design problem the sites from the Sunshine Coast became less important in achieving conservation targets (Figure 11c).

4.4 DISCUSSION

There is increasing pressure on intertidal habitats and species resulting from development (Gray 1997; Thompson et al. 2002), introduced species (Thompson et al. 2002), bait gathering (McPhee & Skilleter 2002; McPhee et al. 2002), recreational and commercial fishing and a trend for greater human occupation of the coast. This increases the urgent need to review current approaches to intertidal marine conservation. The equitable allocation of marine resources remains a contentious issue and a challenge for marine reserve decision-makers, particularly given the open access of the marine environment for fishing activities. Restrictions on access associated with protection of representative examples of biodiversity are increasingly required to be based on sound scientific advice.

Rocky shores in Queensland, similar to many locations around the world, lack detailed information about the distribution and abundance of intertidal plants and animals, and the processes structuring intertidal assemblages (Endean et al. 1956a, 1956b; Underwood & Petraitis 1993; Coates 1995, 1998). In the absence of detailed information on the distribution of intertidal biota, marine reserve selection has been based on broad intertidal categories (i.e. sandy beaches, rocky shores) (Ardron et al. 2002; Ardron 2003; Breen et al. 2003). The risk with such approaches is that a system of reserves will continue to fail to represent environmental, habitat or microhabitat diversity, and therefore is not likely to achieve goals of representing the range of intertidal biota.

The study demonstrated that the spatial extent of intertidal habitats and the presence/absence of microhabitats varied significantly based on the regional location of the rocky shore. This was particularly evident when comparing rocky shores influenced by oceanic conditions and those influenced by estuarine conditions. There was evidence that rocky shores influenced by oceanic conditions were more similar to each other in the spatial extent of habitats and presence/absence of microhabitats than locations classified as estuarine. There was also a different suite of habitats on these rocky shores (e.g. patches of sand, boulder fields) compared to estuarine rocky shores (e.g. patches of mixed fines sediment, mangroves). It is likely that prevailing oceanic conditions in south east Queensland are likely to be an important factor that influences the structure of intertidal habitats on rocky shores, on exposed coastlines, when compared to estuarine coastlines, and therefore determine the habitats available for intertidal species to occupy.

There was no relationship between shoreline types based on physical properties of the coastline and the spatial extent of habitats or presence/absence of microhabitats. This raises the question about the validity of the shoreline types as a surrogate for intertidal biodiversity, although it is not known whether the shoreline types reflect differences in the abundance and diversity of intertidal biota. The value of the shoreline types is that they represent alongshore changes in intertidal habitats, which is important for identifying other habitats (e.g. boulder fields) along a section of coast. Boulder fields are known to contain unique species that influence the structure of intertidal assemblages (McGuinness 1986, 1987a; Underwood & Chapman 2001; Chapman 2002; Le Hir & Hily 2005). Further characterisation of coastlines into units such as shoreline types (Banks & Skilleter 2002) would enable habitat diversity to be considered as a minimum requirement to support reserve selection within a biogeographic region.

There were notable habitats or microhabitats not present on some rocky shoreline types. For example, rock cliffs did not contain rock pools or other unconsolidated substrate such as boulder fields or cobbles. Therefore it could be assumed that the use of broad intertidal categories (i.e. rocky shore that includes rock cliffs and rock platforms) as a basis for the identification of a system of marine reserves may fail to represent rock pools, boulder fields or other habitats present on a rock platform. If marine reserve selection was based on broad intertidal categories it would be unknown what types of rocky shore (i.e. rock cliff, rock platform) are potentially available in a region for inclusion into a marine reserve without finer-scale data such as shoreline types. There would be a risk that rock cliffs are selected as an example of a rocky shore more than is warranted, due to their inaccessibility and therefore the protection they receive from human activities. A lack of access could make these types of habitats more socially/politically acceptable for inclusion into a reserve system, than a rock platform that is easily accessible. This would potentially lead to a failure of a reserve system to protect habitats or microhabitats that are often present on rock platforms (e.g. rock pools, boulder fields) but absent or rare on rock cliffs.

The selection of marine reserves must be based on a combination of information on the distribution of along shore units (i.e. shoreline types) within a region, plus the habitats and microhabitats present on rocky shores. Inclusion of all these features in reserve selection will increase the likelihood that the range of intertidal biota is protected in marine reserves. The identification of features such as boulder fields and rock pools, among others, on rocky shores will benefit reserve design while their absence from a reserve system would reduce the certainty that the system is protecting the range of intertidal biodiversity. Additional information was acquired rapidly, about the presence/absence of rock pools and other fine-scale features from rocky shores, which enabled them to be included in the data matrix for reserve system identification. Utilisation of such data should therefore be appealing to agencies charged with the management and conservation of coastal resources but faced with limited financial resources.

The use of siting algorithms with shoreline types representing changes along a coastline and intertidal habitats and microhabitats representing variation on rocky shores will assist conservation decisions. However, if decision makers decide to select sites only that are highly irreplaceable (i.e. irreplaceability score of 0.91–1.0) based on shoreline types there could be a large number of intertidal features not included in a reserve system. There should be caution in the application of reserve design criteria that rely on single components of biodiversity, for example shoreline types only. This research supports the findings of Bonn and Gaston (2005) that focusing on single biodiversity components alone is insufficient to protect other components of biodiversity. The use of a combination of shoreline types, habitats and microhabitats on rocky shores would ensure protection of environmental diversity and features known to influence the distribution and abundance of intertidal biota.

Using a mix of biodiversity components would maximise the range of suitable living conditions for different species, which should guarantee the representation of a diversity of species (Faith & Walker 1996). In addition to using environmental diversity to support reserve site identification other design criteria should be used including, for example, available knowledge of: (i) species distributions, (ii) biodiversity hotspots, and (iii) biogeographic patterns of species. Using a range of criteria and factors to support reserve site selection will further increase the likelihood that reserves will ensure long-term persistence of species. Further work remains to investigate the relationship between shoreline types and species distributions across broad geographic areas.

A number of reserve solutions were identified that would result in the conservation of representative examples of intertidal shoreline types including the mosaic of intertidal habitats and microhabitats present on rocky shores. The habitat surrogate (i.e. shoreline type) used as the baseline data for the reserve design problem is generally reflective of the range and variation of intertidal habitats and microhabitats that are considered to influence the abundance and diversity of intertidal organisms, or contain a suite of unique species present in particular features such as rock pools. The likelihood that a system of reserves would capture the diversity and abundance of intertidal organisms is expected to increase by using a finer-scale physical habitat surrogate and reserve siting algorithm than *ad hoc* or random approaches (McNeill 1994). There are also likely to be advantages in using 'mixed-scale' data to assist reserve selection. For example the use of detailed information on the distribution of habitats and microhabitats for rocky shores but broader-scale information for beaches may be more time and cost effective to achieve conservation goals.

The optimal reserve scenarios proposed here to protect representative examples of intertidal biodiversity could be seen as a first step in the development of a representative system of marine reserves. The scenarios proposed provide decision makers with a scientific basis for the selection of sites to be closed to all forms of extraction. There are at least four strategies that could assist with achieving representation of intertidal habitats and species in reserves: 1) declare new small marine reserves that prohibit extractive activities; 2) establish large multiple-use marine parks with no-take zones protecting representative examples of intertidal habitats; 3) extend national parks to low water and prohibit taking of marine plants and animals in national parks; or 4) declare fishing closures over rocky shore habitats prohibiting the collection of intertidal plants and animals.

There are likely to be benefits to marine reserves if they are established in areas adjacent to existing national parks and nature reserves because the intertidal assemblages will benefit from a more natural set of interactions with adjacent terrestrial systems (Ward & Hegerl in litt.). Ward and Hegerl (in litt.) also suggested that there is potential to share management controls and systems through the adjacent terrestrial reserve. This may benefit management of intertidal habitats through more effective compliance and visitor management.

Extending all or part of coastal terrestrial national parks to low water could contribute to representation of intertidal habitats in marine reserves. The intertidal zone is often a contentious area for inclusion in marine reserves due to the ease of access for fishing and collecting. The political concerns about closing access and excluding extractive uses from the intertidal zone are likely to contribute to lack of representation of intertidal habitats and species in marine reserves. If all, or part of terrestrial national parks were extended to low water and there was an ability to manage fishing and collecting activities it would provide a mechanism to enable the terrestrial reserve system to contribute to protecting intertidal biodiversity.

To overcome the shortcomings of using surrogates, marine reserve planners should use a range of other criteria to assist with prioritisation of sites for protection. The surrogate should not be considered in isolation and should only be used as one of a number of tools to assist decision-making. In cases where the surrogate has been assessed and it has been concluded that it is not useful, utilising other criteria to guide decision-making may overcome this problem (Stevens & Connolly 2004). The challenge for marine reserve planners is to balance the need for urgent action to protect biodiversity and the need to collect additional fine-scale information and data to support decisions. Marine reserve planners, politicians and the community must accept that there is uncertainty with data and define other biodiversity criteria, in addition to available coarse information or habitat surrogates, to increase the likelihood that marine reserves or systems of marine reserves will achieve their goals better than *ad hoc* approaches to reserve design.

CHAPTER 5 — INTEGRATING MARINE CONSERVATION POLICY, SCIENCE AND DECISION-MAKING TO IMPLEMENT MARINE RESERVE NETWORKS

5.1 INTRODUCTION

Protected areas have been recognised as an essential and effective approach to conserving biodiversity in both the terrestrial and marine environments (Possingham et al. 2006; Grantham et al. 2008). They contribute to the conservation of living resources to achieve three conservation objectives: (1) maintenance of essential ecological processes, (2) preservation of genetic diversity, and (3) ensuring sustainable utilisation of species and ecosystems (Kelleher 1999). They are also considered to contribute to broader marine management objectives through habitat conservation, rebuilding depleted fish stocks, enhancing productivity and insuring against fisheries management failure (Kripke & Fujita 1999; Tuck & Possingham 2000; Gerber et al. 2003; Claudet et al. 2008). To achieve these objectives the aspirational goal of marine biodiversity conservation is to conserve the full range of marine biodiversity in marine reserves (no-take areas), from gene pools to populations, species, habitats and ecosystems, and to ensure their long-term persistence (World Resources Institute 1992; Halpern & Warner 2003; Lubchenco et al. 2003; Secretariat of the Convention of Biological Diversity 2004).

International agreements and conventions (e.g. Convention on Biological Diversity) have called for the establishment of a system of MPAs that protects 10-30% of each habitat type in marine reserves by the year 2012 (Kelleher et al. 1995; United Nations 2002a, 2002b; IUCN 2003; Convention on Biological Diversity 2006; Wood & Dragicevic 2007; Laffoley 2008; Wood et al. 2008). Such targets are important in providing guidance and stimulating political leadership for marine reserve network establishment (but see Agardy et al. 2003). Many countries have responded to these commitments by developing conservation policy frameworks to guide the establishment of national and regional networks of MPAs (see for example Mercier & Mondor 1995; Australian and New Zealand Environment and Conservation Council 1999; Anonymous 2000; NSW Fisheries et al. 2000; Department of Conservation & Ministry of Fisheries 2005). There is concern, however, that the lack of coordination and consistent policy frameworks for marine conservation at international, national and regional levels is a problem affecting progress (Tisdell & Broadus 1989; Roff 2005; Wood & Dragicevic 2007). Even where national and regional conservation policy frameworks are in place, the pragmatic implementation of conservation goals has been difficult to achieve because of the complexities with establishing marine reserves. Conservation outcomes are a result of decision-making that is influenced by polarised views and lobbying by stakeholders (see for example Wescott 2006; Klein et al. 2007, 2008).

Increased public interest in the use of MPAs to conserve and manage the marine environment (Halpern & Warner 2003) has led to considerable growth in their use around the world. There are approximately 4600 MPAs established around the world, providing some level of protection to an estimated 0.6% (2.2 million square kilometres) of the world's marine habitats, but only 0.08% (36,000 square kilometres) of this area is no-take (Partnerships for Interdisciplinary Studies of Coastal Oceans 2007; Wood & Dragicevic 2007; Laffoley 2008). These no-take areas are referred to as marine reserves. The existing collection of marine reserves is a result of a fragmented approach to establishment that has generally been based on iconic species or sites (Kenchington & Bleakley 1994; Ward et al. 1999; Lubchenco et al. 2003). This has led to claims that the current size and placement of marine reserves and MPAs falls far short of comprehensive or even adequate to achieve conservation objectives (Boersma & Parrish 1999; Ballantine & Langlois 2008; Wood et al. 2008). Hence, many argue that we need to take a more systematic approach to conservation planning (Mace et al. 2000).

Conservation planning is usually based on surrogates for biodiversity in the absence of comprehensive data on ecosystems, habitats and species (Pressey 2004; Rodrigues & Brooks 2007). Surrogates are biodiversity features used to guide planning with an expectation that their protection will be effective for the conservation of unknown or poorly understood biodiversity (Rodrigues & Brooks 2007). While there remains a lot of uncertainty associated with the use of biodiversity surrogates for conservation planning, many authors believe that significant progress can be made towards establishing networks of marine reserves through their use (Zacharias & Roff 2001a; Sarkar & Margules 2002; Sarkar et al. 2004; Williams et al. 2006; Post 2008). There is also a view that stakeholders and politicians need to accept that a surrogate or suite of surrogates is an effective approach to representing biotic diversity for the purposes of planning a marine reserve system (set of connected marine reserves) (IUCN/WCMC 1994; Halpern et al. 2006; Possingham et al. 2006; Dudley 2008; Post 2008). Sites valuable for their biodiversity need to be identified based on the best information available (Roberts et al. 2003b; Possingham et al. 2006; Beger et al. 2007; Grantham et al. 2008), which is likely to be reliant on surrogate measures of biodiversity.

The design and implementation of a global system of marine reserves is considered to be the next great challenge for marine conservation policy and conservation practitioners (Lubchenco et al.

2003; Young et al. 2007). One reason for this being a significant challenge is because determining where to place marine reserves requires data on the location of marine ecosystems, habitats and species whose distribution is a result of poorly understood ecological processes that are impossible to define precisely, particularly over large geographic areas (Gray 1997; Sarkar et al. 2006; Ballantine & Langlois 2008). The identification of areas suitable for marine reserves requires biodiversity features or their surrogates to be spatially defined (Zacharias & Roff 2000, 2001a; Rodrigues & Brooks 2007). Collecting such information can be expensive, time consuming and often impractical when trying to meet timeframes for establishing reserve networks (Pressey & Ferrier 1995; Schoch & Dethier 1996; Ferrier 1997, 2002; Ward et al. 1999; Gladstone 2002; Pressey 2004; Possingham et al. 2006; Sarkar et al. 2006).

This section discusses how to move from scientific and theoretical approaches for establishing a system of marine reserves to a practical plan for forming a system of marine reserves. The chapter discusses: 1) the role of reserve network goals and criteria for identifying sites for marine reserves; 2) the scale (i.e. fine- and large-scale) at which surrogate measure of biodiversity can be applied and the relative importance of identification criteria in decision-making; and 3) provides guidance on the pragmatic implementation of marine reserve networks.

5.2 MARINE RESERVE NETWORK ESTABLISHMENT: TRANSLATING THEORY INTO PRACTICE

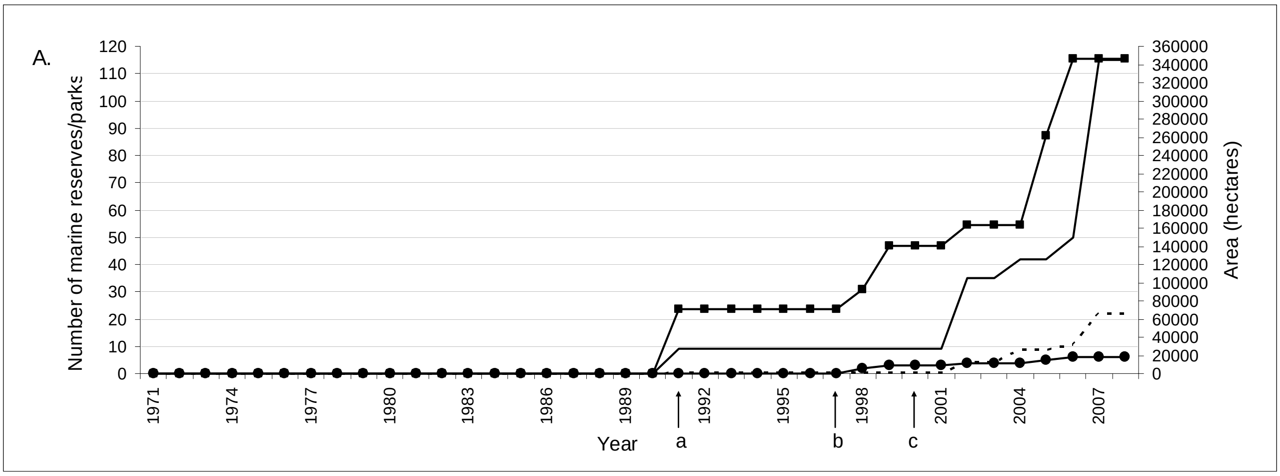
There is growing recognition of a gap (the ‘implementation gap’) between scientific and theoretical approaches to reserve design, and their subsequent implementation (i.e. designation of networks of reserves) (Knight et al. 2008). There are many conservation plans (see for example Possingham et al. 2000; Leslie et al. 2003; Stewart et al. 2003; Banks et al. 2005; Banks & Skilleter 2007; Klein et al. 2007, 2008; Knight et al. 2007, 2008; Leathwick et al. 2008), but achieving a systematically designed marine reserve network in the real world is more challenging. This is because implementation of conservation action must also address social, political and economic complexities of regional and local communities, in addition to the core goals of preserving biodiversity. This section discusses issues associated with implementing marine reserve networks in New South Wales (Australia) and New Zealand. In New South Wales the marine reserve network has been guided by national and regional conservation goals, site identification and selection criteria. New Zealand’s approach until 2006 has been focused on iconic sites resulting in a scatter of marine reserve around the mainland (Ballantine & Langlois 2008). The approach in New South Wales has been to establish large multiple-use marine parks that contain a network of marine reserves. In comparison, New Zealand’s approach has been to establish small individual marine reserves around the mainland to comprise the network. In both cases there have been limitations with spatially defining and mapping biodiversity to support the creation of the marine reserve networks.

5.2.1 Implementing marine reserve network goals

In order to develop networks of marine reserves, many countries have established frameworks for marine conservation policy (see for example Australian and New Zealand Environment and Conservation Council 1999; Anonymous 2000). These frameworks attempt to translate broad political commitments for biodiversity conservation into goals and objectives for marine reserve network design to be implemented at national and regional scales. Conservation goals are typically broad, defining the outcome for the network as a whole. The goal of both New South Wales and New Zealand networks is to establish a comprehensive, adequate and representative system of MPAs that includes the full range of marine biodiversity at ecosystem, habitat and species levels (NSW Fisheries et al. 2000; Department of Conservation & Ministry of Fisheries 2005).

The application of ecological and network design theory that was developed to meet national and regional goals may be difficult to implement at local scales (Lubchenco et al. 2003), but it is at this scale that it is possible to identify the biodiversity features to be protected and the levels of protection that are needed. For example, while conservation goals that seek to protect all levels of biodiversity in marine reserves provide a vision for the network as a whole (see for example Australian and New Zealand Environment and Conservation Council 1999; Anonymous 2000; Department of Conservation & Ministry of Fisheries 2005), they do not provide conservation practitioners or stakeholders direction on the types of biodiversity features to be protected or of the levels of protection that are needed. Further work is needed in order for regional goals to be placed in a local context.

Conservation goals are an important factor in the successful implementation of networks of marine reserves. In 1991, Australia commenced developing a marine conservation program to guide the establishment of a network of MPAs (including marine reserves) (Ray & McCormick-Ray 1992). Following this, New South Wales released a regional scale policy that outlined goals for conservation of marine biodiversity (NSW Fisheries et al. 2000). Prior to completion of the policy, only approximately 710 hectares in nine marine reserves (i.e. no-take sanctuary zones) had been established (Figure 14) (Clayton 1991). Commencement of legislation (i.e. *Marine Parks Act 1997*) and completion of the policy led to the establishment of a further 65,129 hectares of marine reserves. The development of legislation, conservation policy and associated goals has been important in the rapid progress of implementing marine reserves in New South Wales over the last 6 years.



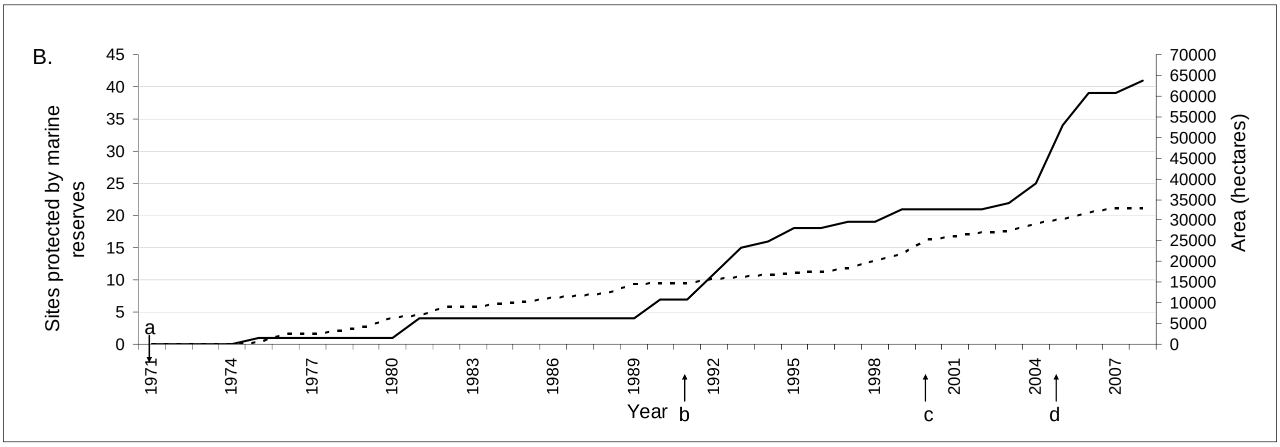


Figure 14: Cumulative growth in marine reserves. **A.** New South Wales (Australia) – number of marine reserves (no-take sanctuary zones) (solid line), total area of marine reserves (dashed line), number of multiple-use marine parks (solid line with dots) and total area of multiple-use marine parks (solid line with boxes). **Note: a.** commencement of coordinated bioregional approach to MPA planning by Australian governments; **b.** *Marine Parks Act 1997* (New South Wales) commenced; and **c.** release of New South Wales's MPA policy (NSW Fisheries et al. 2000). **B.** New Zealand (excluding two large and remote offshore island marine reserves) – number of marine reserves (solid line) and total area of marine reserves (dashed line). **Note: a.** *Marine Reserve Act 1971* (New Zealand) commenced; **b.** commencement of bioregional approach to MPA planning collaboratively with Australia; **c.** New Zealand Biodiversity Strategy released (Anonymous 2000); and **d.** New Zealand's MPA Policy and Implementation plan released (Department of Conservation & Ministry of Fisheries 2005).

In contrast, New Zealand has had a long history (since 1971) of marine reserve establishment. Progress in New Zealand has been continuous since legislation (i.e. *Marine Reserves Act 1971* (New Zealand)) was introduced in 1971 and the creation of Cape Rodney-Okakari Point (Leigh) Marine Reserve (518 hectares) in 1975, but the system is considered to be far from adequate (Walls

1998; Ballantine & Langlois 2008; Shears et al. 2008). After 37 years of implementation there is approximately 32,775 hectares of marine reserve around New Zealand's mainland (Figure 14) and 1,246,000 hectares around remote offshore islands. A conservation policy to guide MPA network establishment has only recently been released (Department of Conservation & Ministry of Fisheries 2005). The policy establishes conservation goals and guidelines for implementation of a network of MPAs with marine reserves as the centrepiece for biodiversity protection. A feature of the policy was the proposal to use a range of legislative tools (e.g. Marine Reserves Act; Fisheries Act) to contribute to New Zealand's target of protecting 10% of the marine environment by 2010 (Anonymous 2000; Department of Conservation and Ministry of Fisheries 2005). Divergent stakeholder views exist on the proportion of this target that needs to be included in no-take marine reserves, compared with protection provided by other legislative tools.

5.2.2 The role of ecological criteria and planning principles

Specific ecological criteria and guidelines have been developed to bridge the gap between national and regional conservation goals and the implementation of marine reserves networks, (Kenchington 1990; Kelleher et al. 1995; Salm & Price 1995; Done & Reichelt 1998; Fernandes et al. 2005). Criteria focused on ecological factors of the marine environment include representativeness, comprehensiveness, ecological importance, naturalness and biogeographic importance (Table 15). These criteria define the ecological factors that should be used to identify locations of ecological or biological importance (Kelleher 1999; Robert et al. 2003a, 2003b), independent of region or political boundaries (Ballantine & Langlois 2008).

There are, however, significant challenges in obtaining information to assess these ecological criteria because they depend on the availability of data on the distribution, abundance and life histories of marine biota or at least on appropriate surrogate measures (Table 15). For example, a conservation practitioner assessing ecological importance of an area may require data about its importance for migration, breeding and feeding for a range of species (Table 15), information which is likely to be difficult to obtain. Similarly, ecological importance may also involve categorising a habitat as unique, which requires information on the extent and distribution of the habitat. It also depends on the scale (e.g. national, regional or local) at which the habitat is to be assessed as unique. Further work is required on the use of ecological criteria when data are absent or limited to enable conservation practitioners to use them effectively in design and implementation of marine reserve networks (Beger et al. 2007).

In an attempt to narrow the ‘implementation gap’, planning principles have been developed to define the ecological and scientific requirements of a reserve network (see for example Day et al. 2002; Fernandes et al. 2005; Queensland Government 2007). These principles have been used in conjunction with ecological criteria (Table 15) with the aim of developing a more ecologically sound marine reserve network. For example, to support the rezoning of the Great Barrier Reef Marine Park, biophysical operational principles were developed to underpin the choice of the number, size and location of marine reserves that were incorporated in the zoning plan (Day et al. 2002; Fernandes et al. 2005). The biophysical operational principles were also supported by socio-economic operational principles, which sought to maximise biodiversity conservation with consideration of detrimental impacts to local communities and stakeholders (Great Barrier Reef Marine Park Authority 2002). The proposed marine reserves were publicly exhibited to provide stakeholders the opportunity to comment on the scale, location and potential impacts of the marine reserve proposals. The development of planning principles appears to have evolved as an alternative to ecological criteria that are often difficult to define or measure in practice. Conservation practitioners have used planning principles to translate ecological criteria into measurable principles that contribute to successful implementation of marine reserve networks.

5.2.3 Defining and mapping biodiversity

To establish networks of marine reserves, the marine landscape needs to be sub-divided into conservation features that can be mapped. Conservation features are most often defined and mapped using surrogate measures, which assume the distribution and abundance of biota is explained at regional and local scales by these surrogates (Margules et al. 1988; Zacharias & Roff 2001b; Banks & Skilleter 2002; Day et al. 2002; Stevens & Connolly 2005). It has been concluded that the true effectiveness of surrogates and their ability to predict biodiversity, between and within regions, will never be achieved (Flather et al. 1997; Possingham et al. 2006). This is because our knowledge of the marine environment is based on patchy and unrepresentative (i.e. in both time and space) information and limited in terms of details on the distribution, abundance and taxonomy of species (Ferrier 1997; Possingham et al. 2006; Harris et al. 2007). This means that network design decisions and site selection is often made in the face of considerable uncertainty (Ludwig et al. 2003). Choice of surrogates should be guided by the presumed effectiveness of the surrogate(s) and based on the availability of data to define the surrogate in a cost-effective way (Possingham et al. 2006). Carefully selected and mapped biodiversity surrogates can assist conservation practitioners to identify sites for marine reserves, particularly where surrogates have the potential to be defined at local scales (10s to 100s of metres) and mapped across bioregional scales (100s to 1000s of kilometres) in a cost-effective way (see for example Banks & Skilleter 2002).

Table 15: Criteria for identification of marine reserve networks and their application using fine- and regional-scale surrogates measures for biotic diversity.

Criteria	Surrogates	
	Fine-scale (10s–100s of metres)	**Regional-scale 10s–100s of km)**
Representative – identify:		
• Representative ecosystem types	Yes	Yes
• Representative habitat types	Yes	Assumed
• Areas that contain the range of known species (ability to predict species distributions)	Assumed	Assumed
• Representative examples of genetic diversity	Assumed	Assumed
Comprehensiveness – identify:		
• Biogeographic extent of ecosystems	Yes	Yes
• Biogeographic extent of habitats	Yes	No
Ecological importance – identify:		
• Unique habitats	Yes	No
• Areas important for spawning or nursery grounds	No	No
• Areas important for migration	No	No
• Areas important for feeding, breeding or rest areas	No	No
• Areas that contain rare, threatened or depleted species	No	No
• Threatened species habitats	Yes	No
• Areas of high species diversity	Assumed	Assumed
• Areas for depleted species and threatened ecological communities	Assumed	Assumed
Naturalness – identify:		
• Areas vulnerable to natural processes	Yes	Yes
• Areas vulnerable to, or protected from human-induced change	Yes	Yes
Biogeographic importance – identify:		
• Rare biogeographic qualities	Yes	No
• Unique or unusual geologic features	Yes	No
Application scale(s)	Site to regional	Regional to provincial
Application of surrogate to conservation planning – identify		
Sites at a local scale (10s to 100s of metres)	Yes	No
Locations at a regional scale (100s to 1000s of kilometres)	Yes	Yes
Replicate sites for habitat conservation within a region	Yes	Assumed
Condition or state as a result of ecological factors	No	No
Condition or state as a result of anthropogenic factors	Yes	No
Cost effectiveness		
Cost of mapping within bioregion(s)	Low-Mod	Low-Mod
Cost of mapping across multiple bioregions	Low-Mod	Low-Mod
Time required to collect information over large geographic areas	Mod	Low-Mod

Source: Kelleher et al. (1995); Australian and New Zealand Environment and Conservation Council (1999); Ward et al. (1999); Zacharias & Roff (2001b); Banks & Skilleter (2002); Day et al. (2002); Fernandes et al. (2005); National Marine Protected Areas Center (2008a).

Maps allow conservation features to be identified, located and described and their relative extent to be determined (Pressey & Bedward 1991; Flather et al. 1997). Knowing the type and extent of conservation features also helps understand how trade-offs amongst stakeholders will affect network implementation (Shafer 1999; Lubchenco et. al 2003; Wescott 2006). Maps also enable stakeholders to gain rapidly an enhanced understanding of the marine environment in which they have an interest, as well as contributing their knowledge in developing data sets to inform decision-making. Conservation features have been defined and mapped using 1) physical properties of the environment (i.e. environmental surrogates) (Dethier 1992; Banks & Skilleter 2002); 2) combinations of oceanographic and physical processes (Zacharias & Howes 1998); and 3) biophysical features that include physical properties and predicted or known distributions of species and other elements of the marine environment (Day et al. 2002; Valesini et al. 2003, 2004; Fernandes et al. 2005; Beger et al. 2007). The quality and extent of information to support mapping is influenced by expense, time and the practicalities to obtain information in order to meet community and political timeframes for establishing marine reserve networks (Pressey 2004). Maps of conservation features also enable the success of implementation of conservation goals to be measured and reported (Pressey & Bedward 1991).

5.3 KEY ELEMENTS FOR SUCCESSFUL IMPLEMENTATION OF MARINE RESERVE NETWORKS

To guide improvements to marine reserve planning and management four broad steps have been defined to support implementation: (1) establish a strategic framework (i.e. defining goals and objectives of the network), (2) systematic conservation assessment (e.g. mapping biodiversity features, identifying gaps, network identification and design), (3) conservation planning (e.g. stakeholder involvement and collaboration, site selection, designation), and (4) management (e.g. compliance, monitoring, ensuring regulation of uses) (Figure 15) (Thackway 1996; Kelleher 1999; NSW Fisheries et al. 2000; Department of Conservation & Ministry of Fisheries 2005; Lundquist & Granek 2005; Knight et al. 2006, 2007; Gilliland & Laffoley 2008). However, despite guidance on steps to implement marine reserve networks, progress continues to be slow and fragmented (Knight et al. 2006; Wescott 2006; Ballantine & Langlois 2008; Wood et al. 2008). This is because implementation of marine reserves generates opposition by stakeholders and local people that might be affected by their establishment and the restrictions placed on user behaviours (Wescott 2006; Jentoft et al. 2007; Klein et al. 2008; Weible 2008). In this section, I discuss network establishment in New South Wales and New Zealand, and argue that there are four essential elements to successful implementation of marine reserve networks: (1) political and agency leadership, (2) dedicated marine conservation legislation, (3) information on natural and social sciences, and (4) processes for stakeholder involvement and collaboration. It is important that these elements are considered before embarking on a marine protection planning process.

5.3.1 Political and agency leadership

Implementation of marine reserve networks requires leadership and commitment at the political level and by the agencies responsible for their establishment (Jentoft et al. 2007; Ehler 2008; Weible 2008). As pressure on marine resources continues, the future of marine reserve network implementation will increasingly depend on a strengthening commitment of governments to protect the oceans and their commons (Agardy 1999), which has developed through countries ratifying commitments to international targets (United Nations 2002a, 2002b; Convention on Biological Diversity 2006). Implementation is often the responsibility of fisheries and/or conservation agencies that either have a primary mandate for fisheries management or terrestrial protected area management, with marine conservation as a secondary priority. A key factor required for success is there must be a willingness amongst these government agencies and decision-makers to protect marine ecosystems, habitats and species (Agardy 1999).

In Australia, the Commonwealth's MPAs program, which was developed in cooperation with State and Territory governments, was a factor that led to an initial increase in marine reserves in New South Wales and development of conservation policy and dedicated marine park legislation (Figure 14) (Australian and New Zealand Environment and Conservation Council 1999; NSW Fisheries et al. 2000; Wescott 2006). The momentum shifted towards establishment of reserves in the early 1990s after Australian governments made commitments to a national representative system of MPAs (Australian and New Zealand Environment and Conservation Council 1999). This demonstrated that where political will and leadership exists progress will be made. The importance of political will and commitment to implementation of marine reserves was further demonstrated, in New South Wales, where political leadership led to the declaration and zoning of two large multiple-use marine parks (i.e. Batemans and Port Stephens-Great Lakes marine parks) in less than 18 months. Prior to this it took on average approximately 4.5 years to develop a zoning plan following declaration of a marine park.

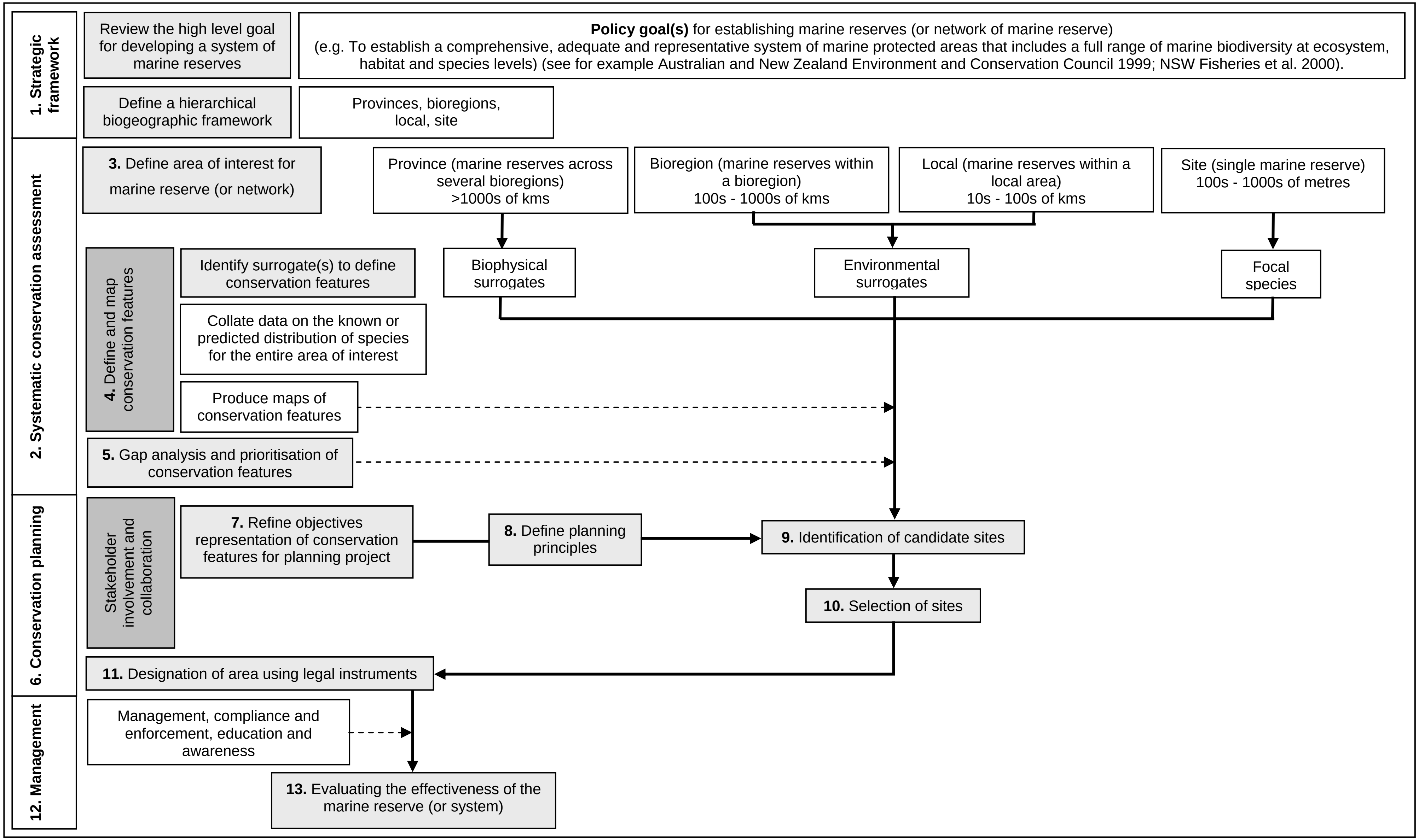


Figure 15: Key steps to identify and select marine reserves.

In contrast, New Zealand has not had a coordinated approach to marine reserve establishment until recently (Department of Conservation & Ministry of Fisheries 2005), despite having the necessary legislation in place since 1971. A factor that contributed to the slow progress in New Zealand has been the view that marine reserve implementation prevents the Ministry of Fisheries from taking action to provide for sustainable utilisation, as required by the *Fisheries Act 1996* (Bess & Rallapudi 2007). Rather than seeing marine reserves as part of ocean sustainability they have been viewed as impeding the potential for utilisation of resources. Thus, the situation exists that the government agency charged with the responsibility to maximise utilisation of marine resources is also asked to protect biodiversity in MPAs, or in the case of marine reserves in New Zealand, must provide concurrence to their establishment. This overlap in jurisdictional authority between and within government agencies is a factor hindering progress in marine reserve establishment (Cocklin et al. 1998). It leads to greater difficulties with implementing a network of marine reserves and is based on fishing (both commercial, recreational and customary) being recognised as the main sector of the community to have a 'right' to the oceans. This must change to enable progress to be made and requires political and agency leadership to implement the necessary changes.

5.3.2 Dedicated marine reserve legislation

Legislation with a primary purpose of protecting marine biodiversity has been developed to protect single no-take marine reserves (see for example *Marine Reserves Act 1971* (New Zealand)) or to establish large multiple-use marine parks that contain a network of marine reserves (see for example *Marine Parks Act 1997* (New South Wales)). There has been considerable debate about the role of single no-take marine reserves versus MPAs that allow multiple-uses in biodiversity conservation (Possingham et al. 2006). There is a view that single no-take marine reserves are unlikely to achieve biodiversity goals alone (Reid 1996; Possingham et al. 2006). The same might be said of poorly designed multiple-use marine parks though, especially where the no-take zones are of insufficient size to contribute to biodiversity protection goals. Implementation of networks using these approaches is either through recognising a collection of single marine reserves (possibly planned as a network) or as a network established by zoning a multiple-use marine park. Both approaches aim to achieve the broad aspirational goal of biodiversity conservation.

5.3.2.1 New Zealand: single no-take marine reserves

New Zealand has established no-take marine reserves (IUCN Category II (Dudley 2008)), using the *Marine Reserves Act 1971* (New Zealand), for the purpose of preserving areas in their natural state for scientific study. The network includes 31 marine reserves (protecting 37 sites) around mainland New Zealand and two large marine reserves surrounding remote offshore islands (Auckland Islands

(498,000 hectares); Kermadec Islands (3 sites protecting 748,000 hectares)). The average size of the marine reserves (37 sites) around mainland New Zealand is 886 hectares ranging in size from 20 to 2,452 hectares. The network consists of a collection of single marine reserves that have been established independently of each other. They protect iconic areas or areas of known scientific interest but have not been designed based on any systematic design criteria or principles.

There is a view that single marine reserves are considered to be rarely of adequate size or scope to be able to achieve conservation of marine biodiversity and there is a critical need to establish representative reserve networks (Possingham et al. 2006). There are, however, research findings that small single marine reserves may be effective in increasing local population size and protecting biodiversity (see for example Babcock et al. 1999; Shears & Babcock 2003; Parsons et al. 2004; Ballantine & Langlois 2008). The assumption used by those promoting reserve networks is that any positive effects from single reserves may be strengthened through a network of marine reserves systematically designed to include representative examples of ecosystems, habitats and species. Further research is required though to investigate the ecological changes resulting from a systematically designed network of marine reserves.

Despite marine reserves only protecting a small fraction of mainland New Zealand (0.2% of the territorial sea) there is a high level of opposition to their establishment from the fishing industry and many recreational fishers (Cocklin et al. 1998; Bess & Rallapudi 2007). In order to develop a more systematic approach to marine biodiversity conservation and to increase stakeholder involvement and collaboration, the New Zealand government released a MPAs policy (Department of Conservation & Ministry of Fisheries 2005). The objective of the policy was to develop a representative network of MPAs. The policy sought a broader approach to biodiversity conservation by recognising that other legislative tools might have a role in protecting some elements of biodiversity (e.g. benthic habitats). This approach establishes different levels of protection comparable to zones in multiple-use marine parks. However, in New Zealand it involves multiple pieces of legislation covering, for example, areas that do not have a biodiversity focus but are closed to some fishing methods using the *Fisheries Act 1996* (Bess & Rallapudi 2007).

The New Zealand MPAs policy proposes the retro-fitting of the *Fisheries Act 1996* (New Zealand) and other legislative tools to biodiversity protection rather than the creation of dedicated legislation that accommodates multiple-uses. Such areas have been referred to as ‘de facto’ or ancillary MPAs (National Marine Protected Areas Center 2008b). There is little known about the effectiveness of these MPAs for protection of biodiversity (but see for example Shears et al. 2006; Lester & Halpern

2008). The use of a range of tools results in inconsistencies in application of legislative obligations for government agencies, which have led to disagreements about implementation of MPAs (Bess & Rallapudi 2007), slowing progress towards achieving conservation goals.

5.3.2.2 New South Wales: multiple-use marine parks

New South Wales has adopted a multiple-use approach to achieve the goals of a representative network of MPAs. Marine parks are established under the *Marine Parks Act 1997* (New South Wales), which provides a network of marine reserves (i.e. sanctuary zones equivalent to IUCN Category II (Dudley 2008)) within a marine park. The multiple-use approach establishes a management regime over a large area (New South Wales marine parks range in size from 22,000 to 97,200 hectares) where biodiversity protection is a primary purpose. The implementation of marine reserves (i.e. sanctuary zones) representative of biotic/abiotic diversity is a core part of multiple-use marine parks.

Six marine parks have been established in New South Wales containing 115 individual marine reserves (i.e. individual sanctuary zones) with an average size of 573 hectares. The size of the marine reserves ranges from 0.01 to 6,580 hectares. Approximately 60 percent of the marine reserves are smaller than 100 hectares and 15 percent are larger than 1000 hectares. It is unknown whether each individual marine reserve will protect marine biodiversity; however, it is assumed that the collection of marine reserves in a network will lead to biodiversity protection (see for example Roberts et al. 2001; Russ et al. 2008). Further work is required to investigate the benefits of such an approach to marine biodiversity conservation (but see for example Butcher et al. 2002).

5.3.3 Spatial information on natural and social features

Information on the natural and social features of an area is essential for implementation of marine reserve networks (see for example Ehler 2008; Gilliland & Laffoley 2008; Klein et al. 2008; Pomeroy & Douvere 2008). Spatial information on the natural features of an area would include, for example, maps of conservation features (e.g. ecosystems and habitats), species' distributions and features or locations important to marine species. This helps stakeholders gain a better understanding of: (1) the complexity and location of conservation features in the marine environment; and (2) the consequences of human influences on ecosystems, habitats and species (Pomeroy & Douvere 2008). Geographic information systems (GIS) are increasingly enabling the presentation of such information in a form that is readily understood by stakeholders and decision-makers. Involvement of stakeholders in deriving this information also provides the opportunity to gain additional data on the distribution of conservation features and areas important for commercial and recreational use (i.e. social features).

Spatial information on the natural and social features of an area, required to support implementation of marine reserves, has been difficult to obtain at local scales (10s to 100s of metres) in both New South Wales and New Zealand. This, however, has not impeded progress in making decisions about selection of areas for marine reserves. Available information has been collated and additional data obtained to assist decision-making. A key feature of both approaches has been an increasing use of GIS to present information on the spatial extent of habitats and the distribution of species to stakeholders and politicians. There remain considerable challenges with obtaining information on the spatial extent of habitats in the marine environment over large geographic areas. However, technological advances in mapping systems (e.g. side scan sonar, multi-beam sonar) are increasingly allowing shallow and deep water habitats to be mapped in a more cost-effective way (Jordan et al. 2005). Further work is required to develop these cost-effective approaches to map the spatial extent of habitats to support marine reserve selection.

Gathering information on social features (e.g. the location and effort of commercial and recreational fishing) at a local-scale is essential to evaluating the potential impacts of marine reserves on users. It also enables the reserve network design to be adjusted to minimise these impacts. In both New South Wales and New Zealand it has not been possible to include local-scale information on commercial fishing because such data are only available for administrative areas defined for fisheries management. These fisheries management areas are usually defined at regional scales (100s to 1000s of kilometres) compared with marine reserves that are implemented at local scales. There is even less known about recreational fishing effort and the locations targeted by these users. In the absence of such information, commercial and recreational fishers are likely to continue to overstate the impacts of even small marine reserves on their activities and income. Describing the fishing effort and location of these activities is a significant challenge for marine reserve practitioners; however, it is also essential for further development and use of decision-support tools in the future. Establishing a requirement for commercial fishers to install vessel monitoring systems and to report accurately the location of their fishing activities will be increasingly essential for marine reserve network implementation. Similarly, developing reporting systems to help understand areas of importance to recreational fishing will assist planning.

5.3.4 Stakeholder involvement and collaboration

Collaboration and involvement of stakeholders is essential when planning the identification and selection of sites for marine reserves (Mize 2006; Gilliland & Laffoley 2008; Klein et al. 2008). The challenge for conservation practitioners is striking a balance between achieving conservation policy goals and providing for access to marine resources. Conservation practitioners have adapted approaches to consultation and planning for the location of marine reserves by providing greater opportunity for stakeholders and local people to contribute to decisions on the location of marine reserves.

The location of marine reserves is as much about social sciences as it is about seeking representation of biodiversity. Implementing marine reserve networks will result in a change to, or restrictions on, behaviours, and such changes are challenged by some stakeholders (Bess & Rallapudi 2007). It is well known that there will generally be polarised views towards marine reserve establishment (see for example Wescott 2006). An impediment to progress has been the debate, often led by a vocal minority opposed to the marine reserves, on placing restrictions on the ‘right’ of access to fishing resources (Cocklin et al. 1998; Bess and Rallapudi 2007). To overcome such barriers, there has been recognition of the importance of collaboration and involvement of stakeholders in selecting areas for marine reserves and also mapping the distribution of different types of fishing (Cocklin et al. 1998; Lundquist & Granek 2005; Wescott 2006; Compas et al. 2007; Gilliland & Laffoley 2008; Pomeroy & Douvere 2008). Despite high levels of involvement and consultation with stakeholders to identify the location of marine reserves in New South Wales and New Zealand there are some stakeholders who will continue to oppose their establishment. Such opposition is something that is unlikely to change despite the efforts of conservation practitioners to provide all information, and involve and collaborate with stakeholders during site selection. Often dissatisfaction with outcomes, and a failure to understand consultative processes, is likely to lead to concerns from some stakeholders about the adequacy of consultation and decision-making (Commonwealth of Australia 2007).

While broad-based involvement of the community is essential to successful implementation of marine reserves, timely decisions on the location of marine reserves are also important. The establishment of marine reserves in New Zealand has followed lengthy and complex discussions. For example, it took 12 years to establish the first marine reserve, Cape Rodney-Okakari Point (Leigh) Marine Reserve (518 hectares), and most recently Taputeranga Marine Reserve took close to 17 years to establish from when it was first mooted (Pande & Gardner 2009). The length of time it has taken to establish marine reserves has resulted from many unhelpful side-tracks (Ballantine &

Langlois 2008), additional consultation (required by the Ministry of Fisheries) with stakeholders, and changing views in communities including a diminishing of support in some cases. Following establishment, marine reserves have been found though to be socially popular and scientifically useful in conservation terms (Ballantine & Langlois 1998). A lack of political commitment and leadership is likely to be a key factor in these lengthy processes to establish marine reserves in New Zealand.

In New South Wales it has taken between 14 months to six years to implement a network of marine reserves (i.e. sanctuary zones) following declaration of a multiple-use marine park. On average, the development of a zoning plan following establishment of a marine park has taken 3.5 years. Implementing a network involves extensive consultation with stakeholders on an advisory committee, which precedes a three month statutory consultation period. During the statutory consultation period, conservation practitioners hold further stakeholder meetings and open days for the general community to gain an understanding of the marine reserve network. There is also extensive media coverage of the proposals for a network of marine reserves.

Despite extensive efforts by conservation practitioners to gain an understanding of the potential impacts of different marine reserve network proposals there is often a minority of stakeholders who do not support any closures to fishing. Opposition to marine reserves can be disguised, by opponents, as requests for delays to their establishment until their effectiveness is proven in the local area or region. Further research on the effectiveness of marine reserves is important. However the need to do this in every part of the world and for every type of ecosystem/habitat is not necessary as the ecological benefits of marine reserves have been demonstrated in many areas (see for example (Babcock et al. 1999; Shears & Babcock 2003; Shears et al. 2006; Pande et al. 2008). Following implementation of the network of marine reserves there is a high level of support from residents and users that live adjacent to the park (Anonymous 2008a, 2008b).

5.4 PROGRESSING IMPLEMENTATION OF MARINE RESERVE NETWORKS

Increasing political commitment to progress establishment of marine reserve networks requires conservation practitioners to build urgently further understanding by politicians of the practical issues associated with implementation. Politicians need to accept that a minority of users will not support any restrictions on their activities no matter how much stakeholder collaboration and consultation occurs, but effective stakeholder participation in marine reserve network design is crucial and can reduce the size of this minority and support political will to designate such networks in the face of objections. Other issues that politicians should understand include the need to make timely decisions on the location of marine reserves and ensuring a separation of fisheries management and conservation in the agency mandated to implement a marine reserve network.

Making final decisions on the location of marine reserves in a timely manner, and following an extensive consultation program is essential to ensure support for their establishment does not diminish. It is essential to help politicians understand that there are likely to be minimal and only short-term political consequences of their decisions. For example the broad support shown by the community to networks of marine reserves in the Jervis Bay and Solitary Islands marine parks (Anonymous 2008a, 2008b) demonstrated that over longer timeframes communities and stakeholders broadly accept that marine reserves are important for biodiversity conservation. To further assist politicians evaluate support or opposition to marine reserve networks additional research on community views following their implementation is needed.

Politicians and agency leaders need to ensure there is a clear separation of fisheries/stock management and conservation responsibilities in decision-making related to the establishment of marine reserve networks. Fisheries management agencies should not have a decision-making role in determining the location of marine reserves where the primary goal of these reserves is biodiversity conservation due to a divergence of goals (Jones 2007). Involvement of fisheries management agencies in marine reserve decision-making leads to confusion over trying to implement conflicting objectives for biodiversity conservation compared with promoting utilisation of fish stocks. A single agency should be mandated to implement marine reserve networks, develop policy and legislation, define and map biodiversity features, engage and collaborate with stakeholders and advise politicians on site selection. A good example of such a model is the Great Barrier Reef Marine Park Authority (the Authority), which is mandated for management of the Great Barrier Reef Marine Park. The Authority has used an ecosystem-based approach to management with a primary purpose of biodiversity conservation (Commonwealth of Australia 2007).

The development of dedicated marine conservation legislation for marine reserve network implementation is more likely to lead to progress than using a collection of legislative tools that do not have biodiversity conservation as a primary purpose. Through this legislation it can be made clear that there is a separation of the biodiversity conservation goals from those of fisheries management, which can then be implemented through an appropriately mandated agency. From the analysis here it appears that legislation for multiple-use marine parks is likely to result in more rapid progress towards achieving conservation goals than legislation for single no-take marine reserves. Conservation practitioners should aim to develop dedicated marine conservation legislation that clearly defines the purpose, consultation process and management arrangements to secure biodiversity protection.

Spatial information on the natural environment and patterns of use by stakeholders will become increasingly important in the future. New technology will enable habitats to be mapped in increasingly cost-effective ways. Further research should be undertaken to develop approaches to broad-scale habitat mapping. Whilst such research is occurring, conservation practitioners should continue to gather and use existing information to build an understanding of the spatial extent of habitats using cost-effective approaches (e.g. new technology, surrogates measures). Completing mapping of the spatial extent of habitats over large geographic areas should not be a factor that delays the decision-making process.

There is an urgent need to develop requirements for commercial fishers to install vessel monitoring systems or similar reporting mechanisms. This would lead to accurate reporting of the location of fishing activities at fine spatial scales. Similarly, developing reporting systems or surveys to help understand areas important for recreational fishing will be increasingly important. In the absence of fine-scale spatial information on commercial and recreational fishing activities there is likely to be a continued over statement of the impacts of a marine reserves on fishing. Knowing the spatial extent of fishing activities would provide evidence for conservation practitioners to assess the potential impacts of marine reserves on these users. This would reduce the reliance on anecdotal evidence provided by fishers themselves, who may be philosophically opposed to marine reserve establishment due to perceived effects on their ‘right’ to access all areas for resource extraction and because of their motivation to maximise potential compensation for perceived displacement. This information will also support the use of decision-support tools in the future. Further research should focus on cost-effective ways to obtain accurate information on the location and effort of commercial and recreational activities. Participative approaches to stakeholder involvement in designing a network of marine reserves also provides a good basis for mapping various commercial and recreational activities.

5.5 CONCLUSION

There is no easy solution to the implementation of marine reserve networks. The marine environment is a common resource that is over-exploited by many parties with little or no accountability for continuing degradation (Gravestock et al. 2008). Polarised views of stakeholders, inconsistencies in legislation and lack of political and agency leadership will mean that implementation of marine reserve networks is likely to continue to be slow and fragmented. At the same time, fisheries can be expected to decline from over-exploitation and failure of management systems (see for example Quentin Grafton et al. 2007), and habitats will continue to be degraded. The future of marine reserve network implementation requires further integration of marine conservation policy, science and decision-making. This requires political commitment and strong agency leadership, dedicated marine conservation legislation and information on the spatial extent and effort of commercial and recreational fishing. This should lead to increased resources to better define conservation features, sound consultation processes to engage stakeholders in site selection, and timely decisions by agencies and politicians. These factors will also assist conservation practitioners to overcome philosophical opposition to marine reserves which should enable more rapid progress on implementation.

CHAPTER 6 — THESIS SUMMARY

This thesis examines a number of issues and problems involving the design of a network of intertidal marine reserves. First, the development of an approach to classification of habitats, which enabled the mapping of 24,216 kilometres of coastline. Second, a series of conservation planning problems were described using data for intertidal habitats. This provided the basis for marine reserve networks to be configured using the mathematical optimisation program MARXAN to examine trade-offs between planning objectives and design constraints. Finally, the issues and problems were expanded into broader implementation issues associated with translating the theory of marine reserve network establishment to a practical plan for implementation.

In this chapter, I summarise the key findings and discuss the implications of this research to marine conservation planning and the protection of intertidal biodiversity. I conclude with some views on future research priorities.

6.1 SUMMARY OF RESEARCH FINDINGS

The research undertaken in this thesis provides a number of important insights for the design and implementation of a network of marine reserves. Despite detailed information being available on the factors structuring intertidal rocky shore assemblages (Underwood 2000), and an ability to rapidly map intertidal habitats, these systems continue to be ignored by conservation practitioners. This research demonstrated the ability to classify and map inter-tidal habitats at fine-scales (10s-100s of metres) over large stretches of coastline (i.e. 24,216 kilometres). The scale at which classification and mapping, and subsequently marine reserve planning is undertaken is important to the success of systematic reserve design. Theoretically the mapping of habitats at fine-scales over such a large stretch of coastline should enable more efficient representation of intertidal habitats in a network of intertidal marine reserves. The use of the mathematical optimisation program MARXAN enables the integration of biodiversity and socio-economic (i.e. costs) objectives for the design of an efficient network of intertidal marine reserves. More broadly, the challenges of implementing marine reserves generally remains constrained by political and agency leadership. This means implementation is likely to continue to be slow and fragmented even for intertidal systems, which can be described and mapped more easily than other habitats.

6.2 DISCUSSION OF RESULTS

6.2.1 Classification of intertidal habitats

In the absence of detailed information relating to biological distributions there has been increasing use of biodiversity surrogates to determine marine reserve priorities at local (10s to 100s of kilometres) and regional levels (100s to 1000s of kilometres). The development of biodiversity surrogates at fine-scales (i.e. habitats (10s to 100s of metres)) will have an increasingly important role in the identification of sites that will contribute to a representative network of marine reserves. Chapter 2 outlines an intertidal classification used to subdivide the coastline of Queensland into alongshore (i.e. lineal) units that described the physical characteristics of all intertidal habitats at low tide. The intertidal habitats defined by the classification were mapped for 24,216 kilometres of Queensland's coastline. The physical properties, which were selected for use in the classification, are known to influence the distribution of intertidal assemblages. This was possible for rocky shore habitats because information and data exist on many aspects of the processes and other factors known to influence species assemblages (see for example Underwood 2000). The mapping of habitats, using the classification, was achieved in a cost-effective way, and enabled the protective status of intertidal habitats along the length of Queensland's coast to be evaluated for the first time.

6.2.2 Marine reserve network design

The benefit of systematic over *ad hoc* planning approaches is shown for the first time for essentially one-dimensional data, using fine-scale (10s to 100s of metres) intertidal habitats to identify a network of marine reserves. Firstly, for intertidal habitats over the entire length of the Queensland coast (Chapter 3), and then mapped with additional data on microhabitats for rocky shores in south-east Queensland (Chapter 4). This enabled an evaluation of marine reserve design to determine whether the finer-scale (i.e. microhabitats) increased the certainty that biotic diversity would be represented in a network.

6.2.2.1 Reserve design for the Queensland coastline

To address intertidal marine reserve design for the Queensland coastline I described a process for the systematic identification of state-wide priorities for the selection of sites for inclusion in a network of marine reserves to achieve protection of a representative example of the full range of intertidal habitats. I evaluated the success of different reserve network scenarios in achieving conservation targets and the potential influences of reserve boundary compactness and the relative cost of each solution in identifying sites to be included in a representative network of marine reserves. The findings of Chapter 3 demonstrated that the design of a network of intertidal marine reserves that meet pre-specified goals can be made more efficient through the use of the mathematical optimisation program MARXAN and a consistent fine-scale (10s to 100s of metres) classification of intertidal habitats.

6.2.2.2 Fine-scale planning – south east Queensland

To further investigate the need to define fine-scale features of intertidal habitats I included in the reserve network design problem additional information on microhabitats (i.e. presence/absence) for all rocky shores in south-east Queeensland. The use of fine-scale intertidal habitats increased the likelihood that reserve networks achieved representation goals for the mosaic of habitats and microhabitats and the associated biotic diversity present on rocky shores than that provided by the existing marine reserve protection. Firstly, the results demonstrated that using broadscale surrogate measures (e.g. rocky shore, sandy beach) for biotic diversity are likely to result in poor representation of fine-scale habitats and microhabitats and therefore intertidal assemblages in marine reserves. The use of finer scale physical data to support marine reserve design is more likely to result in the selection of reserves that achieve representation at habitat and species levels, increasing the likelihood that conservation goals will be achieved.

6.2.3 Translating theory into practice

Marine reserves are important tools that need to be established in a systematic way to support approaches to larger scale oceans management and governance. There are many conservation plans for establishment of reserves (see for example Possingham et al. 2000; Leslie et al. 2003; Stewart et al. 2003; Banks et al. 2005; Banks & Skilleter 2007; Klein et al. 2007, 2008; Knight et al. 2007, 2008; Leathwick et al. 2008), but achieving a systematically designed marine reserve network in the real world is more challenging. Chapter 5 explores two different approaches to implementation of marine reserve networks, and discusses key issues influencing their successful implementation. The findings of the research are relevant to marine reserve network implementation generally and will assist conservation practitioners and scientists understand some of the practical problems associated with establishing networks irrespective of the types of habitats that are the conservation priority. Key factors that influence marine reserve establishment include the need for: (1) political commitment and strong agency leadership; (2) dedicated marine conservation legislation; and (3) information on the spatial extent and effort of commercial and recreational fishing and other users. In the absence of these factors marine reserve establishment will continue to be fragmented and slow.

6.3 FUTURE RESEARCH

The directions that future research could take is summarised in two main parts. The first relates to improvements that could be made to the approach to classification and mapping of habitats and testing the relationship between habitat surrogates and species abundance and distribution. The second deals with more general aspects of conservation planning and intertidal marine reserve network design.

6.3.1 Classification and mapping

Future research on the classification and development of surrogate measures of biodiversity should focus on testing and evaluating the relationship of surrogates to biotic diversity and ecological communities. In an attempt to test the utility of the fine-scale habitats for rocky shores, defined in Chapter 2, I included additional information on the presence/absence of microhabitats. Further research on the biota associated with rocky shore habitats and microhabitats will increase our understanding of the effectiveness of physical surrogates to support marine reserve design. Similarly, the physical properties used to describe other intertidal habitats (e.g. soft sediments) defined by the classification, need to be further tested and examined to ensure that the correct suite of properties used relate to biotic diversity and ecological communities. This research could also focus on approaches to remotely defining biogenic habitats (e.g. mangroves, seagrass) of the intertidal system, which would strengthen conservation practitioners' ability to differentiate the full range intertidal habitats to reflect biotic diversity.

The intertidal habitat classification and mapping could be further developed to investigate and monitor possible changes to the coastline either resulting from further artificial modification or by processes (e.g. sea level rise) associated with climate change. The current mapping took account of across-shore (i.e. low to high tide) changes in the physical properties of the coastline. This would enable an assessment of possible changes in physical habitats present along the coast to be monitored and also used to assess the impacts of different sea-level rise scenarios. For example, with knowledge and data on the across-shore physical properties, shoreline width and tidal range an assessment could be made of the possible loss of soft sediment habitats which may be found to be replaced by artificial structures (i.e. rock walls) that exist on their landward side. While this assumes that other oceanographic processes may not change it could improve decision-making for reserve design under future climate change scenarios.

6.3.2 Conservation planning

The application of systematic methods such as MARXAN, to different conservation planning problems highlights key areas for the ongoing development and future refinement of conservation evaluation procedures. Research conducted in this thesis highlights a number of improvements that would broaden the scope of the approach to design a network of marine reserves representative of intertidal habitats. First undertaking systematic reserve design using smaller scale planning units (e.g. 200 or 500 metre lengths) to evaluate the optimal length of planning unit to achieve biodiversity goals. Second investigating cost-effective approaches to obtain accurate information on the location and cost associated with recreational and commercial use of the marine environment will enable more accurate evaluation of the costs associated with network design scenarios. Finally research should investigate approaches to effectively use the mathematical optimisation program MARXAN in community forums where its use has generally been limited.

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