Aquatic Protected Areas

What works best and how do we know?

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Cairns, Australia - August 2002

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PREFACE

The inaugural World Congress on Aquatic Protected Areas was held in conjunction with the 31st Annual Conference of the Australian Society for Fish Biology in August 2002 in Cairns, Queensland. The Congress was an outstanding success with over 360 delegates from 12 countries represented. The purpose of the Congress was to examine what we know works in selecting, managing and monitoring aquatic protected areas and how we know what works best. The five main themes covered defining the beneficiaries of APAs, design and selection, success factors, measuring performance and what role such areas could play in the aquatic ecosystem.

Such a topic was timely and well received, since Australian natural resources agencies, as well as those in other countries, are currently grappling with these very issues. A measure of this interest was the financial support the Society received from the Fisheries Research and Development Corporation, the Queensland Department of Primary Industries, the CSIRO, the Department of Natural Resources and Environment Victoria, the Great Barrier Reef Marine Park Authority, the New South Wales Marine Park Authority, the Environment Protection Authority Queensland, the Murray Darling Basin Commission, the Third Billfish Symposium, the Australian Fisheries Management Authority and the Northern Prawn Fishery Management Advisory Committee (NORMAC). All of these organizations have a vital interest in the management of APAs.

The central question posed to delegates was what is the evidence that such areas work and how well do they work. Management authorities contemplating their use for management are well aware of the political and social difficulties in proposing and implementing such regimes, as well as in the monitoring of their success. In fact, the Great Barrier Reef Marine Park Authority currently has a new zoning scheme out in the public domain for consideration. Scientists are also well aware of the difficulties in defining what size and types of areas are required and viable, and again, how to effectively measure impacts of such regimes on the greater ecosystem. Interest groups including fishers, conservation groups, tourism professionals and the public also hold a keen interest, as end users, of all of the above.

None of this is surprising when one considers the ever-growing impacts that society is placing on our natural resources. The level of public interest has never been higher. We were fortunate in our keynote presenters - Elliot Norse, Tundi Agardy, Billy Causey, Jon Day and Peter Cullen- who performed superbly, setting the scene in each of their theme areas and really posing the crucial questions upon which successive presenters built. As well, our own scientific superstar, Sir Gustav Nossal of Walter and Eliza Hall Institute of Medical Research fame, gave a most entertaining and informed Congress Dinner presentation. The Society owes a debt of gratitude to our Organising Committee, and in particular Dr John Beumer. They were a great gang to work with and put in many long hours. A special thanks is also due to Rochelle Manderson and OzAccom, the Congress organizers, who were simply superb. The talents and sustained efforts of Diane Mahon in formatting and production have led to this fine publication which we commend as a record of the Congress. A great Congress and a really, solid, scientific contribution to our knowledge of aquatic protected areas.

J. P. Glaister
Congress Chair

D. C. Smith
Program Chair

J. D. Koehn
ASFB President

WHO AND WHAT ARE THE BENEFICIARIES OF AQUATIC PROTECTED AREAS?

Theme 1

KEYNOTE PRESENTATION

WHY MARINE PROTECTED AREAS?

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Abstract

Two key ideas in conservation—biological diversity and marine protected areas—have evolved dramatically in the past two decades. Experience on land and in the sea has shown that the place-based (ecosystem-based) approach to conservation offers some powerful advantages over activity-based and species-based approaches. In the past decade, scientists have realized that the greatest threat to the sea's biodiversity—fishing, most of all fishing methods such as bottom trawling that destroy marine habitats—is one that most existing marine protected areas (MPAs) do not protect against. Australia and the USA became world leaders on MPAs in the 1970s, but opposition from user groups and a lack of political backing for MPAs in the USA means that Australia now has an opportunity to become the uncontested world leader in establishing and managing MPAs. Doing so would have benefits far beyond Australia's borders.

Keywords: biological diversity, marine protected areas, marine reserves, bottom trawling, place-based management, ecosystem-based management

MARINE BIODIVERSITY AND MPAS: ORIGINS OF TWO IDEAS

Although it isn't any longer, it was lonely when I began working on MPAs as a tool for marine conservation in 1978, so I have had the privilege of being both witness and participant for much of its evolution. Rather than writing the usual sort of review paper, I would like to provide some personal reflections on marine protected areas (MPAs), more broadly on the idea of marine biological diversity, and on the reasons for my interest in these.

I did my graduate and postdoctoral studies on the ecology of Caribbean and Tropical East Pacific blue crabs in the genus *Callinectes*, family Portunidae, the swimming crabs. Under normal circumstances I would have become a faculty member and continued my research and teaching, but as luck would have it, that was not to be my life's course. Instead, I began my career in 1978 at the Ocean Programs Branch of the US Environmental Protection Agency, working on the impacts of offshore oil and gas operations on a proposed National Marine Sanctuary (NMS) in the Gulf of Mexico called the Flower Garden Banks. Then, in late 1979 I was invited to become the Staff Ecologist at the President's Council on Environmental Quality (CEQ) in the administration of President Jimmy Carter, a statesman with a strong personal interest in conservation. This was a-once-in-a-lifetime opportunity to do things from the inside that I could never have done from outside the halls of power. While I was at CEQ, I managed to accomplish two things that have had lasting impact. One was a new idea in conservation; the other was to help secure protection for four pieces of undersea real estate.

My first assignment was to write a chapter for CEQ's annual report on a novel, unprecedented and dauntingly broad topic, namely what's happening to life on Earth. At that time, the dominant paradigm in conservation was utilitarian: species were good if you could shoot, hook or saw them, bad if they ate species you could shoot, hook or saw, and not worth noticing if they were neither, a category that includes the vast majority of life (Fig. 1). The astute reader will realize that this is not the most profoundly enlightened conservation ethic. A new concept embodied in the US Endangered Species Act of 1973 was that all species are intrinsically important whether they are useful or not, and that government should intervene on their behalf if it could be shown that there is high risk of extinction. This was a great leap forward, but the problem with this idea is that, by the time a species is shown to be in danger of extinction, human intervention is often too late to save it, and is almost always difficult and expensive.

Fig. 1. Wapiti or American elk (*Cervus elaphus*), Mendocino County, California. Elk exemplify the kinds of species that utilitarian conservationists consider "good" by virtue of the fact that one can either shoot, hook or saw them. Long after endangered species and biodiversity ethics joined utilitarian ethics in shaping conservation practices on land, marine conservation is still shaped mainly by humans' interest in acquiring meat (Elliott A. Norse).

There needed to be a newer, deeper ethic, one that embraced the idea that species are both valuable to humans and intrinsically good, but that is operationally robust, putting conservation in force long before they become endangered whenever possible. Moreover, a new ethic had to go beyond species, reflecting the growing understanding that hierarchical levels of organization above and below the level of species are also critically important to conserve. My personal experience and readings to that point made it clear that the Earth was rapidly losing the diversity of life at three hierarchical levels—the diversity of its genes, species and ecosystems—so my coauthor and I grouped these phenomena as the loss of biological diversity (Norse and McManus 1980). Ironically, completely unbeknownst to us, Tom Lovejoy, another conservation biologist working in the same city, Washington DC, had twice used the same term just months earlier (Lovejoy 1980a, 1980b), but he hadn't defined it, although the context of his two brief mentions made it clear that he meant loss of species diversity. The chapter Roger McManus and I wrote was the first document to define the concept and explore its dimensions, so it seems fair to say that Tom and we share parental pride in the idea of biological diversity. Ed Wilson (Wilson 1988) and many others subsequently brought this idea—often shortened to biodiversity—to the eyes of the public and decision makers worldwide.

Seeing that the appropriate goal of conservation is much more than merely increasing the population of species that we use, even more than protecting species about to disappear, but, rather, is maintaining the diversity and functioning of life, is the most important thing I have ever done or ever will do. To my gratification, this idea has proved to have legs. Maintaining biological diversity has since become the primary focus of conservation worldwide. On land, that is. Thinking about marine biodiversity has lagged terrestrial biodiversity thinking, and it was not until the last decade that there was there a comprehensive examination of marine biodiversity conservation worldwide (Norse 1993). In the sea, the prevailing ethic is still utilitarian, and marine conservation and management are principally concerned with extraction of tonnage from a small fraction of fish species. That seems equally true in Australia and the United States, although there are now changes in the wind in both.

As an American addressing a largely Australian audience, I must admit that I am fascinated by the ways that our peoples relate to the geography and biological diversity of the places where they live. Australia and the USA have some really striking similarities. In the State of Washington, where Marine Conservation Biology Institute (MCBI) is headquartered, we have many towns with names such as Aberdeen, Bellingham, Everett and Kirkland, reflecting the spread of people who came from the British Isles. Australia has Innisfail, Rockhampton, Gladstone and Ipswich. But the dominant cultures in both Australia and America are relative newcomers. In Washington we are reminded of cultures that were established thousands of years before the European invasion by place names such as Hoquiam, Duwamish, Skykomish and Snoqualmie. Similarly, Australia has Dirrinbandi, Cunnamulla, Toowoomba and Oenpelli. Our peoples, newcomers and ancient ones alike, recognized that our lands are comprised of distinct places that are crucial to our nations' identities. The waves of cultural succession indicated in these place names hint at the profound changes in the ways with which Australians and Americans deal with biological diversity.

As Alfred Crosby (1986), Tim Flannery (1994, 2001), Jared Diamond (1997) and others have explained, the people who came to Australia and the USA in the past several centuries quickly built frontier societies whose independent spirit and livestock helped them to subdue the land. If one can see what is missing, there are many signs of our having done so in the form of species now extinct and places now irrevocably altered. Fortunately for our native biota, a few people were observant and wise enough to notice that treating our lands as frontiers had profoundly harmful effects. As a result, America and Australia were among the very first nations to set aside some places as national parks. The USA (Fig. 2) first did so in 1872, Australia in 1879. Other nations followed our example. And in both Australia and the USA, it took about a century after the invention of national parks to recognize the need to protect places in the sea. The USA passed the *National Marine Sanctuaries Act* in 1972; Australia passed the *Great Barrier Reef Marine Park Act* in 1975. As with protected places on land, other nations followed our example.

Fig. 2. Yosemite National Park, California, USA, was established in 1890, making it one of the world's first national parks. Marine protected areas have lagged their terrestrial counterparts by about a century (Elliott A. Norse).

REMEMBERING WHAT UNCLE BEN TOLD US

When it comes to marine life, the USA and Australia have ample reason to assume leadership roles; both countries can legitimately claim marine biodiversity world records. From Guam and Saipan in the West Pacific to Alaska and California in the East Pacific, and from Maine and Florida to the Virgin Islands in the West Atlantic, the USA probably has greater marine ecosystem diversity than any other nation. But Australia is clearly the world champion when it comes to the diversity of species that scientists have described. For example, the USA has 11 species of squirrelfishes and soldierfishes, family Holocentridae, in our Caribbean waters (Robins *et al.* 1986), but the Great Barrier Reef has some 27 (Allen 2000). These are but a tiny fraction of a Great Barrier Reef fish fauna approaching 2000 species, to which we must add the species of Australia's western waters and the extraordinary rich and endemic fish fauna in southern Australian waters. And fishes constitute only a fraction of the species in the seas off Australia. I unashamedly love marine life everywhere on this Earth, but I don't think it is excessive to say that our two lucky countries have an exceptional wealth of marine biodiversity.

Of course, we all need to remember what Uncle Ben taught Peter Parker, aka Spider-Man, and the rest of us: "With great power comes great responsibility". Australia and the USA have undertaken many measures to protect marine life, and, individually and together, have accomplished some impressive things, such as the near-cessation of commercial whaling. But despite these, we are both facing loss of marine biodiversity in our own waters and far beyond. Unless we behave responsibly, intelligently and quickly, we will certainly lose much more of the marine biological wealth we inherited.

PLACE-BASED AND OTHER APPROACHES

This is where the idea of marine protected areas comes to the surface. There are really only three basic approaches to marine conservation. One can focus on activities, an example being the prohibition against oil drilling in the Great Barrier Reef Marine Park (GBRMP). One can focus on species, for example, by setting catch quotas for each fish species. Or one can focus on places or ecosystems, for example, by establishing marine protected areas, which are best defined as places that are managed to protect against at least one kind of threat. Although eliminating one or a few kinds of threats—such as ocean dumping, oil and gas operations or spearfishing—can be a useful tool for conservation, the most effective of the various kinds of MPAs is no-take marine reserves, areas that are fully protected against *all* preventable threats, both things that humans extract and ones that we add to the natural ecosystems.

There are situations in which the activity-based or species-based conservation approaches work best, but the place-based (or ecosystem-based) approach has some really compelling advantages for marine biodiversity conservation. One is that it is based on the realization that places are heterogeneous, that both biological and human communities differ markedly from one place to another. Humans can value places for diverse reasons, including historic, economic, recreational, spiritual, educational, scientific or ecological importance. Places that people consider important to conserve for whatever reason can be protected from threats that are allowed in other places without resorting to "one-size fits all" management. The sea is no more homogeneous than the land.

Another great advantage of the place-based approach is that scientists and managers don't have to know everything about all the components of an ecosystem to be effective in conserving them. For example, we don't have to know habitat needs, age structure, reproductive biology and effects of environmental variations on every one of the thousands of species in an ecosystem, nor the myriad of trophic, symbiotic and other interactions among them. As the number of species increases beyond a few tens, that kind of information quickly becomes prohibitively expensive to gather, and there just aren't enough scientists to do it. Moreover, stock-assessment and ecosystem models are only cartoons, heuristic tools that interpolate or extrapolate information when we don't have it, and greatly oversimplify the behavior of real species and ecosystems. Nature is far too complex for us to understand all that is necessary to "manage" it with the degree of confidence that befits its importance to us. With the place-based approach, however, it is not necessary to micromanage. One needs only to protect enough of the sea to encompass viable, interacting populations that can meet their habitat needs, reproduce successfully, function in their communities, maintain ecosystem services and retain their evolutionary potential to deal with inevitable changes, as they did in the eons before we came upon the scene.

Of course, I recognize that some species are in such deep trouble that they need special help. In the USA, MCBI established a valuable precedent by successfully proposing the listing of white abalone, *Haliotis sorenseni*, under the *Endangered Species Act*. It was the first marine invertebrate ever listed under this crucial law. As a result, US scientists are now moving forward with captive breeding of these endangered gastropods so that we can outplant them in places where we can be confident they won't be fished. Left to their own, white abalone could not recover because they are broadcast spawners whose populations are now so low that they have apparently had no successful recruitment for three decades, thus exhibiting what ecologists call the Allee effect. But on the whole, it is less expensive and more effective to remove the threats from some places and let species recover on their own wherever possible. Three and a half billion years of experience shows that Nature generally knows best, and the best tool for conservation is maintaining or restoring the conditions in which organisms can do what they have been shaped to do on the forge of evolution.

Another advantage of the place-based approach concerns compliance and enforcement (Fig. 3).

Fig. 3. Coast Guard cutter off Haida Gwaii (Queen Charlotte Islands, British Columbia, Canada). Fully protected marine reserves greatly ease the task of enforcement. When enforcement agencies, perhaps based on tips from law-abiding fishermen or data from vessel monitoring systems, find certain kinds of vessels and equipment within marine reserve boundaries, they can presume that someone is violating the law (Elliott A. Norse).

It is much easier to determine whether a person is doing something prohibited if he is physically in a place where society has decided he shouldn't be. If, for example, a trawler loaded with prawns is caught inside a no-trawling zone, the captain has some serious explaining to do. Indeed, the development of vessel monitoring systems (VMS) allows enforcement agencies to ensure that our society does not have to rely solely on fishery observers or self-reporting by people at sea. Clearly, boats might have a need for innocent passage across MPAs. But one can easily imagine a device on a trawler like a flight-data recorder that simply signals to a satellite whether the net is stored or deployed, along with the trawler's coordinates. If a trawl net is deployed in a place where trawling is prohibited, that constitutes compelling evidence that somebody is doing something he shouldn't.

I must also point out that MPAs aren't a panacea. They won't provide much protection against the effects of global warming, although I suspect that scientists such as Terry Done can provide some useful suggestions for reducing impacts. MPAs cannot stop the entry of non-native organisms that have been introduced from other areas of the world in ships' ballast tanks and by other means, although scientists such as Nic Bax might offer some useful observations on invasibility of intact versus disturbed ecosystems. And for MPAs adjacent to or near the land, the greatest threat may be land-based human activities such as logging, agriculture or urbanization. However, most MPAs—with rare exceptions such as the GBRMP—lack the legal authority to prevent damaging uses of adjacent lands, which can profoundly affect the sea. Of course, those legal authorities aren't of much use if authorities don't have the courage to use them. In Queensland and Florida alike, nutrient runoff from sugar plantations on land is a major threat to coral reefs, and requires continued attention and creative thinking from our political leaders.

LOST MOMENTUM AND NEW UNDERSTANDING ABOUT THREATS

One difference between the USA and Australia is our record on marine protected areas. It is a bit embarrassing for me to admit this, but Australia, a nation with just 1/15th the population of the USA, has done a better job of establishing MPAs. Many of my colleagues in the USA and elsewhere see Australia as the world leader on MPAs. But there are things both of our nations really need to improve. I would like, first, to discuss the USA experience, because that is what I know best, and then I will be cheeky enough to offer some thoughts about ways that I think Australia can continue its leadership on MPAs.

Although the USA has had a national MPA program for three decades now, this program has been plagued by lack of vision, timidity and interference by politicians who are beholden to fishermen and other user groups. In those 30 years, America has managed to establish just 13 National Marine Sanctuaries. There were only two in 1978, when I got my first job in conservation, which was focused on establishing a NMS in the Gulf of Mexico. Although my efforts did not achieve much short-term success, it was a good, if painful, learning experience. But in 1980, when I worked for President Carter, I helped the birth of four others, at that time bringing the total to six. The pride I once felt at having helped establish these four MPAs has dimmed, however, as scientists, conservationists and managers have learned more, because in the USA, Sanctuaries are sanctuaries more in name than in fact. Most are too small, and, together, they are too few to sustain populations of many species that have larval dispersal. The largest sanctuaries are only a tiny fraction the size of the Great Barrier Reef World Heritage Area, but their scant size and numbers are only part of the problem. An even greater weakness is that NMSs in the USA provide very little protection from the greatest threat to marine biodiversity.

Although oil platforms, spilled oil and fouled seabirds are highly visible indicators of the harm humans are doing in the sea, oil pales in comparison with another threat to marine biodiversity: fishing (Jackson *et al.* 2001). It has not escaped my notice that my family, my staff and I are very much part of the reason this is so. We all eat some kinds of commercially caught seafood and a number of us are recreational fishermen (Fig. 4); we are anything but anti-fishing. We understand fully that seafood is an important, even crucial component in the diets of many peoples around the world, and both commercial and recreational fishing are economically significant activities in many localities. MCBI is not saying that people should stop fishing.

Fig. 4. The author as a graduate student in 1971, having caught a spotted cabrilla (*Epinephelus analogus*), Pacific Coast, Baja California Sur, Mexico. Recreational and commercial fishing are important economic activities, but increasing evidence also indicates that fishing is the leading threat to marine biodiversity worldwide (Richard Huddleston).

But it is now unmistakably clear that, around the USA and around the world, more marine biodiversity loss is due to fishing than to any other cause. Moreover, commercial and recreational fishing interests in the USA are generally opposed to protecting places in the sea. They have been very successful in lobbying the US Congress and Bush Administration officials to prevent the establishment of fully protected marine reserves. As a result, the USA has a very few of them, the newest—thanks to the dedicated leadership of Billy Causey—being portions of the

Florida Keys NMS. But they are nowhere near what we need to conserve the full range of America's marine biodiversity, and the future is worrisome because opposition from fishermen is hardening.

This is how weak our NMSs are: one of them—Hawaiian Humpback Whale NMS—offers no protection within its boundaries that does not already occur outside them. Only a microscopic portion of our NMS system is managed as fully protected no-take marine reserves. Indeed, even the world's most destructive fishing methods—trawling and dredging—are allowed in most sanctuaries.

I have been contemplating trawling for a long time; I spent my first night on a trawler in the Sea of Cortez, Mexico, as a Ph.D. student in 1971. I was studying the ecology of blue crabs, and decided to hitch a ride on a trawler because I had been told that they caught great numbers of these crabs, particularly *Callinectes arcuatus*. As it turned out, about 95% of the biomass in the trawls I observed were not the targeted shrimp species, but were *C. arcuatus* and other portunid, calappid and majid brachyuran crabs, hermit crabs, stomatopods, starfishes, elasmobranchs and bony fishes (Fig. 5).

Fig. 5. "Trawl trash" on trawler deck, Gulf of California, Sonora, Mexico. 95% of shrimp or prawn trawl catches can consist of non-target species that die and are shoveled overboard. This greatly underestimates the damage, however, as trawling crushes, buries or exposes many more organisms on the seafloor that do not come up in the net. Marine protected areas that prohibit trawling are a crucial tool for maintaining the diversity and integrity of marine ecosystems (Elliott A. Norse).

These animals comprise what fishermen in the USA call "trawl trash," and what I later called biological diversity. Moreover, this and other experiences on trawlers sowed another seed in my mind: the question, "What happens to all the organisms that don't come up in the nets?"

In 1990, when I was Chief Scientist of a nongovernmental organization called the Center for Marine Conservation, a conversion about impacts of shrimping with CMC's Fishery Biologist Harry Upton led to a back-of-the-envelope calculation that the Texas and Louisiana trawling fleets for the brown shrimp (*Farfantepenaeus aztecus*) swept the brown shrimping grounds an average of 300% per year. We were astounded to think that large areas of the seafloor could be disturbed with an average return interval of only four months, and I began looking through the scientific literature for studies on the impacts of trawling. Published information was very sparse, and in whole regions of the world—including the waters of Asia, Africa and the US Gulf of Mexico—I could find no published studies at all.

In 1996, after I founded MCBI, the first thing I did was to hold a scientific workshop on effects of trawling on the world's marine ecosystems in Maine, USA. It was the first such workshop to integrate this information from around the world. It included outstanding scientists from Australia, New Zealand, the UK, Canada and the USA. From it we published seven papers in *Conservation Biology* (e.g. Watling and Norse 1998) and another (Norse and Watling 1999) in a book published by the American Fisheries Society. Since that time we, and a growing number of other scientists, have learned even more about effects of trawling. Indeed, earlier this year, the National Research Council of the US National Academy of Sciences (2002) issued a report on impacts of trawling and dredging.

Trawling for prawns or demersal fishes and dredging for scallops on the seafloor are not unlike catching kangaroos with bulldozers. They crush, bury, expose species to scavengers and remove structure-forming species, including stony corals, gorgonians, sponges, bryozoans, tubiculous polychaetes and amphipods, as well as a host of others. Some of these animals are large and visible; others extend only a centimeter or two above the substratum. But by smashing them, trawling dramatically reduces structural complexity on the seafloor, and eliminates feeding and hiding places crucial to the young and adult stages of many species, including commercially important fishes (Sainsbury 1987). A single pass of a trawl or scallop dredge can remove anything from a few percent to 76% or more of macrobenthic organisms (National Research

Council 2002). The effects of even one pass can last months, years, decades or centuries. Repeated trawls increase harmful effects on species abundance, community composition and benthic productivity. The overwhelming preponderance of evidence compels me to conclude that trawling and dredging are incompatible with maintaining biological diversity in areas that people want to be protected. These are the kinds of threat that highlight the need for MPAs.

The most encouraging development that has happened for MPAs in the USA grew out of a scientific workshop that MCBI held in early 2000 in partnership with The Cousteau Society. The participating scientists (from Australia, the Philippines, UK, Canada and USA) called upon then-President Clinton to issue an executive order establishing a comprehensive national system of MPAs that would fully protect 20% of US waters by 2015. That would have meant no trawling or other fishing, or other kinds of preventable harm. The Clinton Administration was interested, and we negotiated for months with Administration officials. But by the time President Clinton finally did issue the Executive Order in mid 2000, it had been weakened in many ways. And since George Bush became President in 2001, government progress on marine protected areas has ground virtually to a halt. Until there is either a profound and almost unimaginable change-of-heart or else a new administration, the USA is not going to be setting an example worldwide on MPAs. We have dropped the ball.

REGAINING LEADERSHIP

What does this have to do with Australia? Well, stimulated largely by concern about the prospect of oil and gas drilling, Australia immediately became the world leader in MPAs when it established the GBRMP nearly three decades ago. Scientists, conservationists and managers have learned a lot since then, including the fact that commercial and recreational fishing is an even greater threat to marine biodiversity than oil drilling. It stands to reason that the GBRMP and other MPAs in Australia should reflect this new understanding, as should the National Marine Sanctuary system in the USA. Yet Australia, like the USA, still allows a broad range of commercial and recreational fishing in most of its MPAs, including bottom trawling. Not all; some coral-covered seamounts off Tasmania are still intact because they were too deep to trawl and are now protected from all fishing except for pelagic species. But as CSIRO's Tony Koslow explained at the *First International Symposium on Deep Sea Corals* in Halifax, Nova Scotia, in 2000, trawling has had a profound impact on coral communities of the shallower seamounts.

Australia now protects less than 5% of the Great Barrier Reef from fishing and only about 50% from trawling (WWF-Australia 2002). Even areas that are officially protected are illegally trawled (Poiner *et al.* 1998). So if my Australian colleagues, who invited me to address the First World Congress on Aquatic Protected Areas, are open to hearing the view of an outsider, I will say this: knowing the profound impact of trawling and other methods of fishing in our countries on marine biodiversity and the fact that the world's scientists are increasingly calling for more fully protected no-take marine reserves, something has to change. Australians are the stewards of what many people consider the world's best piece of underwater real estate. That is why the United Nations designated it as a World Heritage Area. If Australia wants to continue being the world leader in marine protected areas, you need to protect a much larger portion of the Great Barrier Reef World Heritage Area from trawling and other methods of fishing. Indeed, to show real leadership, Australia needs to fully protect a sizeable fraction of the waters around the continent and in the waters of Australia's island territories. It doesn't make a lot of sense for Australia to allow the activity that is most destructive to marine biodiversity in the world's most diverse marine ecosystem.

Of course, conservation of your incredibly rich marine biodiversity will benefit Australians today and tomorrow. That is what Americans call a "no-brainer." But there are two other reasons why you might consider dramatically expanding your system of fully protected no-take reserves. The first is that the world's most diverse area of ocean is the so-called "coral triangle" that includes the northern portion of the Great Barrier Reef. And this triangle is the heart of the world's richest shallow-water marine biogeographic region, the Indo–West Pacific. The other countries in the Indo–West Pacific are nearly all poor nations. And poor people all around the world catch seafood any way they can, even if that means using explosives and cyanide. Poverty makes people do desperate things, so the prognosis for marine life in Australia's poorer neighbors is not good.

I know that the world's economy, including Australia's, is not exactly thriving right now. But no other nation in the Indo–West Pacific region has the knowledge or the wealth to be able to protect, recover and sustainably use marine life that Australia does. Australia stands out as the very best hope for the myriad species of the Indo–West Pacific, from highly visible ones such as dugongs (*Dugong dugon*), humpback whales

(*Megaptera novaeangliae*), Napoleon wrasse (*Cheilinus undulatus*) and *Acropora* corals to the myriad small invertebrates that have not yet even been described. If anyone is going to conserve the miraculous diversity of marine life in this vast region, both the individual components and the web of connections among them, it is going to be Australia. Moreover, given the rate of marine biodiversity loss, it will have to happen in this generation. It is unfair and unrealistic to expect nations with far lower per capita gross domestic products and far fewer scientists, such as Tanzania, India, Papua New Guinea, Indonesia, the Philippines or the Federated States of Micronesia, to take the lead in this.

The second reason is that, if Australia takes the bold but scientifically justified step of fully protecting marine life in a much larger portion of your waters, you will have impact far beyond Australia's seas. In the USA, when advocates for marine protected areas see that Australia has, once again, leaped beyond the USA in protecting its wealth of marine life, they will hold Australia up as a shining example of a nation that does it right. Decades of observing our political system leads me to suspect that American politicians will rise to the bait. Being proud of their country, and more than a little competitive, they won't like the idea of America's not being the best at something. So, the biggest impact that Australia could have by creating a national system of fully protected no-take marine reserves might actually happen in the USA, Canada, New Zealand, Italy, Mexico and other nations showing increasing interest in conserving marine life. Who knows: perhaps even Japan, Taiwan, China and Spain—nations that relate to marine biodiversity mainly on a plate—might follow Australia's example. That is what leadership is all about.

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As a non-Australian, I feel deeply honored to have been invited to address the importance of marine protected areas in the opening plenary session of the First World Congress on Aquatic Protected Areas, and I congratulate the organizers for assembling such an informative and thought-provoking program. I thank Jane Lubchenco and her colleagues in the National Center for Ecological Analysis and Synthesis (NCEAS), Partnership for Interdisciplinary Studies of Coastal Oceans (PISCO) and Communication Partnership for Science and the Sea (COMPASS) for generating so much of the information about the utility of marine protected areas. I am also indebted to my colleagues Jim Bohnsack, Callum Roberts, Lance Morgan, Peter Auster, Rod Fujita, Larry Crowder and Michael Weber for teaching me about the biology and politics of MPAs. I thank Graeme Kelleher, former Chair of the Great Barrier Reef Marine Park Authority and Jon Day of the GBRMPA staff, Keith Sainsbury of CSIRO, Trevor Ward of University of Western Australia and Imogen Zethoven and Doug Beattie of WWF-Australia, for helping me understand some of the finer points about MPAs in Australia. Of course, as a nonprofit nongovernmental organization, everything MCBI does to advance understanding of MPAs depends on the generosity of the funders who care enough about the sea to give us resources we need to do this work, including the David and Lucile Packard Foundation, Homeland Foundation, J.M. Kaplan Fund, Moore Family Foundation, Edwards Mother Earth Foundation, Curtis and Edith Munson Foundation, Henry Foundation, Pew Charitable Trusts, Bullitt Foundation, William H. and Mattie Wattis Harris Foundation, Horizons Foundation, Park Foundation, Sandler Family Supporting Foundation, Russell Family Foundation, an anonymous foundation, Benjamin Hammett, Sally Brown, Bertram Cohn, Mark and Sharon Bloome, and Jennifer and Ted Stanley. Finally, I thank the love of my life, my wife Irene, who puts up with my many absences and my severe jetlag in-between so that I might learn and share my understanding about MPAs with old and new friends around the world.

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BLAME MY GRANDMOTHER – SHE SAID IT WAS OK!

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Abstract

For generations Australians have been taught to fish by someone close to them – usually Dad, an uncle or friend of the family. However, it was my Grandmother who taught me and, in the process of being taught, I was told to respect the fish and the environment that they lived in. But never was there the concept that the very area where she taught me would be locked up and declared a no-take zone. It was always going to be an area that was special to me, and still is today, and one I continuously go back to, to fish and reflect on the past. Today, however, many areas like this are being considered or have already been declared protected areas – much to the amazement of many people who have fished there for years. You ask what is wrong with this? I hope that this paper will perhaps shed some light on the views and feelings of the everyday angler and why they may well not understand or believe in the concept of aquatic protected areas.

Keywords: rehabilitation, consultation, solution, stakeholders and social

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INTRODUCTION

Anglers are a simple mob. They really only want to go fishing. For years, they have provided a source of extra food for the family, but today that has for the most part changed and continues to change quite rapidly.

I can see the arguments for and against these things known as protected areas. As President of Recfish Australia I could easily take the line that I am representing anglers around Australia and it is it my role to protect their fishing interests by opposing these 'lockouts.' I call them 'lockouts' as that is how many recreational fishers see them.

But, with a little persuasion, I could just as easily argue that these protected areas will in fact, in the long term, lead to better fishing – so I am told. The spill-over benefits widely promoted by advocates of protected areas as positive outcomes still require a great deal of work to convince the anglers.

RECREATIONAL FISHING AND MARINE PROTECTED AREAS

So why do anglers or recreational fishers or game fishers or bottom bouncers or whatever name you use kick up such a fuss about the idea of protected areas? I hope that I will perhaps be able to shed some light on this ubiquitous question and to look at this whole issue in total rather than isolating it to simply locking up an area. My views are based not on science, not on management, but on a passion for a unique and fascinating pastime.

The anger displayed by anglers after the declaration or concept of a Marine Protected Area (MPA) is announced can in part be attributed to an ignorance factor and is often also associated with a lack of consultation. This leads to some sectors within the recreational fishing industry throwing their hands in the air and saying 'one out, all out'! Or digging their heels in and saying unless there is compelling biological evidence to support the establishment of these areas we are not interested and will not support them.

It is very interesting to listen to a group of anglers discussing the logic behind an area being declared or given some status of protection. Without the knowledge of the reasoning for the decision there soon becomes a rapidly circulating rumour that the greenies and scientists are in bed with each other simply to have an area set aside for their exclusive use as a dedicated playground without the resource extractors being allowed access. From this you end up with a festering lesion that unless addressed will lead to a yawning gap between the so called protectionists and the anglers.

What then happens is a stubbornness and "pig-headedness" to avoid and resist any moves to look at or assess the possibility of reserves. The gap gets wider and the impasse increases.

The anglers ask why must the bureaucrats of this world continually put these so-called saviours of the marine resources right in our face or in our backyard where we have fished for years? They question the need to have them right there. Do they work? Why not have them in areas where they can be tested without interfering with our fishing? Consultation and involvement by the anglers will, in most cases, lead to cooperation from the angling community. Once ownership and greater understanding are established within the recreational fishing sector I am confident that progress can be made. This is a far better approach than the option of dictatorial/ autocratic announcements.

Put us in the defensive mode and we will bite – engage and consult us and we will assist to protect and enhance the very resource from which we derive so much pleasure. The ball is firmly in the hands of everyone who wants to see the resources not only survive but in fact thrive. It does not help when politicians, preservationists, bureaucrats and others put in unrealistic and tight deadlines for no other purpose than political point scoring, greed or 'cover your behind' reasons.

Is locking up an area and excluding the recreational sector the only answer? Not only is the ethos of the angler changing but their need to experience the angling world for the fun and escape from the so-called fast lane is fast becoming the 'norm'. Supplementing the family's food supply is diminishing as a reason people go fishing. With 29–59 % of the catch of the angling community in Australia released for a variety of reasons, we need to assess this changing culture and determine how it can interact with the need of future generations in mind.

So what about options – catch and release, special limits, seasonal openings, etc. With so many management tools at our disposal we should be able to come up with an array of fishing options for areas that are designated as in need of special care whilst still accommodating the needs of the angling community. We don't have to keep everyone out with the attitude of lock it up and leave. The alternatives include a catch-and-release fishery only (Jones 2001). This way the impact of concentrating more anglers in less water through lockouts can also be considered in the overall management of the fisheries. Converting 10% or 20% of the available fishing grounds to protected areas will greatly increase the pressure on the remaining grounds. This will only lead to concentrated effort, overfishing and increased conflict.

I thought the following comment from the MPA News (2000/2001) enlightening:

> *"Indeed it is rather surprising that the fairly abysmal performance of MPAs has been the basis for a global movement towards marine reserves for fisheries management. Current estimates place the number of 'paper parks' at over 80 – 90% in some countries, and rich countries have fared no better than poor ones. Rather than charging ahead to create hundreds of new MPAs, it makes sense to determine (1) whether or not a no-take marine reserve is the best management strategy for a particular fishery, and (2) how we can better implement and manage current MPAs so they reach their stated objectives."*

Simply declaring an MPA as a means of protecting the fish stocks within the area is not the sole way of sustaining the stocks – in fact, I would argue that we can do a lot more for fish resources by actually looking outside the area in question and looking at all of the other impacts that are evident.

As I said two years ago, at a workshop on Ecologically Sustainable Development (Harrison 2000), now is the time, more than ever, for the recreational and sport fishing industry to stand up and be counted. The long-term future for fishing lies in habitat, nurseries, and unpolluted and unobstructed waterways. Habitat and nurseries have been, and still are, being destroyed through a culture of "develop at all costs". Admittedly, there is an awakening amongst the governments and planning authorities that the wetlands and habitats affected do play a very important role.

But we have an enormous challenge to reverse the damage that has already been done. Decade after decade we have seen critical areas of fish habitat drained and dammed, weirs built, canal estates developed, agricultural impact, industrial runoff, acid soils, and the list goes on.

Let's look at one example: - in 1928 a weir was constructed on Sportsman's Creek, a tributary of the Clarence River in northern New South Wales. This was for the purposes of providing fresh water for stock upstream of the weir. No thought was given to the impact on the aquatic resources, because the thinking in that day and age meant that fish did not come into the equation. With a catchment of 285,000 hectares, the permanent swamp of 3000 hectares was known as the Everlasting Swamp and provided a year-round haven for aquatic resources. That system virtually disappeared when the weir was built. Before the weir was built, commercial fishermen netted parts of the swamp – areas commonly known as the "lakes" – for all sorts of fish from mullet to flathead to mulloway.

Sportman's Creek was also known as a prolific prawn nursery and generated a high percentage of the prawn stocks for the Clarence River. It was an excellent nursery and, with the vast area of wetlands, it was the 'lungs and heart' of the creek and river system for aquatic life.

Today, with the original purpose of the weir obsolete because reticulated town water is available, there is a pressing need to have the weir removed and return the 3000 hectares to its original and natural purpose – a nursery and wetland sanctuary for fish, prawns and the like. There are hundreds of examples of these types of obstructions around the country.

More and more of the locals are saying that the fishing is not like it used to be. Simply declaring an aquatic protected area around Sportsman's Creek with the intention of protecting and increasing the fish stocks will not work; the management solutions need to be much broader.

We still have some people within government and private enterprise who believe wetlands can be developed with so called "minimal" impact on fisheries. "Let's use this barren useless land and have some economic return instead of seeing it lying idle and being wasted" is a common catch-phrase among many so-called developers. It is seen as cheap land and ripe for exploitation.

It all starts at the very highest physical point in the country– what we pour onto the ground or dig up will somehow impact on something somewhere downstream. You only have to look at the effects of the cyanide spill on the Tiza and Danube rivers to see what I mean. We must make sure that what we do from now on, anywhere on land, is challenged and the possible consequences for the marine and freshwater environment are realised and taken into account before proceeding. This is not an easy task with about 700 local councils, eight State governments and one Federal government all wanting to move forward on the so-called "development" of Australia.

Long-term sustainability of our fisheries resources is inextricably linked to provision of an environment where the fish can live, breed and thrive. If there are no fish to argue over, we can all go to the pub and drown our sorrows. No habitat and no nursery **does** mean no fish!

The drive to introduce aquatic protected areas or no-take zones or marine parks – call them what you like – is not going to go away. For some time many people have been saying if there is a clear need, based on scientific or biological evidence, then maybe we will agree to areas being set aside with a range of take options.

The time has come when anglers are beginning to realise and acknowledge that there are more than just extractive user groups interested in the water and its contents. Passive users also like to know that there will be marine life for future generations. We have to accept that these protected areas will be introduced. What we have to do is to be part of the process that decides where they go and how they are managed. Being involved in the decision circle is a must. Sticking our heads in the sand only exposes a target!

But let's look at this as a complete picture. If there is a need to have a protected area, ask the question why? What has caused this, why do we need to take the steps to protect the aquatic resources? What has the human race done to affect this? All these need to be asked to assess the reason and to ensure that we can do something in the long term to rectify it.

If these questions are asked, a vast majority of the answers will lie in what has happened upstream and in fact started on dry land. Primary industry, urbanisation and industrialisation have had profound effects on the wet wobbly bits.

Again from an article in MPA news (2002) discussing a new atlas of the world's coral reefs:

"Unfortunately, many protected areas exist on paper only – they are poorly managed and have little or no support or enforcement. Equally worrying is that in almost every single case, protected areas are aimed solely at controlling the direct impacts of humans on coral reefs. Fishing and tourist activities may be controlled, but the more remote sources of threats to reefs, notably pollution and sedimentation from adjacent land continue unabated. Without a more concerted effort to control all of the impacts of humans on coral reefs, even the best managed marine protected areas may be managed in vain."

The measures being put in place or suggested for protected areas are not actually addressing the cause. Treating the symptoms and not addressing the cause will get us nowhere in the long term.

So when we think about a wet protected area let's also look at the upstream impacts and initiate some changes to processes that, in many cases, have been going on for decades. To address this, the 1995 National Policy for Recreational Fishing states:

> *"That all levels of government should initiate urgent action to ensure the conservation of critical habitats for wild fish. Such action should include legislative protection for spawning and nursery grounds; increased research on the ecological and economic functions and significance of these areas; and steps to restore habitats and ameliorate existing impacts."*

Australia as a country has done little in the way of addressing this policy. We are seemingly moving at a rapid pace to introduce protective legislation in the form of protected areas but we are not looking at the complete picture. If a MPA is to be introduced to supposedly protect the aquatic resources, then it is essential to look upstream and see what rehabilitation and remediation work is needed to complement the 'fix' that is supposedly required.

Australia can, as a nation, sustain the fisheries resources in and around the coastline by being smart. All available tools need to be used – not just one.

Ideally any protected area should be 'bottom up' driven rather than 'top down' and should be a partnership with all stakeholders. Explore the knowledge and understanding of the local people of an area and ask them for their input. In many cases they have lived, worked and played in the area and have an intimate understanding of its workings. In particular, the traditional owners have successfully managed to live in harmony with the land and sea without inflicting the vast changes that have happened since 1788. Use that source of wisdom and make sure that we understand what we are trying to do. Clearly identified goals, objectives and expectations with characteristics that are appropriate, strategic, timely, reasonable and measurable are essential for any proposed protected area. We need to look at these from a quadruple perspective: economic, environmental, social and cultural.

As a final comment I would like to bring in an issue that perhaps has not been considered in the deliberations over MPAs (Schipps 2000). Many anglers have fished in the same area for decades and have handed down to their children some fundamental values: to cherish recreational fishing along with a philosophy of being caretakers and custodians of the resources and habitat from which they derive so much pleasure.

Is this social or cultural heritage a right? Let's assume it is. What level of compensation is due to these anglers who are forced to move to another location to continue to pursue their hobby, recreation or sport of fishing because an area is declared a no-take zone? I suggest that this will become a really hot issue both politically and socially.

Who will be the first recreational angler to stand up and challenge the creation of a MPA and seek compensation from the proponents? Their arguments may well stand in a court. With such a void in research into the social value of recreational fishing it may well mean that a judge or decision maker will lean towards the angler.

To recap, I have three messages:

1. engage and consult with all stakeholders at the very beginning and use the bank of knowledge in the community;
2. do not be myopic in looking for a solution to diminishing aquatic resources, because the cause is likely to be a long way from the box that you wish to draw on a map; and
3. look for a total solution incorporating remediation works on wetlands/ nurseries that have been destroyed by past practices.

Today, anglers from my generation are but one group of anglers that were introduced to the sport of fishing by a wide range of people. In my case I can blame my current lack of finances on my therapeutic need for angling gear, and my passion for the sport on my maternal grandmother – God rest her soul – she has a lot to answer for! For it was she who told me to enjoy fishing and never lose the passion because it can give vast pleasure and an inner peace within oneself, and most importantly you can fish **ANYWHERE**. Little did she realise what would be dictating the management of the fisheries 35 years on.

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MARINE PROTECTED AREAS AND FISHING CLOSURES AS FISHERIES MANAGEMENT TOOLS

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Abstract

Marine protected areas (MPAs) and fishing closures are among the suite of tools available to those managing the marine environment. The effectiveness of these tools for fisheries management depends on spatial and temporal boundaries and species within the boundaries. MPAs regulate a range of user groups to pursue biodiversity conservation objectives. Fishing closures, on the other hand, regulate fishers to pursue fisheries management objectives. Debate exists about the extent of secondary benefits of MPAs to fisheries conservation. In practice there can be considerable overlap between the two wherever there is potential to achieve common objectives. The purpose of the paper is to summarise the debate by focusing on management in Australian offshore waters, and to make some suggestions for moving forward.

Keywords: marine protected areas, fishing closures

FISHING CLOSURES

Fishing closures are one of the tools available to fisheries managers. Shipp (2002) describes a fisheries management tool as one that sustains and/or increases through time the yield of a fish stock, or several stocks, of an ecosystem. Fishing closures are areas closed to defined fishing activities. Only fishers are regulated within fishing closures.

Used in inland, estuarine and inshore fisheries in Australia since the mid-nineteenth century, fishing closures have been accepted by management agencies and stakeholders. They are usually developed in conjunction with the fishing industry and implemented by fisheries management agencies through fisheries management legislation. In federal fisheries, fishing closures are developed and administered by the Australian Fisheries Management Authority in consultation with scientists, management advisory committees (which include industry representation) and other stakeholders.

The objective of a fishing closure is often to conserve or ensure benefit to fish stocks, for example by limiting access to spawning or nursery grounds. Other objectives include protection of benthic habitat or migratory species, conservation of biodiversity, bycatch mitigation and management of conflict between resource users. In practice most fishing closures achieve multiple objectives. Examples of federal fishing closures include

- those over seagrass beds and other sensitive marine habitats that are highly productive nursery grounds for prawns and other marine species in the area of the Northern Prawn Fishery
- targeted shark fishing prohibitions in Victorian waters to protect juvenile sharks in the Southern Shark Fishery
- fishing prohibitions to protect the local mackerel icefish (*Champsocephalus gunnari)* stock at Shell Bank within the Heard and McDonald Island Fishery
- fishing prohibitions to protect biological diversity around the Tasmanian Seamounts in the Southern and Eastern Shark and Scalefish Fishery.

MARINE PROTECTED AREAS

The *Environment Protection and Biodiversity Conservation Act 1999* allows for an area of sea within a federal marine area to be proclaimed a federal reserve. The term 'marine protected area' (MPA) is more commonly used when referring to a marine reserve, although the two terms are used interchangeably (Environment Australia, Marine Protected Areas section, *pers. comm.*). The Commonwealth of Australia (2001) summarises the intent of MPAs as 'areas of sea established by law for the protection and maintenance of biological diversity and of natural and cultural resources'.

Historically, MPAs have been established to protect unique or iconic areas. They may also be designated to protect an ecosystem or habitat type, high species diversity, a location of intense biological activity, special cultural values, a tourist attraction or critical habitat for particular species or groups of species (adapted from Salm *et al.* 2002).

In 1991 the federal government initiated a long-term marine conservation program that included a commitment to expand Australia's existing system of MPAs through the establishment of a National Representative System of Marine Protected Areas (NRSMPA). The aim of the NRSMPA is to protect areas that represent all major ecological regions and the communities of plants and animals they contain (Environment Australia 2003). The NRSMPA will include protection and management of habitats significant to the life cycles of economically important species, including propagation areas (ANZECC 1998). Key characteristics of a NRSMPA site include that the area has been established especially for the conservation of biodiversity, can be classified into one or more IUCN Protected Area Management Categories[1] and contributes to the representativeness, comprehensiveness or adequacy of the national system (ANZECC 1998).

As part of the NRSMPA, several iconic MPAs have already been declared including Ningaloo Marine Park and Macquarie Island Marine Park. Potential MPA sites have been identified within the seagrass beds in the Northern Prawn Fishery and at several other locations within Australia's exclusive economic zone.

Whereas only fishers can be regulated in fisheries closures, users who can be regulated in MPAs include fishers, the tourism industry, shipping, defence and mining. Activities are regulated according to the risk they pose to the objectives assigned to conserving the area. Users are sometimes classified as extractive users, for example fisheries and mining, and non-extractive users, for example tourism, shipping and defence. Although extractive users often have greater impact on the marine environment, non-extractive users, such as marine tourism activities, can also impact the environment.

Commercial fisheries are the main extractive users of the offshore marine environment and, because of this, have been subject to more regulation through MPAs than most other users. Regulation of fishers through MPAs is in addition to regulation imposed for fisheries management purposes. Increased regulation of fisheries and the potential for even further regulation through the development of the NRSMPA has created uncertainty among commercial fishers about future access to the marine environment and fisheries resources.

Objectives of fishing closures and marine protected areas

Under the *Fisheries Management Act 1991*, several objectives must be pursued in managing federal fisheries. The following objective provides for the introduction of fishing closures:

> ensuring that the exploitation of fisheries resources and the carrying on of any related activities are conducted in a manner consistent with the principles of ecologically sustainable development and the exercise of the precautionary principle, in particular the need to have regard to the impact of fishing activities on non-target species and the long term sustainability of the marine environment.

ANZECC (1998) and Australia's Oceans Policy (Commonwealth of Australia 1998) identify the primary goal of the NRSMPA as

> to establish and manage a comprehensive, adequate and representative system of marine protected areas to contribute to the long-term ecological viability of marine and estuarine ecosystems, to maintain ecological processes and systems, and to protect biological diversity at all levels.

There is considerable overlap between the goal for the NRSMPA and the above objective for federal fisheries management, providing an avenue for integration between fishing closures and MPAs. Despite this overlap, Kearney *et al.* (2001) state that fisheries management and biodiversity conservation are poorly integrated in Australia. The impacts of MPAs on commercial fisheries and their management are poorly understood, which in turn influences the fishing industry's attitude to the implementation of marine protected areas.

Do marine protected areas provide benefits to fisheries?

Debate about whether MPAs provide benefits to fisheries has led to extensive research on existing MPAs to consider possible benefits to fisheries. Recent reviews (Kearney *et al.* 2001; Ward *et al.* 2001) present the varying opinions on MPAs in

[1] The World Conservation Union (IUCN) has identified seven international categories that form the basis for the Australian IUCN Reserve Management Principles. The categories represent varying degrees of human intervention (National Heritage Trust, 2002) from IUCN Category Ia Strict Nature Reserve (protected areas managed mainly for science) to IUCN Category VI Managed Resource Protected Areas (protected area managed mainly for the sustainable use of natural ecosystems).

relation to fisheries. The evidence is equivocal. Some research suggests that MPAs provide increased fisheries yields and that there are possible net economic benefits to fisheries, while other research indicates that MPAs have little direct benefit to fisheries. Ward *et al*. (2001) state that there is usually very little controversy about whether an MPA will conserve biodiversity. However, MPAs are rarely established with fishery benefits in mind, and any benefits to fisheries are incidental and may take a long time to be substantiated.

Theory suggests that MPAs conserve biodiversity by removing user impacts on habitats and target species. Ward *et al.* (2001) state that there is an overwhelming body of ecological theory and knowledge suggesting that MPAs can provide important benefits to fisheries, provided the MPAs are appropriately designed, sited and managed. Advocates of MPAs argue that the processes of spillover, and especially larval export, from well designed protected areas will increase fisheries recruitment, and thereby produce higher fisheries catches and yields over time (Ward *et al.* 2001). In summary, potential benefits of a MPA to fisheries can depend on the site, its boundaries, species within the protected area, whether they are sedentary or migratory and the nature of these species.

Despite this theoretical reasoning and the general belief among conservationists that MPAs should make a positive contribution to fisheries management, Ward *et al.* (2001) state that there appear to be few well documented examples of fisheries that have benefited from the introduction of MPAs. Evidence of benefits to fisheries generally comes from fisheries on tropical coral reefs, those in areas with high topographical relief, and those that are overfished. There is little documented evidence that, in a ecologically sustainable fishery, no-take areas offer advantages additional to those offered by more classical fisheries management (Ward *et al.* 2001).

There is confusion in Australia on the role of MPAs in biodiversity conservation and fisheries management. The selection and design of MPAs is strongly driven by the reliance on geophysical ecosystem surrogates (for example iconic areas) and an over-simplified use of the precautionary principle, demonstrating a level of unjustified optimism about the value of MPAs as a management tool (Kearney *et al.* 2001).

In the debate about benefits of MPAs to fisheries, potential negative impacts on commercial fisheries are rarely recognised. It is well known among fisheries managers that MPAs can lead to increases in fishing activity outside the protected areas and greater activity in other fisheries. Concentration of fishing effort into a smaller area could result in a much larger rate of damage to the environment (Parrish 1999). The combination of lost access to fishing grounds, poor planning and poor consultation, mixed and confusing messages on whether MPAs achieve their objectives, and lack of commitment to monitoring and enforcement, give fishers little confidence in the value of MPAs (Kearney *et al.* 2001).

From a fisheries perspective the crux of the issue is whether the establishment of an MPA will have a negative or positive impact on fisheries and those dependent on fisheries for their livelihood (Ward *et al.* 2001). Fishers rightly point to the obvious loss of a portion of their fishing grounds and potential subsequent loss of yield and profit that may result. MPA proponents point to environmental improvements the protected area will almost certainly bring, and to the potential, if the protected area is designed intelligently, to actually enhance medium and long-term yields to the fishery (Munro and Polunin 1997).

OVERLAP BETWEEN FISHING CLOSURES AND MARINE PROTECTED AREAS

Historically, fisheries management has been based on managing fishing characteristics (effort and catch) in relation to the target species (Ward *et al.* 2001), however the focus is changing to ecosystem-based fisheries management, with consideration of ecologically sustainable development and the precautionary principle. These are both an integral part of the development of MPAs. The shift reflects a philosophical shift in natural resource management (Kearney *et al.* 2001). In a practical sense, fishing closures are being implemented as an effective technique to mitigate against the impacts of the fishery on the broader marine environment, particularly in those fisheries involving benthic impacts and/or high bycatch (adapted from Kearney *et al.* 2001). Increasingly, the selection of areas as suitable for fishing closure is predicated on protection of the environment while maximising the benefits to the fishery.

Although benefits of MPAs to fisheries have not always been proven, theory indicates that both marine protected areas and fishing closures can benefit fish and ecosystems. The use of closed areas is a simple precautionary approach, removing some major anthropogenic impacts and protecting habitats. To date, MPAs in Australian offshore waters have arguably had most impact on commercial fishers. Wherever the objectives of both can be met, there are clear benefits from establishing MPAs where fishing closures exist or are to be established, especially because fishing is generally the activity most regulated by MPAs.

Merging fishing closures and MPAs through careful site selection could maximise the benefits of closed areas by minimising loss of fishing grounds, providing benefits to fisheries and achieving both biodiversity conservation and fisheries management objectives.

Flow-on benefits of fisheries closures are being increasingly recognised in the recent shift from target-species management towards ecosystem-based fisheries management. The Tasmanian Seamounts closure is one example – the fishing industry highlighted the unique biological diversity of the area and its importance to fisheries, and sought protection of the area through the introduction of a fisheries closure. As its biological importance became better known, the area was declared an MPA, restricting access by other users. The Tasmanian Seamounts are now part of the NRSMPA. Another example is the seagrass beds in the Northern Prawn Fishery that are permanently closed to fishing. The seagrass beds have been proposed as potential NRSMPA sites with support from the commercial fishing industry. The sites will be assessed to determine their suitability as MPAs. Aside from broader conservation benefits, the fishing industry and fisheries managers recognise that establishing MPAs will provide further protection of the seagrass beds by regulating other users of the area.

WHAT IS NEEDED NOW?

There are clear benefits in establishing parallels between fishing closures and MPAs. The potential for integration is being explored through government agencies and users of the marine environment. Integration will be refined through experience, application and increasing input from all stakeholders. It is important that the benefits of sharing experiences and achieving common objectives are continually and increasingly recognised.

Empirical evidence is needed to support claims that MPAs can directly benefit commercial fisheries. Well designed studies are needed to identify sites that, if protected, may provide benefits to both conservation biodiversity and fisheries. When such sites are protected, the ensuing changes will present opportunities for assessing their effects. Experimentation is warranted, given that the benefits of MPAs to fisheries are uncertain and fisheries are generally most affected by the introduction of MPAs.

The key principles of establishing MPAs as experiments for fisheries management should include

- explicit hypotheses, for example increased catch will be obtained because ....;
- experimental contrasts and replications;
- experimental design and analytical approach decided in advance;
- success criteria and evaluation timetable decided in advance; and
- a commitment to long-term funding to ensure enforcement and ownership.

Experimental programs can be costly, but the costs are an investment and the aim of any research will be to allow informed decisions to be made in light of new information.

CONCLUSIONS

Historically, fishing closures have been used to regulate fishing activity to protect target stocks and are increasingly being used to mitigate the impacts of fishing on the broader marine environment. MPAs regulate users and have various conservation objectives. Because of the overlap between the objectives, uses and even potential locations of the two, increasing consideration is being given to integrating fishing closures and MPAs. MPAs have a strong theoretical attraction, but there is little evidence that MPAs chosen only to conserve biodiversity will also benefit fisheries. Consultation with stakeholders as well as careful site selection will assist in maximising the benefits of closed areas to commercial fishing and in aligning fishing closures with MPAs. Once sites have been selected and implemented, experimental studies can provide information on the benefits of the closed areas to fisheries and broader biodiversity conservation.

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GIVING UP FISHING GROUND TO RESERVES: THE COSTS AND BENEFITS

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Abstract

Marine protected areas offer a range of potential benefits to fisheries for the conservation of biodiversity and some commercially exploited species. An earlier review identified 58 potential fisheries and biodiversity benefits that 'no-take' protected areas, which are a specific form of marine protected area, may be able to provide, including benefits to the fishery, the biodiversity, the fishery and ecosystem management system, the economics of coastal communities, and social benefits. Achieving this range of benefits for a fishery depends on reserve design including specific objectives, boundary placement, the species being exploited, and the effectiveness of the prevailing fishery management system. Securing such benefits requires stock assessment models that are spatially explicit at the scale of the protected area, criteria for ecosystems that can be incorporated into fisheries management systems, a careful evaluation of all the costs and all the benefits related to specific protected areas, and a willingness for institutions to work together. No-take protected areas jointly designed for fisheries and conservation may be complex to design, but offer the opportunity for fisheries to simultaneously achieve benefits for both their business and regional biodiversity conservation. Given the modest up-front cost of no-take area design, the minimal ongoing maintenance requirements, their high value for conservation, and possible benefits for fisheries themselves, no-take areas appear to offer fisheries a cost-effective opportunity to deal with a number of environmental and fishery management issues. Fishing ground may thus be forgone, but giving up some ground now for carefully designed reserves may assist a fishery to achieve enduring sustainability.

Keywords: ecosystem-based management, design objectives, no-take reserves, biodiversity conservation, fishery sustainability

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INTRODUCTION

Ecosystem-based management, the use of precautionary approaches, and the use of no-take reserves, are oft-cited elements of the new fisheries management paradigm that is developing in response to concerns about the effectiveness of classical fisheries management (Pauly *et al.* 2002; Ward *et al.* 2002). Fisheries management systems are often criticised for failing to take proper account of broader ecosystem issues, and in many cases for also failing to properly manage stocks of the target species. However, suggested improvements in fisheries management to respond to such issues have not been immediately embraced by fisheries scientists, managers, fishers or the public at large, and, indeed, are actively opposed by some (Sant 1996; Suman *et al.* 1999). No-take areas are commonly suggested as a useful tool to help resolve some of the problems of classical fisheries management, but proposals for no-take reserves usually meet strong opposition from the fishing sector and others (Roberts and Polunin 1993; Ballantine 1995; Gubbay 1995; Bohnsack 1997; Williams 1998; Ward *et al.* 2002).

A key question often posed is whether the establishment of no-take reserves will have a negative or positive impact on fisheries and those dependent on fisheries for their livelihood. Fishers correctly point to the obvious loss of a portion of their fishing grounds, and the subsequent potential for a loss in yield and profit that this may bring (Munro and Polunin 1997; Sanchirico and Wilen 2001). On the other hand, proponents of reserves point to the myriad of environmental improvements that could be achieved by the reserve, and to the potential, if the reserve is well designed, to enhance medium- and long-term yields to the fishery (Munro and Polunin 1997). Some argue that the loss of fishing grounds and its effect on displaced fishers is an 'invented problem', because it only looks at one aspect of a multi-dimensional problem. In this view, the key question is not what is the immediate impact on fishers, but rather what are the long-term benefits to fisheries and other stakeholders including the public interest (Ballantine 1995).

Classical fisheries management has long used areas closed to fishing as a specific tool to protect parts of the stock, for example when it is at a highly vulnerable stage of the life cycle, or to protect habitats considered critical to recruitment or spawning of the target species.

Also, beyond the specific purpose of stock management, in many fisheries there are areas that cannot be fished by the permitted gear types. For example, in a trawl fishery, there will normally be areas of seabed where there are rocky outcrops or canyons, or other obstacles such as shipwrecks, navigation markers, or undersea cables. In some areas, the risk of gear loss is too great, such as in deep waters, or in areas of high current or exceptionally soft sediments. And further, in many fisheries there are areas that have traditionally not been fished because they are beyond the reach of the available gear types or vessel capability. Together, these unfished areas, although in place for a variety of purposes, have contributed in the past to the provision of refugia for the target species, and, coincidentally, have probably substantially assisted with conservation of target and non-target species and habitats.

Several recent analyses of the literature have concluded that reserves provide benefits for species that are fished, in terms of increased abundance in reserves and increased catch in adjacent areas (Cote *et al.* 2001; Roberts *et al.* 2001; Halpern and Warner 2002). The limited empirical evidence that is available supports the theoretical contention that, given certain fishery situations and effective design of the reserve, no-take reserves can provide benefits for fisheries (Murawski *et al.* 2000; Fisher and Frank 2001; Roberts *et al.* 2001; Mapstone *et al.* in press). The types of benefits that closed areas (no-take areas) could offer both to fisheries management systems and to broader marine conservation objectives have also been recently reviewed (Ward *et al.* 2001). In the present paper, I summarise the potential benefits and costs of declaring no-take areas within existing fishing grounds. These are considered in the context of the integration of marine no-take reserves into fisheries management systems to achieve the objective of providing realisable net benefits for both fishing and regional conservation as a ‘double payoff’ (*sensu* Sanchirico and Wilen 2001). I use the term ‘Marine Protected Area (MPA)’ to mean any effective form of marine protection for an area of seabed and overlying waters, and the term ‘no-take area’ to be a form of MPA where all extraction of any living or non-living resources is prevented with any enduring form of effective control or rules.

THE POTENTIAL BENEFITS

The potential benefits of no-take reserves for both fisheries and conservation of biodiversity have been widely discussed in the mainstream science literature (e.g. Roberts *et al.* 2001; Halpern and Warner 2002). An extensive and well organised list of potential benefits is provided in Bohnsack (1998), and is summarised and extended here (Table 1, after Ward *et al.* 2001). In order to more precisely determine how these benefits may be realised and delivered in any specific fishery, a simple conceptual model of the processes that might operate to deliver such benefits was developed (Ward *et al.* 2001). The model enabled the potential benefits often discussed in the literature to be resolved into a more specific group of potential benefits that might be applicable to a single fishery in the circumstances where a no-take reserve was to be introduced into a fishery (Table 2).

Table 1. Potential Benefits of No-take Reserves (after Ward *et al.* 2001)

Fisheries for target species	increased abundance and spawning biomass
	increased mean age and size
	improved reproductive potential
	enhanced settlement and recruitment
	protection of genetic diversity
	protection of a critical supply of reproductive stock
	maintenance or enhancement of yields in adjacent fished areas
	reduced variability and uncertainty in fisheries yields
	increased likelihood of sustainable exploitation
Conservation and biodiversity	habitat protection
	increased biodiversity complexity
	protection of ecosystem structure, function and integrity
Broader benefits to science, fisheries management, the fishing industry and the public	provision of reference sites where scientific knowledge and understanding of natural populations of target and non-target species and ecosystem dynamics can be improved, and benchmarks established
	simplification of management regulations and compliance monitoring
	reduction in data requirements for fisheries management
	protection against fisheries management failure
	reduction of conflict amongst marine users
	improve opportunities for nature-based marine education and tourism
	improve stability in regional employment opportunities

Table 2. Potential Benefits for Fisheries (after Ward *et al.* 2001)

1. Biological Outcomes—fishery benefits, inside the reserve	Increased size/age of focal species of fish
	Increased abundance (density) of focal species of fish
	Increased size of spawning stock
	Increased reproductive output at age for focal species of fish
2. Biological Outcomes—fishery benefits, outside the reserve	Net movement of adult focal species of fish from inside to outside of reserve
	Increased abundance (density) of focal species (across total fishery)
	Increased individual size of focal species (across total fishery)
	Increased yield of focal species, standardised for fishing effort (across total fishery)
	Yields in other fisheries in region/district maintained
3. Biological Outcomes—non-fishery benefits, inside the reserve	Establishment/maintenance of areas of undisturbed habitat
	Enhanced habitat complexity
	Enhanced species diversity
	Enhanced community complexity (e.g. trophic complexity)
	Improved populations of fishing-affected species
4. Biological Outcomes—non-fishery benefits, outside the reserve	Maintenance/enhancement of habitat complexity, species diversity and/or community complexity
	Maintenance/enhancement of populations of fishing-affected species
5. Management Outcomes	Simplified enforcement
	Contributes to integrated ecosystem-based management of marine ecosystems
	Reduced data-collection requirements
6. Economic Outcomes	Enhanced and diversified local and regional economic opportunities
	Enhanced opportunities for employment in local industries
	Enhanced and diversified regional economic opportunities
7. Social Outcomes	Maintenance and enhancement of the social and cultural well-being of local communities

These 7 types of fisheries benefits relate to 58 specific attributes that can be considered to be the direct or indirect benefits for fisheries and ecosystems from the implementation of no-take reserves within a fishery (Ward *et al.* 2001, section 7.4). These attributes relate to benefits potentially available from a no-take reserve in an area of seabed and overlying waters. These benefits can also form the basis for identifying a set of criteria and related indicators that can be used in designing a reserve, and for determining whether a specific no-take area is achieving a desired level of performance (Ward *et al.* 2001).

But despite the potential of these benefits, no single set of benefits could be expected to apply to all fisheries, and not all the benefits described in either Table 1 or Table 2 could be expected to be achieved in any single fishery. Also, not all fisheries will stand to gain benefits equally across the full geographic scope of the fishery, or in all circumstances. For example, the benefits that accrue from the closed areas (Marine National Park zones) to the Reef Line Fishery in the Great Barrier Reef are variable across the region, ranging from almost zero to several-fold (Mapstone *et al.* in press). Nonetheless, some types of fisheries could capture many of these benefits. The fisheries that may be able to gain the most, and the largest benefits, are those that

- are being overfished (in any definition of 'overfishing'—see Ward *et al.* 2001),
- are fully exploited,
- have substantial demonstrated or suspected effects on ecosystems, habitats or species,
- exploit species that are associated with specific areas of seabed or have obligate habitat requirements,
- exploit species that can have high levels of recruitment, or
- exploit species that have a well studied life history.

Although many benefits may flow from no-take areas in certain circumstances, some fishery managers, fishers, fishery scientists and local communities still oppose the use of no-take areas in fishery management. If the benefits are intuitive, then why is it that there is still so much resistance to incorporating no-take reserves into existing fishery management systems?

There appear to be three key issues (related to potential benefits) that underpin resistance to no-take reserves by the fishing sector:

1) *the purpose of the reserves*: it is not yet fully acknowledged that the impacts of fisheries on the ecosystem fall within the boundaries of a fisheries

management system, and therefore to some extent are the responsibility of a fishery to address and resolve. Hence, the putative benefits of no-take reserves to the ecosystem, other than those strictly for stock management purposes, are not often considered to provide benefits to a fisheries management system. Here, the main benefits accepted as the basis for designs are those relating to stock management alone, and those relating to the ecosystem are discounted;

2) *no net benefit*: the benefits that potentially could apply are not sufficiently clear, in their nature and extent, so the apparent offset of the costs by the potential benefits can not be clearly determined by fishers to be adequate to gain their support; and

3) *the mismatch between benefits and costs*: even if the benefits outweigh the costs, and the net outcome is positive, the benefits do not offset the costs in an equitable manner; for example, fishers displaced may not be able to secure adequate value for their rights in the fishery, and benefits may accrue to a small profile of fishers who are not displaced, to other fishing sectors such as recreational sector, or to other interests entirely, such as tourism or conservation interests.

THE POTENTIAL COSTS

Irrespective of where a no-take reserve is to be placed, there will be costs that include the establishment costs, ongoing costs such as restriction on access or use, and costs of ensuring compliance. These costs will be imposed on users including any user who is denied access to the area, such as a fishery, or oil or gas companies or commercial shipping. Such costs can be direct, as in the denial of access, or indirect, as in the reduction of local employment and associated economic activity.

The economic cost to fishers of implementing no-take reserves is far from clear. It is tempting to speculate that the direct economic cost to a fishery is simply proportional to the areas closed from existing fishing, or to a projected loss of yield, and so on, but generic bio-economic models indicate that the cost of introducing no-take areas into a fishery is highly complex, and ultimately is likely to be a fishery-specific matter (Arnason 2001). And this is without accounting for the costs of management of the fishery or the reserves, or the less tangible cost offsets such as the potential for increased security for the stocks or insurance against adverse environmental conditions or mismanagement.

The costs of implementing a no-take area are most evident in the situation where the seabed to be reserved is presently within a fishing ground. Here, a fishery can clearly identify a loss of access to a specific area, and this may translate into a series of direct and indirect costs to the fishers and to the fishery as a whole (Table 3). In general, such costs are very hard to quantify. Many will be expressed in terms that are hard to measure, such as the cultural change in local communities, and many will be indirect costs and confounded with other dynamic aspects of regional communities and industries. Even the presumed direct costs, such as loss of yield in a fishery, may be very difficult to quantify in advance.

Table 3. Potential Costs to Fisheries from No-take Areas

Costs to fishers	Reduced access to fishing ground in proportion to the area reserved
	Loss of property and existing-use rights
	Reduced yield and consequent profits
	Need for larger vessels, vessel modifications, different fishing gear, and increased investment capital, to cater for longer travel distances or different fishing grounds
	Increased travel time and associated staff costs for each unit of yield
	Shift in available landing locations, ports or markets, because of distance from fishing grounds
	Increased concentration of fishers and effort in non-reserved areas may intensify competition, resulting in increased risks of overfishing, accidents, and elimination of less-efficient fishers
Costs to the fishery as a whole	Increased uncertainty in the industry
	Loss of property and existing-use rights
	Increased conflict with other resource users over access
	Destabilisation of existing fishery management systems
	Reduction in potential for competition in the fishing industry
Costs to ecosystems	Increased potential for environmental damage in non-reserved areas (from displaced fishing effort)
	Increased concentration of vessel infrastructure and shore-based processing facilities
Costs to regional communities	Reduced local employment in fishing support industries
	Loss of lifestyle and culture of local communities

In order to address the benefit–cost relationship in a manner that could lead to a set of equitable outcomes for fisheries, the following matters, which form the basis for arguments against the use of no-take areas in fisheries, need to be addressed:

- A clear analysis of the nature, timing and extent of the possible costs as well as the possible benefits—this will be closely related to the design objectives and the quality of the reserve design process;
- Provision of useful precedents and models that can guide a fishery in its assessment of the likelihood that the potential costs and benefits will be realised, and hence which specific costs will not be directly offset by benefits and so have to be resolved in other ways;
- A clear analysis of who will bear the costs and who will reap the benefits; and
- A clear process that describes how the areas to be reserved will be decided, and what arrangements will be made to manage the withdrawal of fishing access and any related fishery re-adjustments.

To enable the costs and benefits to be reasonably resolved in an equitable manner, taking account of the range of benefits, beneficiaries, fishery issues and the public interest, a comprehensive and inclusive system of marine assessment and management is required. Classical fisheries management, with its focus on stock assessment and resulting input/output controls, cannot in isolation resolve such issues. Ecosystem-based approaches for fisheries that involve substantial stakeholder participation and outcomes-based management (such as that proposed by Ward *et al.* 2002) will greatly assist in identifying how no-take areas can be used within fisheries and marine management systems in an equitable manner, and will permit both fisheries and conservation benefits to be sustainably secured.

MANAGEMENT SYSTEM BENEFITS

Many of the biological and within-reserve benefits that could be available from no-take areas have been widely debated in the literature. However, of perhaps greatest complexity, and less often discussed, are the potential improvements in the efficiency and effectiveness of marine management systems that may be available through the use of no-take areas as fishery management tools.

It is understood that no-take areas provide benefits for biodiversity conservation. But introduction of a network of no-take areas may also streamline fishery management and result in a series of cost reductions in the medium and long term (Li 2000). The process of design and negotiation of an agreed network of no-take areas will have a high initial cost, compared with existing levels of investment in this area by fisheries agencies, but ultimately it could result in lower levels of dispute, increased resource security, lower cost for compliance and monitoring, and more stable operating conditions for the fishery.

The other possible management benefits of no-take areas include reduced variability of yield, and the better estimation of stock-assessment parameters. A reduced variability in yield would probably be most obvious in fisheries that are fully or over-exploited, and may be mediated through a restoration of a more natural age structure in the fish populations and a broader spatial distribution of the population. These two factors may lead to greater natural resilience to the effects of unpredicted stress, such as climatic extremes, or weaknesses in stock-assessment models and consequent inappropriate settings for a Total Allowable Catch in a fishery.

The reserve effect has been well documented in the literature, but the implications of this for stock assessment have not yet been fully developed. The existence of no-take areas and populations of exploited species that could be considered to be living in their natural range of conditions offers the opportunity for specific population parameters, such as maximum size, natural population size and age structure, and natural mortality rates, to be estimated within both a fishery area and an unfished natural area. Stock assessments commonly depend on estimates of these parameters, and the inclusion of more robust estimates of natural variability and ranges in such parameters may improve the performance of the models in some fisheries (Punt *et al.* 2002). The mortality of an exploited species in a fishing ground may be heavily influenced indirectly by the fishery itself, such as through removal of important predators or prey, or changes in genetic composition in the fished population, and estimates of natural mortality in fishing grounds may well be different from those made on the same species in unfished areas. The ecosystem importance of this is that it may indicate the extent of fishing-induced impacts in fishing grounds, and may be a useful measure of broader ecosystem changes. But for fishery management, the contrast of these parameters between fished and unfished areas may also assist in establishing targets and limit reference points for fishery ecosystems. And further, exploration of such contrasts may assist to better resolve the nature of natural mortality processes that operate in fishing grounds and how they may differ from those of a natural population; ultimately, this could provide important insights into management of the exploited populations.

Other aspects of benefits for fishery management are perhaps easier to conceptualise. No-take areas make the rules of a fishery in those places clear and simple—no fishing is permitted. This means monitoring of compliance becomes easier for those areas, and can be simply based on existing VMS technology. Although the introduction of an area-based monitoring programme will be new for some fisheries, even there it should be possible to reduce the costs associated with the other forms of control applied in the fishery, perhaps in proportion to the extent of the no-take areas in relation to the size of the fishing grounds. In some fisheries, the cost of VMS compliance might be shared across a number of sectors, and the cost of fishery management might thus be reduced by reducing redundancy within overall marine management. Although these spatially based controls will have a significant initial cost, in the medium term, and beyond, the management costs should be reduced because of the overall simplification of the management system and the reduction in environmental disputes and requirements.

Beyond compliance and surveillance issues, increased security of access to specific fishing grounds should be paralleled by increased security of resource allocation, provided that the reserve design process is set within a broader ecosystem-based approach to marine management (see below). In such circumstances, it may be possible to drop some of the traditional input controls in a fishery and focus directly on output controls. This is because in many fisheries the environmental concerns are expressed as input controls on the fishery (such as limitations on permitted gear types and deployment methods) and some environmentally oriented input controls may therefore be replaced by the simpler single-input control of closed areas. This may enable a clearer focus on output controls, such as a quota. In these circumstances, the ecosystem basis for objectives in management should enable the fishery to have much more security over resource allocated for its exploitation, and this would also enable the fishery contribution to biodiversity conservation (through support for the no-take areas) to be more broadly recognised. This increased recognition would occur through an increased participation of conservation stakeholders in the fishery management. Taken together, the use of no-take areas that are highly valued for conservation purposes, an ecosystem-based management system, and the increased awareness and participation of conservation stakeholders should assist a fishery to become more stable in terms of environmental issues.

Where there is coordination between stakeholders on the design and management of closed areas, there is also the prospect of sharing management activities for management measures that might be held in common amongst stakeholders. For example, where fisheries compliance officers patrol fishing grounds they may also be able to identify breaches of reserve rules, assist with management of protected species, and provide coastal surveillance to assist with law enforcement in remote regions. Similarly, conservation rangers may be able to identify breaches of fishery rules and assist with compliance monitoring. The potential for sharing of such management activities is enhanced if the reserve strategies and objectives are established in common amongst stakeholders. This, may contribute to increased coordination amongst stakeholders, and hence an increase in management efficiency.

DUAL OBJECTIVES–THE DOUBLE PAYOFF

Modern concepts of fishery sustainability incorporate aspects of ecosystem protection, and fisheries are increasingly being required to demonstrate their lack of impacts in marine systems in order to be permitted to continue to fish. In other words, fisheries are being expected to take a more active part in ecosystem management issues, many of which may be the primary responsibility of other agencies or other sectors. Where it is difficult to demonstrate that fishing can be conducted with only minimal impact on non-target organisms and habitats, no-take areas offer fisheries managers an opportunity to provide for the conservation of species and habitats that may otherwise be affected by fishing.

Protection of a range of species in no-take areas may provide a fishery with an efficient tool to provide for protection of these non-target species, and if the no-take areas are designed correctly, they could simultaneously provide support to the target species and possibly the fishery. In such situations, fisheries can rightly claim to be supporting conservation objectives for the region, and be able to appropriately reject spurious claims of high levels of environmental damage by a fishery. This situation would, potentially, assist a fishery to avoid very expensive and long-term research programs designed to fully evaluate environmental impacts within fishing grounds, provided that non-target species and habitats were reasonably represented in no-take areas. Where dual objectives were being achieved, a fishery could appropriately claim to be delivering the 'double payoff', where both conservation and fishing achieve benefits (Sanchirico and Wilen 2001). Of course, there are many tools and mechanisms other than reserves that could be used (and are used in many fisheries) to reduce impacts on ecosystems and non-target organisms. Empirical evidence suggests that in well managed fisheries the effect of the fishery on non-target organisms can be reduced to a low and acceptable level through the classical tools of fishery management, such as effort control. Mapstone *et al.* (in press), show through a comparison of fished and unfished areas, that the Reef Line Fishery on the Great Barrier Reef has little detectable impact on non-harvested fish species at the present level of fishing effort. If the only objective of a fishery is to minimise impact on non-target organisms, it may well be true that one of the many classical tools available to fishery managers (such as gear modification, effort control in space and time, modification of fishing techniques, and so on) is the most appropriate strategy. However, the usual situation is that a fishery has multiple objectives, including minimising damage to habitats, non-target species and protected species, as well stock objectives such as maintaining spawning biomass and harvestable biomass, and no-take areas can be an effective and efficient tool to help meet such multiple objectives. After a comprehensive analysis of the fishery, Mapstone *et al.* (in press) conclude that closed areas would be an effective tool in the Reef Line Fishery to reduce effort and ensure that stock objectives continued to be met. Such closed areas would simultaneously assist with nature conservation in the GBR region and provide support for the Reef Line Fishery.

In many situations, MPAs are being initiated by conservation agencies to meet specific objectives for nature conservation. In Australia, a national program has embarked on securing a comprehensive, adequate and representative (CAR) system of marine protected areas for conservation purposes (the National Representative System of Marine Protected Areas—ANZECC (1998)). Although this network is also intended to provide support for sustainable use of fishery resources, the design process has not explicitly taken fishery values into account, and the resulting set of protected areas may not provide much support, if any, for fisheries.

If MPAs are to meet multiple sets of objectives, then it is clear that they must be designed with reference to selection criteria that reflect these multiple objectives. The success of MPAs for biodiversity conservation depends on the quality of the design process (Halpern and Warner 2002), including the use of specific selection criteria (Day *et al.* 2001). The success of no-take areas for fisheries is likely to be similarly critically dependent on the design process (Mayfield *et al.* 2000; Acosta 2002; Gerber *et al.* 2002). However, the selection criteria for no-take areas that will provide robust levels of protection of biodiversity (to the level required for regional conservation purposes) and simultaneously provide support for fishery production and management have yet to be developed. Amongst other difficulties, it is clear, for example, that for reserves to provide effective support for a fishery, the criteria need to be based on the specific biological characteristics of the target species, because their life-history characteristics may have a major influence on the effectiveness of a reserve in supporting a fishery (Sumaila 1998; Sanchirico and Wilen 2001). The rate of transfer of exploited species between reserves and fished areas appears to be of particular importance (Tuck and Possingham 2000; Sanchirico and Wilen 2001). And further, simple models of reserve implementation suggest that designs that will achieve the 'double payoff' may need to use parameters and criteria that are relatively complex, to avoid the risk of failing to simultaneously achieve both conservation and fishing objectives (Gerber *et al.* 2002).

Therefore, at this stage, although there is little doubt that such joint criteria can be developed and applied to select a system of MPAs to serve both biodiversity conservation and fishery production, there are only limited precedents that may be used for guidance, and the process may prove to be complex. Undoubtedly, no-take reserves for fisheries have biodiversity values, and the reverse may be true to some extent, but there have been few MPA programmes that have explicitly undertaken to optimise a set of no-take areas to jointly achieve both outcomes for conservation and outcomes for fisheries (but see Villa *et al.* 2002, Day *et al.* 2001). Mapstone *et al.* (in press) document the benefits provided by the closed areas of the Great Barrier Reef Marine Park for the Reef Line Fishery, but these closed areas were designed specifically for nature conservation purposes and their fishery benefits may be considered to be coincidental. Overall, therefore, although there are good reasons to expect improvements in fishery management systems that result from a joint approach to designing no-take areas, this has yet to be demonstrated in practice.

MARINE MANAGEMENT FRAMEWORK FOR FISHERY NO-TAKE RESERVES

The MPA declaration process in most situations is highly complex, incorporating multiple sectors and stakeholders, and some compromise may be required from participants, because, typically, existing users have strong positions in relation to their economic rights and values. In most jurisdictions, the cumulative effects of repeated compromises have resulted in continuously reducing sets of areas for conservation purposes. This process of incremental and continuing erosion has resulted in enhanced efforts to secure no-take areas purely for conservation. However, resolution of the competing interests for the seabed and overlying waters from various uses and interests, including fishing and conservation, needs a clear and explicit planning and assessment framework where all of these matters can be dealt with in an equitable and integrated manner. Several planning and assessment tools are available for use in such circumstances, although few have yet been used in fishery management (Ward *et al.* 1998, 1999, 2002; Villa *et al.* 2002; Day *et al.* 2001).

Given the imperatives for increased integration of marine management systems, a fishery will need to participate in such planning processes in a manner that is effective and consistent with that of the other marine stakeholders so that its engagement is fully effective and achieves a broadly agreed and supported set of outcomes. To be able to secure the benefits of no-take areas, at a minimum, a fishery will need to have the following aspects of management in place:

- an effective and efficient management system that can apply spatial controls over fishing effort;
- stock assessment models that are spatially explicit at the scale of the protected area;
- quality spatially resolved data on fishery catch and effort;
- criteria for ecosystems that can be incorporated into fisheries management systems as targets and limit reference points;
- a capacity to make a careful evaluation of all the costs and all the benefits related to specific protected areas; and
- a willingness for institutions to work together.

For fisheries to be able to secure the possible benefits that a set of no-take areas may offer, integrated marine planning and design processes will need to be inclusive and cover a range of marine management interests. The design approach should be ecosystem-based, and include a range of stakeholders and a broad approach to marine assessment and planning. This will best enable a set of no-take areas to be designed and used as a tool to both conserve regional biodiversity and support existing fisheries management. By designing a set of no-take areas within a planning framework that is based on the principles of ecosystem-based management (Ward *et al.* 2002), the resulting system of MPAs will best offer fisheries the opportunity to secure a number of the benefits described above.

THE COST-BENEFIT TRADE-OFFS

In the present situation, where there are many presumed benefits but few precedents, it is difficult for a fishery to make an informed evaluation of how the benefits of no-take reserves will compare with the costs of displacing existing fishing activities from an area. The first obvious problem is that projected benefits are difficult to compare with projected costs, since both are likely to be highly uncertain. But perhaps even more important than this uncertainty, the presumed benefits of no-take reserves are likely to flow to beneficiaries who do not bear the cost of creating the reserves. This means that there is a limited opportunity to have the benefits of reserve creation directly offset against the costs at the level of individual fishers. And so, given the limited experience with the use of closed areas in stock assessments, together with the lack of spatially based management arrangements in many fisheries, and very often a limited legislative mandate and correspondingly limited set of institutional arrangements that would enable this to work, it is hardly surprising that fishers are often reluctant to support the creation of no-take reserves.

From a fishing perspective, giving up ground is always a direct cost to a fishery, although the magnitude of that cost depends on exactly where the reserve is located. It may be feasible, for example, to place a reserve in a location that is not part of the prime fishing grounds, and so to minimise the impact of the reserve on the fishing activity. This may not always be possible, but is an important design option to be considered. Depending on the reserve location, the fishing grounds may be forced to contract substantially, and this could result in fewer fish caught, possibly lower-quality fish in the catch, and a range of other effects on fishers. The reserve placement may also mean that some fishers have to steam for very long distances to be able to get from a safe haven to where they can fish, and this may impose additional costs that are not supportable. The no-take reserve placement may therefore force some fishers out of business, and result in economic restructuring of a fishery that may not be socially or even ecologically desirable. It may, for example, result in excessive effort in non-reserved areas, and require a substantial reduction in overall capacity in the fishery to reduce effort to acceptable levels outside the reserve. Of course, some of this, perhaps all, may be offset by the reserve effect (*sensu* Ward *et al.* 2001) and the resulting spillover or larval export of exploited species into the fishery.

Clearly, any restructuring of a fishery as a consequence of the introduction of no-take areas could only reasonably be conducted taking these potential benefits into account. However, such potential benefits are highly dependent on the reserve design. Ultimately, a reserve system that is not well designed could reduce yield, reduce employment in the fishing sector, and have undesirable flow-on of social consequences to local townships and communities. The important point here is that many of the costs of implementing a no-take reserve system into a fishery are non-biological in nature (and not expressed solely as changes in catch), and they are critically dependent on the design of the reserve system and any restructuring that a fishery may subsequently adopt.

The implications of this complexity in the nature of costs and benefits are that it is difficult to directly link benefits to costs at the level of the individual fishery operator. It also means that although the ecological and fish stock benefits are the critical driving forces for the establishment of a no-take reserve system, they relate only indirectly to the costs, and are hence not likely to be the key arguments that have to be addressed by fishery and marine managers in the process of design and implementation of a system of no-take reserves.

Because of the highly complex nature of the problem and the lack of precedents, it seems important for fishery and marine managers to begin to establish no-take reserves in fishing grounds as learning and demonstration projects. Such projects would need to be full-scale implementations, not small or pilot-scale exercises, and be explicitly designed to allow for uncertain outcomes, to involve effective collaborations amongst fishers, government agencies and conservation stakeholders, and to document successes and failures. In this way, fishers everywhere would then be able to make more informed decisions in the future about the role and value of no-take areas in their fisheries.

CONCLUSIONS

Many fisheries stand to achieve a net benefit by taking a pro-active approach to the issue of no-take marine protected areas. The main areas where benefits are likely to outweigh the costs for a fishery are as follows:

- Public perception of increased stewardship of marine ecosystems (support for the notion of responsible and sustainable wild capture fisheries);
- Reduced costs for management of the fishery and marine systems more generally in the medium and long term;
- Maintenance of habitats and ecosystems to a standard required by national and State legislation;
- Improved stock management (depending on the target species and the fishery type);
- Improved insurance of the fishery against mismanagement or adverse climatic effects; and
- Improved ecosystem conditions, which directly and indirectly provide support to a fishery.

Trading off the costs of no-take areas against the benefits they might be able to deliver is not a simple process. The design criteria for effective 'double payoff' reserves are likely to be complex and to involve consequent changes to existing management controls. And in seeking the support of fishers to give up fishing ground for reserves, the costs imposed by reserve declarations will need to be considered in the context of both the potential benefits as well as the identity of the beneficiaries, to overcome the problem of a mismatch of benefits and costs.

If there can be an adequate arrangement of institutional coordination and collaboration amongst relevant government agencies, fishing industry groups, environmental groups and local government, a system of no-take areas that provide joint benefits for conservation and fishing will be likely to reduce the overall costs of ocean and estuary management. This will be because management responsibilities will be simplified in reserve areas where there are multiple jurisdictional obligations, and fishery management systems will be streamlined. Provided that fisheries follow the general model of ecosystem-based management in identifying no-take areas, and that interested parties can reach agreements on sharing of responsibilities and costs, there will be a reduced overall cost to fishers and to the public for implementing these management arrangements.

Fisheries operate in a business environment where markets and investors are increasingly being sensitised to the economic, ecological and social implications of business activities. No-take areas that are properly designed and managed would greatly assist a fishery to meet its sustainability objectives in terms of both national legislative requirements and the emerging sustainability and triple-bottom-line (TBL) business reporting systems (see Whittaker (1999) for a brief description of contemporary resource-sector approaches to TBL reporting). Given the conservation-sector imperatives, but lack of successful precedents for joint conservation- and fishery-designed MPAs, the fisheries sector should specifically design and implement a series of full-scale no-take areas in selected fisheries. These should be designed and implemented as learning and demonstration projects, where outcomes are fully analysed and documented, and lessons disseminated.

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WESTERN AUSTRALIAN COMMUNITY-INITIATED FISH HABITAT PROTECTION AREAS

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Keywords: fish habitat protection areas, community consultation, guidelines, fisheries

PROTECTION OF WESTERN AUSTRALIAN WATERS

For many years, the Department of Fisheries has used fisheries legislation to limit the impact of fishing operations on the environment. For example, the Shark Bay Marine Park, which was established in 1990, reflects the boundaries of the prawn-trawl closure that was established to protect the seagrass beds in 1960. Local government authorities and communities have also sought assistance from the Minister for Fisheries and the Department to protect areas from fishing for nature conservation and to provide diving sites that are not affected by fishing. These include the following:

- Gantheume Point near Broome, which provides habitat for a number of rare shellfish that were being overexploited by shell collectors;
- Sampson II and Kumunya Wrecks near Point Sampson, which will enable divers to observe a small exploited population of reef fish that inhabit the wrecks;
- Point Quobba near Carnarvon, which is a small area protected to allow divers to observe unexploited populations of fish and corals;
- Yallingup Reef at Yallingup, which is a nearshore reef system that is a popular diving site;
- HMAS *Swan* near Dunsborough, which is a naval vessel that was scuttled in Geographe Bay to provide a dive site;
- Cowaramup Bay at Grace Town, which is one of the few areas of relatively sheltered water in the Cape Naturaliste district and is a popular dive site;
- HMAS *Perth* Wreck at Albany, which is a naval vessel that was scuttled in King George Sound to provide a dive site;
- Sanko Harvester Wreck at Esperance; and
- Esperance Jetty, which provides a small area of unfished water set aside for the benefit of local divers.

These closures provide only for the control of fishing and not for other human activities, and may occur with minimal consultation.

FISH HABITAT PROTECTION AREAS

Under the Australian federal *Fish Resources Management Act 1994*, the Minister for Fisheries may set aside an area of State Waters as a Fish Habitat Protection Area (FHPA). This enables the Minister to regulate fishing and any other human activity that may affect the marine environment. The first FHPA, established at the Abrolhos Islands, was set aside in 1998. This FHPA was established as an outcome of a Cabinet Decision.

The Minister for Fisheries must prepare a plan of management before gazettal of these FHPAs, and each plan includes a formal public comment process.

GUIDELINES PREPARED

In 1999, the Minister for Fisheries directed that a set of Guidelines for the establishment of FHPAs be prepared and this work was published in October 2001 as Fisheries Management Paper No. 152.

LANCELIN ISLAND FISH HABITAT PROTECTION AREA

In 1998, members of the Friends of Lancelin Island and the Western Australian (WA) Marine Conservation Society approached the Department seeking assistance in establishing a small FHPA in the lagoon next to the Island.

Lancelin Island is an important sea bird rookery and is a Nature Reserve under the *Conservation and Land Management Act 1984*. The Friends of Lancelin Island have been working to protect the rookeries over a long period of time. The aim of the FHPA is to establish a 'no take' area over the fringing reef and lagoon to complement the work being undertaken to protect the Nature Reserve.

After a briefing in 1998, the Minister for Fisheries agreed that the consultation process to prepare a draft plan for a FHPA could proceed. He also agreed that Fishcare WA funding was available to assist in the consultation process. The Minister agreed that the Friends of Lancelin Island and the Marine Conservation Society could undertake the consultation and prepare a draft plan, provided that the work was undertaken under the general guidance of the Department.

The WA Marine Conservation Society undertook a thorough consultation process including advertising and workshops involving local interest groups. The Society developed a draft plan that was released for public comment, and then produced a final plan. The FHPA established in 2001 is a 'no take' area with strict limits on boat use.

COTTESLOE REEF FISH HABITAT PROTECTION AREA

The Cottesloe Reef Protection Society conducted a public meeting in February 1998 to determine whether an increased level of protection for the Cottesloe Reef System could be achieved. Subsequently, the Society sought the assistance of the Minister for Fisheries, who agreed that the consultation process to establish a FHPA could proceed.

The Society advertised the proposal and conducted workshops, reef days and public meetings, and a draft plan was released for public comment. The draft plan dealt with proposals to exclude or reduce fishing pressure, anchoring and jet skis, to improve water quality and to involve the community in the long-term protection and monitoring of the area. The Minister received almost 1000 submissions in overwhelming support for the proposal.

In September 2001, the Minister for Fisheries published the declaration of the Cottesloe Reef Fish Habitat Protection Area.

PROPOSED MIABOOLYA BEACH FISH HABITAT PROTECTION AREA

The Miaboolya Beach FHPA is immediately north of the Gascoyne River Mouth near Carnarvon. It includes the nearshore waters and the adjoining mangrove system which is part of the Gascoyne River Delta.

In 1996, the Carnarvon Senior High School obtained a Fishcare WA grant to undertake research on the habitat of recreational fish species on the eastern shore of Shark Bay. This work demonstrated that the Miaboolya Beach and associated mangrove system is, by far, the most important fish nursery in the region for a number of fish stocks exploited by recreational fishers, including tailor, dart, mulloway, whiting and mullet. The mangrove system also supports a valuable recreational mud crab fishery. It appears that the high nutrient content of the Miaboolya Beach and mangrove system, as well as additional cover provided by the muddy water, sand bars and mangroves, support the young fish in these nursery areas.

The School approached the Department expressing concern about environmental issues associated with flood-plain management on the Gascoyne River and its possible impacts upon the Miaboolya System. The School recommended that the area become a FHPA. This resulted in the proposal for a FHPA becoming a recommendation in the Gascoyne Regional Fisheries Environmental Management Strategy.

A draft plan was produced in February 2002 and public comments received. The Minister for Fisheries has agreed that the gazettal of the FHPA should occur and it is anticipated that the gazettal will proceed in early 2003.

CONCLUSIONS

Community-initiated FHPAs in Western Australia

- are likely to be limited in extent to contain the demands of the consultation process and ongoing management;
- require a high level of community commitment and involvement;
- have been beneficial in terms of increasing community stewardship of the marine environment;
- have a higher likelihood of long-term success where the proponent is an existing community group; and
- are associated with risks that the outcome of the consultation process is not consistent with community expectations.

Is There a Place for Aquatic Protected Areas in the Management of Small Pelagic Fish in Coastal Waters?

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Abstract

Yes. *Sardinops sagax* is continuously distribution, but with uneven abundance, along the south coast of Western Australia (SCWA); stocks are small by global standards and are exploited from three ports along SCWA. Mixing of adult *Sardinops* along the SCWA is insignificant during periods of medium to high abundance. The same has not been found for juveniles, whose spatial relationships remain problematic. An ecosystem-level event, in which mass mortality caused the southern Australian population of *Sardinops* suddenly to decrease in size by 70%, permitted otherwise unattainable observations. Recovery of the *Sardinops* stocks along the SCWA progressed east to west; patterns in timing and magnitude of recovery indicate that the more-western locations were seeded with recruits from the eastern regions. Adults breeding in the eastern region were able to contribute to the recovery in all regions because they suffered less exploitation during the decade preceding the event. The combination of small stocks with limited alongshore mixing and areas of coast not accessible to fishing has resulted in a default-APA (aquatic protected area) that has already benefited the pelagic ecosystem off southern WA. Examples of exploited stocks for which unfished portions act as default-APAs need to be identified, catalogued and subjected to meta-data studies so as to search for those similarities that provide an indication of what works (e.g. relevant spatial scales) when planning APAs. A spectrum of case histories would provide the basis for making many informed decisions without the need to wait for 1–5 years of research in each case.

***Keywords*:** APA, MPA, marine reserve, fisheries management, spatial dynamics

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Introduction

The aim of this paper was to provide a review of the biology and fishery for *Sardinops* in southern Western Australia (WA); because this fishery has been the focus of a dedicated research program for 14 years (since 1988) it was hoped that the amount of knowledge would be sufficient to address some issues relevant to aquatic protected areas (APAs). This aim was developed in consideration of the assumption that available data sets may provide answers to some of the questions regarding APAs prior to the need to instigate further research at baseline levels. Following this reasoning, new research should build on the medium- and long-term data sets already available.

The approach taken was to synthesize the available information for the southern WA *Sardinops* fishery and then regard it from the APA perspective rather than from a fisheries management perspective. Fortuitously for this approach, the contrast in distribution and stock size of the *Sardinops* along southern WA provided by the dramatic mass-mortality-induced stock decline in 1998/99 (Gaughan *et al.* 2000) and subsequent recovery (presented here) permits some current working hypotheses to be examined from the APA perspective and for a conceptual model of the spatial dynamics of southern WA *Sardinops* to be developed. Following a brief outline of stock size and biology of *Sardinops* in southern WA, this paper summarizes the spatial dynamics, including the conceptual models currently used to manage the purse-seine fisheries in this region. This focus on spatial dynamics is required since in the context of APAs as a fishery management tool, knowledge of fish movement is paramount. For example, in a bioeconomic approach to investigating the usefulness of APAs within a fishery context Sumaila *et al.* (2000) concluded that high exchange rates between protected areas and the exploitative fishery are required; knowledge of spatial dynamics is a prerequisite to considering exchange rates.

The *Sardinops* Fishery on the South Coast of Western Australia

Sardinops is one of the most important single species that contribute to world fishery catches owing to the large, and periodically massive (4–13.5 million tonnes), stocks found in southern Africa, waters around Japan and the mid-latitude Pacific coasts of North and South America (Schwartzlose *et al.* 1999). Combined annual landings from these regions have often exceeded

10 million tonnes (e.g. Fréon and Misund 1999). Considerably smaller fisheries for *Sardinops* occur in southern Australian waters. These smaller catches are undoubtedly due to smaller stock sizes available to the purse-seine fisheries; in turn, the smaller stock sizes have been attributed to the substantially lower productivity of coastal waters in Australia (Lenanton *et al.* 1991; Pearce *et al.* 2000; Gaughan *et al.* 2001a). Of particular contrast to regions elsewhere that support very large *Sardinops* fisheries, there are no regions of southern Australian waters with globally significant upwelling systems. Whereas the population dynamics of *Sardinops* in regions that have significant upwelling are typically strongly linked to that upwelling (e.g. Bakun 1996), in south-western Australia there is no regular upwelling; neither is there a clear dominating oceanographic influence on *Sardinops* productivity and population dynamics, or least not one that has been identified.

In southern WA, purse-seine fisheries for *Sardinops* are located off the ports of Albany, Bremer Bay and Esperance (Fig. 1). Management has primarily operated through individual transferable quotas (ITQs) and total allowable catches (TACs); these are adjusted annually for each south-coast management zone. Annual catches for the whole region peaked at 8400 tonnes in 1988, all of which came from Albany in what was the last year prior to the implementation of ITQs for this region (Gaughan *et al.* 2002). In the context of APAs, the limited extent of the coastline actually covered by the fishing fleet is an important feature of the purse-seine industry (see Fig. 1). Failure to fish the whole range of the target species is viewed as advantageous for stock sustainability because, by default, it equates to exploitation of a reduced proportion of the stock (Guénette *et al.* 1998). Indeed, it is this feature of the southern WA *Sardinops* fishery that makes the available data attractive for recasting in an APA framework. Likewise, information for fisheries elsewhere that also fish only part of the range of their target species will be similarly amenable to discussions on APAs.

STOCK SIZE

Since detailed research on the purse-seine fisheries commenced in 1988, the combined spawning biomass of *Sardinops* across the south coast was estimated to reach a maximum of about 85 000 tonnes in 1994 (Hall 2000). This estimate was made using an age-structured model that was tuned with fishery-independent estimates of spawning biomass obtained by the daily egg production method (DEPM, e.g. Fletcher *et al.* 1996). Recent further development of the integrated model (P Stephenson, unpublished) and re-assessment of the DEPM estimates of spawning biomass suggest that the spawning biomass is typically <70% of the maximum (D Gaughan, *et al.* in press).

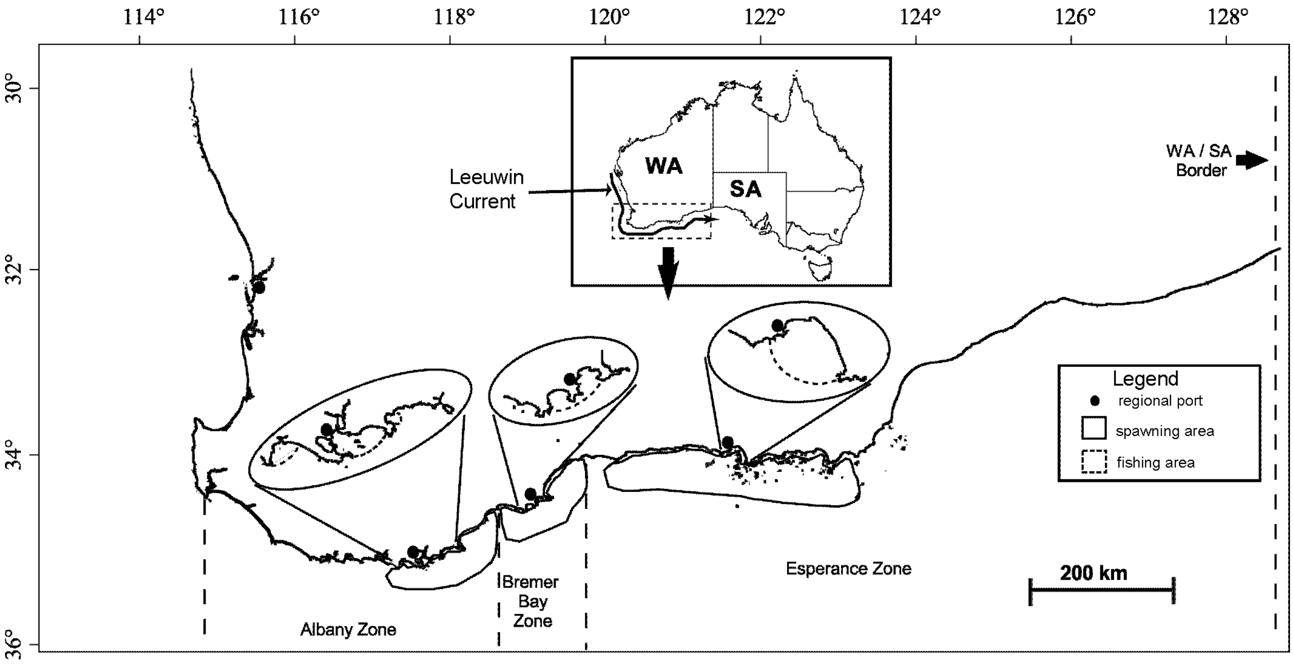

Fig. 1. The purse seine management zones of southern Western Australia. Note the small extent of the fishing areas.

BIOLOGY

Growth and reproduction

Sardinops in southern WA live to eight or nine years old, begin recruiting at 2 years old and are fully recruited at four years (Fletcher 1995; Fletcher and Blight 1996). Maturity is reached at two years of age and at a fork length of ~120 mm. *Sardinops* in southern WA spawn in continental shelf waters (Fletcher and Tregonning 1992; Fletcher *et al.* 1994; Fletcher *et al.* 1996). Gonadosomatic indices (GSI, gonad weight as a proportion of body weight) pooled across years and ages show some key differences in annual spawning patterns. Briefly, GSI was high in Albany from January to August, in Bremer Bay from March to June and in Esperance only from March to April (Gaughan *et al.* 2002). The longer-term pattern is thus for the length of the spawning season to decrease from west to east.

Spatial dynamics

Sardinops from the Albany, Bremer Bay and Esperance regions are considered to constitute functionally distinct adult assemblages (FDAAs; Fletcher *et al.* 1994; Gaughan *et al.* 2002). Thus, the mature *Sardinops* targeted by the fleets that operate out of each of the three ports essentially do not mix to any significant degree once recruited to their respective regions. This is not implying that separate populations of *Sardinops* occur in southern WA. Rather, there is a more-or-less continuous distribution of *Sardinops* between the three regions and eastwards beyond Esperance to the Great Australian Bight. The FDAAs are considered to contribute to a common pool of recruits. Any appreciable increase in the number of individuals within a particular regional FDAA depends on recruitment rather than migration.

Despite the evidence for these FDAAs, little is known about the potential links amongst the management zones during the pre-recruit life-history stages except that there is eastwards transport of *Sardinops* eggs and larvae during the winter spawning season (Fig. 2); this transport is caused by the Leeuwin Current, with assistance from eastward surface drift set up by the winter north-westerly winds (Fletcher *et al.* 1994; Gaughan *et al.* 2001b). It is likely that larvae arising from commercially exploited *Sardinops* populations in WA can be passively transported close to the region of the *Sardinops* fishery on the central coast of SA prior to metamorphosis (Gaughan *et al.* 2001b). Given that there is a potential for links between *Sardinops* management units in WA and SA (i.e. across >1000 km); links across hundreds of kilometres within WA are highly likely. Thus, it can be concluded that fished and unfished parts of the *Sardinops* distribution in southern WA (also see Fig. 3) are linked via the larval life history stage.

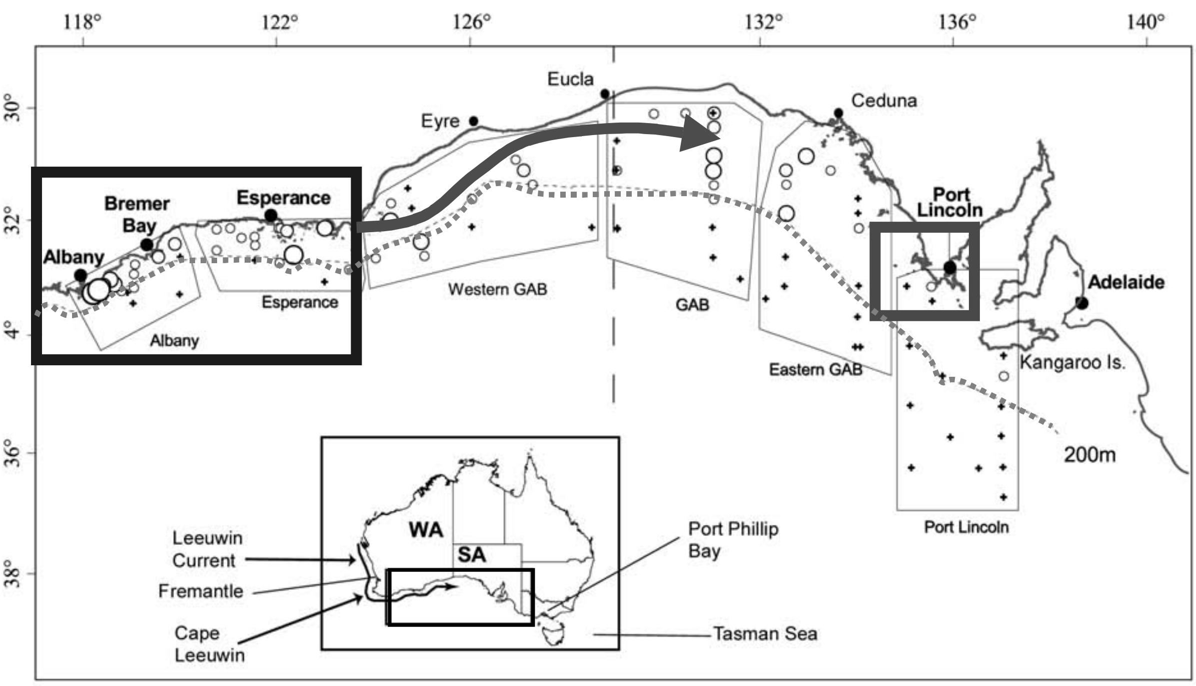

Fig. 2. Plankton sampling stations, with circles denoting relative abundance of *Sardinops* larvae; this 1994 survey established that larvae from the main management areas in Western Australia (large box) could be transported significant distances (arrow) towards the Port Lincoln *Sardinops* fishery in South Australia (small box). Modified from Gaughan *et al* (2001b).

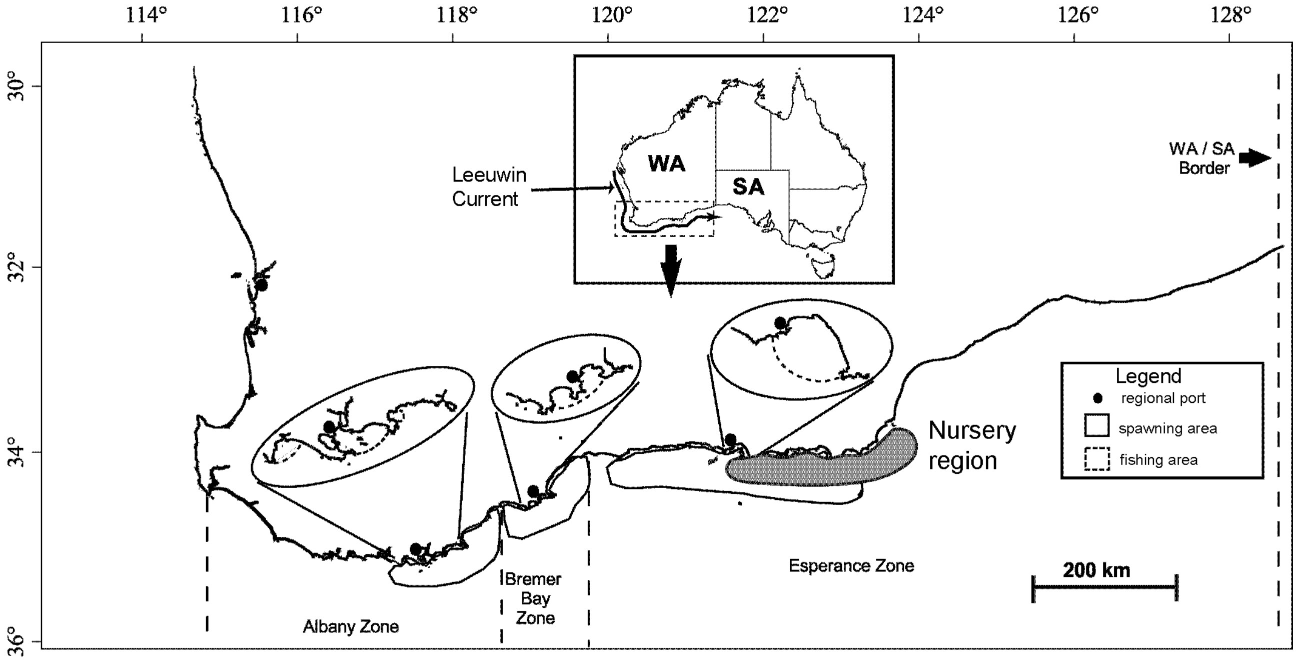

Fig. 3. The proposed *Sardinops* nursery region in southern WA (hatched area), which may extend further east. This nursery region coincides with the unfished parts of the southern WA breeding stock of *Sardinops.*

Eastward transport of larvae requires a return westward migration of juveniles in order for the FDAAs in WA to be maintained. This hypothesis is consistent with anecdotal observations by purse-seine fishers in southern WA that juvenile *Sardinops* recruit from the east. The simplest model is therefore to assume that a "return migration" occurs within the approximate two-year period leading up to initial recruitment to the fishery. The spatial origin of this return migration is not known. No nursery areas for *Sardinops* have been conclusively identified in southern WA (Gaughan *et al.* 2002). This is not to say that nursery areas do not exist, but rather that the majority of juveniles in any one year can not be found as a spatially cohesive group. However, although the precise location of a nursery area cannot be ascribed with certainty, the return migration model includes the implicit assumption that the broad region east of Esperance contains sufficient quantities of pre-recruits to be important for the whole southern WA *Sardinops* stock and is therefore proposed to be a nursery area (Fig. 3). Although a nursery area has not been formally identified, it has been established through examination of otolith chemistry that juvenile *Sardinops* from the three south-coast fishing locations show no difference in environmental history (Gaughan *et al.* 2001c), whereas adults in these same regions were different (Edmonds and Fletcher 1997). This conclusively indicates that the juveniles exhibit spatial behaviour independent of the adults. Thus, both larval and juvenile stages, through eastward and westward movements respectively, provide links between the spatially defined adult *Sardinops* (i.e. the FDAAs) along the continuum of their distribution. If significant proportions of recruitment to each of the southern WA FDAAs result from a pool of westward-migrating juveniles and this pool periodically loses members (e.g. owing to encounters with suitable habitat, predation) there will be a gradual decrease in size of the pool sequentially from east to west (Gaughan *et al.* 2001b). It then follows that through sequential depletion the more westward zones (i.e. Albany and Bremer Bay) could expect fewer recruits than Esperance (Fig. 4). Annual catch-at-age data indicate that this has in fact been the case since consistent monitoring of commercial catches began in 1989 (Gaughan *et al.* 2002).

Developing this concept further, the existence of a pool of juveniles decrees that the overall level of recruitment will determine the level of recruitment at each region. Given the sequential depletion described above, if the overall recruitment level falls then it is possible that the bulk of recruitment in some years may end up in the Esperance region, with less in Bremer Bay and none in Albany (Fig. 4). Annual catch-at-age data have also indicated that relative recruitment levels at Albany and Bremer Bay have also been more variable than that at Esperance (Gaughan *et al.* 2002; unpublished data). The spatial recruitment model developed here (i.e. Fig. 4) can thus be further developed to include a temporal depiction of this same scenario whereby Albany receives recruits only in those years when the overall recruitment is above some threshold value (Fig. 5).

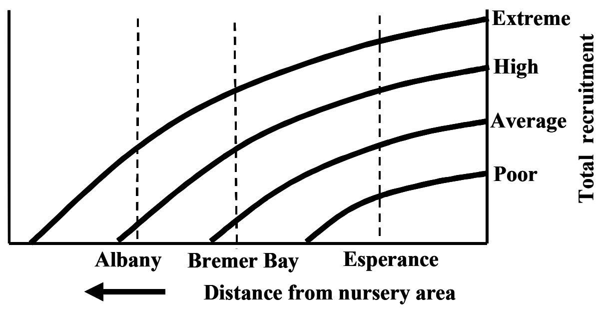

Fig. 4. Conceptual representation of sequential depletion in numbers of westward migrating *Sardinops* recruits from a nursery ground east of Esperance. This model suggests that at poor overall recruitment levels no recruits may reach Albany; conversely, Albany may only achieve outstanding recruitment levels on rare occasions.

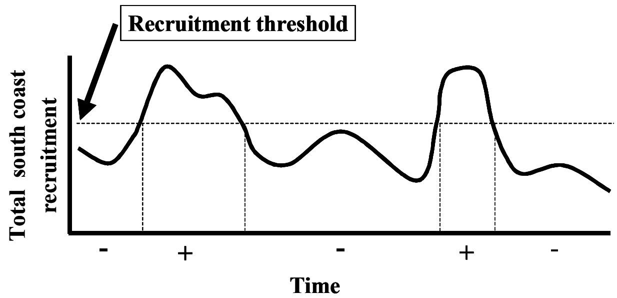

Fig. 5. Conceptual representation of a recruitment threshold for Sardinops at the Albany region on the southern coast of Western Australia. The threshold refers to recruitment levels sufficient to increase the spawning biomass (+ sign) at Albany; low levels of recruitment are those insufficient to counter population decrease due to natural mortality (- sign).

POPULATION DECLINE AND MASS MORTALITY

Poor recruitment to Albany and Bremer Bay occurred during the 1990s, and along with fishing pressure and a mass mortality event in 1995 led to a decline in *Sardinops* stocks in southern WA (Fletcher *et al.* 1997; Gaughan *et al.* 2002; Murray and Gaughan 2003). This was accentuated by a second mass mortality in 1998/99 that further reduced the spawning biomass by 70% in just a matter of weeks (Gaughan *et al.* 2000; Gaughan 2001). A series of DEPM (daily egg production method) surveys along the southern WA coast in 1999 revealed not only the 70% decline in spawning biomass but also a concomitant massive decline in the range of *Sardinops*, particularly in the Albany region (Fig. 6a; D Gaughan, unpublished).

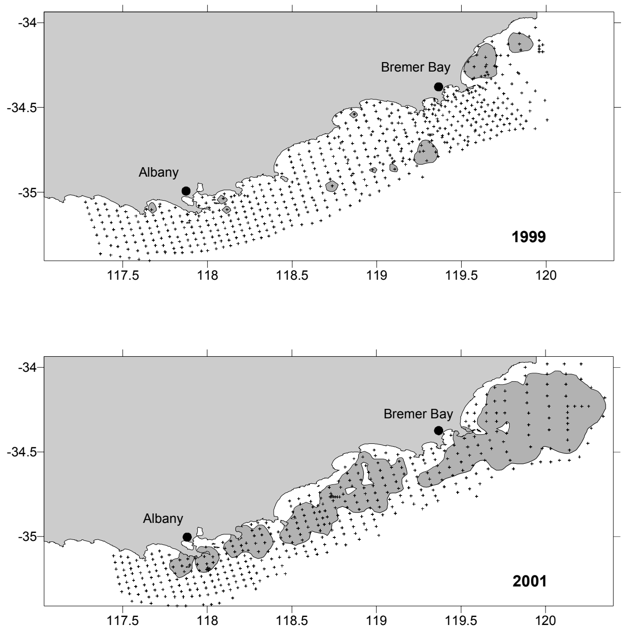

Fig. 6. Spawning area of *Sardinops sagax* at Albany and Bremer Bay as determined by the presence of eggs <24 hours old collected using plankton nets. Crosses denote zero catches of eggs. The spawning area reflects the spatial range of *Sardinops* and is used as a surrogate indicator of relative stock size. (a) June - July 1999; (b) June – July 2001.

STOCK RECOVERY

After the mass mortality event, the *Sardinops* stock at Esperance was still sufficiently large to allow continuation of purse-seine fishing, albeit at a reduced level. However, the *Sardinops* spawning biomass at both Albany and Bremer Bay was too low to allow any commercial take in 2000 and 2001. In order to assess whether there had been any appreciable recruitment, a small group of commercial vessels attempted to obtain samples of *Sardinops* in Bremer Bay and Albany between October 2000 and March 2001. Schools of *Sardinops* were observed at Bremer Bay irregularly during this period and several samples were obtained. However, unsuccessful searches for adult *Sardinops* through the traditional Albany fishing grounds, by competent purse-seine industry members, indicated that until March 2001 there were very few schools of *Sardinops* in this region; none could be located and a catch of 38 individuals was, at the time, considered an extremely positive sign for the recovery of the fishery.

Three months later, in July 2001, another DEPM survey found a widespread distribution of spawning adults in the Albany region, and extending across to east of Bremer Bay (Fig. 6b). Increases in spawning area, a relative indicator of stock size, in both of these regions support the notion of stocks that had undergone strong recovery. As part of the DEPM survey, commercial vessels located and sampled schools of *Sardinops* in the traditional fishing grounds. These fish were predominantly two to five year olds. Given the very low levels of residual stock in Albany after the 1998/99 mass mortality, and the appearance of several cohorts of spawning-age fish over a three-month period, it is apparent that these fish were entering the Albany region as migrants rather than as 'traditional' recruits. This indicates that the influx to Albany did not solely from a return migration of recruits that were derived from the residual spawning biomass. Rather, there was an influx of both adult and recruit stages, the majority of which very likely came from east of Bremer Bay. The proposed nursery grounds east of Esperance were most likely the source of recruits to the Albany and Bremer Bay regions. However, the source of adults needs to be further considered.

REVISITING THE FDAA HYPOTHESIS AND FINALIZING THE SPATIAL RECRUITMENT MODEL

Gaughan *et al.* (2002) suggested that further examination of the time series of age data for southern WA *Sardinops* would help elucidate the population dynamics and in particular address the question of whether the spatial dynamics can be expected to change as the size of the entire south-coast breeding stock fluctuates. The observed eastward migration of large quantities of *Sardinops* into the Albany region in early 2001, which contrasts with the FDAA hypothesis developed from data collected prior to the 1998/99 mass mortality event, indicates that the answer to this question is "yes".

Gaughan *et al.* (2002) further claimed that owing to the presence of FDAAs, migration of *Sardinops* between regions in southern WA could not be relied upon to reduce the impact of any localized over-exploitation. The large-scale migration of *Sardinops* into the Albany region described here indicates that the above hypothesis is flawed in that it did not account for the possibility of changes in spatial dynamics relative to population density. The migration-driven recovery at Albany was possibly the result of decreased competition for food; this could have freed up resources, thereby creating an advantage to move into the area. This is contrary to the spatial population dynamics during the 1990s when larger concentrations of *Sardinops* along much of the coast apparently negated any advantage in migrating. Thus, while there are limited benefits in alongshore migration when stock size is large, there may be benefits in such movement when stocks size is low. That is, a localized depletion may relax intra-specific competitive exclusion, thereby enhancing the possibility of migration into the depleted region.

Recognition of the potentially large role played by productivity levels and resource availability allows a third component of the spatial recruitment model to be conceptualised (Fig. 7). This component of the model suggests that productivity levels influence the distance that an average fish moves either regularly for a specific purpose (i.e. access to specific feeding/breeding grounds) or randomly over the longer term, perhaps in an ongoing search to increase fitness. Given a highly productive environment, there would be more scope for energy to be used for exploring new territory and, if so, then a more highly mixed population would be expected. By contrast, populations in low-productivity habitats would probably mix less through their range. This relationship could be either linear or non-linear, and either with or without a threshold level (Fig. 7).

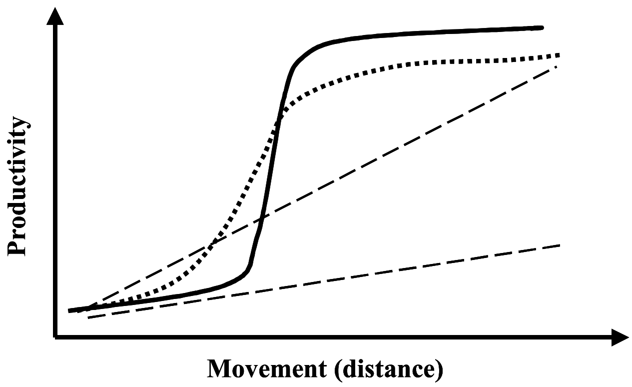

Fig. 7. Conceptual representation of linear, non-linear and threshold relationships between environmental productivity and the distance an average fish moves over its life. This model suggests that mixing rates within the range of a stock may be partly a function of the productivity (both the magnitude and distribution) of the environment that the stock occupies.

Regardless of the form of potential relationships between rates of movement, productivity levels and competitive exclusion, the important point with respect to APAs is that mixing rates within the range occupied by a stock may well be resource dependent and therefore may change in response to stock size. I contend that the contrast between the FDAA form of the spatial dynamics of southern WA *Sardinops* and the migration-driven recovery of the *Sardinops* stocks at Albany provides evidence for the above hypothesis. The relative contribution of each FDAA to overall levels of recruitment and population size is still not known but it does now appear as if significant depletion in the Albany region can be replenished, eventually, by movement of both adults and recruits from the more eastward regions. I further contend that the magnitude of the *Sardinops* abundance available to migrate across to Albany and Bremer Bay partly resulted from the historically lower exploitation rates in the Esperance fishery, combined with the portion of the *Sardinops* continuum that resides in the extensive unfished grounds east of Esperance, as was observed during the mass mortality event of 1998/99 (Gaughan *et al.* 2000).

CONCLUSIONS: THE NEED FOR A LIBRARY OF CASE HISTORIES

The temporal component of the spatial recruit model highlights the question of the efficacy of an APA: what looks appropriate at one level of recruitment may not be so for another level of recruitment. This argument also applies to variation in stock size. The suite of three hypotheses that I have called the spatial recruitment model, and which includes the so-called temporal component, has been generated from an explorative examination of fishery, fishery-independent and biological data for a small pelagic fish that is targeted in coastal waters. The understanding of the spatial recruitment dynamics integrates data collected over 14 years. Importantly, it was only in the fourteenth year of dedicated research that the detection of a non-linear response was able to provide contrast to the picture developed in the first 10 years of research. Isolated studies of less than 10 years' duration may fail to detect such non-linearities and thus may never be able to answer even basic questions relevant to placement and size of APAs.

What aspects of the spatial dynamics and sudden recovery of *Sardinops* at Albany and Bremer Bay are relevant to the question of whether APAs are useful for managing coastal small pelagic fish? A key element, introduced at the start of this paper, is the movement of fish from APAs to regions where they can be exploited: what was the source of the influx of adults and recruits to Albany and Bremer Bay? It is feasible that the unfished parts of the continuum of *Sardinops* provided a source of adult migrants that initially supported the Esperance fishery and eventually contributed to the *Sardinops* recovery at Bremer Bay and Albany. Furthermore, the unfished region east of Esperance is also an important nursery area for the *Sardinops* continuum between Esperance and Albany. Thus, not only did a large unfished area act as a default-APA with respect to the vulnerable age classes, but nursery grounds were also 'protected'. Both the large size of the unfished area and the fact that it encompassed a significant nursery area fall within the key benefits of protected areas summarized by Guénette *et al.* (1998) and Sumaila *et al.* (2000). Both of these works also demonstrated that spatial reserves may provide resilience, in the form of a reserve stock acting as an insurance policy, against stochastic and unforeseeable events; recruitment failure of *Sardinops* in the Albany region combined with mass mortality qualifies well as an unforeseeable stochastic event.

The hypothesis that a default-APA has already provided substantial benefits to the *Sardinops* fisheries at Bremer Bay and Albany could be further addressed, for example, through examination of nuclear-DNA studies on currently available tissue samples that cover the spatial and temporal aspects of the *Sardinops* recovery. However, are further studies really required in a case such as that presented here? Given that scientific advice for fisheries management purposes is inherently imprecise, expecting precise answers in the APA context is unrealistic. The theory behind why APAs can form an important part of fisheries management is well developed. I have attempted here, albeit at a superficial level and from a fishery viewpoint, to show that there are sufficient data to test the theories. At least, a logical sequence of evidence has resulted and will subsequently form part of the scientific advice presented to resource users and to potential investors in the *Sardinops* resource in southern WA.

There are undoubtedly other fisheries around the world for which data, qualitative information and hypotheses could be recast in the context of addressing potential benefits of APAs as a fishery management tool. Examples of exploited stocks for which unfished portions act as default-APAs need to be identified, catalogued and subjected to meta-data studies so as to search for those similarities that provide an indication of what works (e.g. relevant spatial scales) when planning APAs. A library of a broad spectrum of case histories would provide the basis for making informed decisions on individual APA cases without the need to wait for 1–5 years of research in each case. Development of such a library could take years off the process whereby APAs enter the mainstream fisheries-management toolbox. Given the poor global record of fisheries management (e.g. Cochrane 2000; Scheiber 2001), a management-orientated study of available fisheries data as suggested here would appear to be good value for money – let us first use what we have before embarking on expensive field studies whose outcomes may yet be of debatable relevance in the decision-making process.

ACKNOWLEDGEMENTS

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WHERE DO MARINE PROTECTED AREAS FIT WITHIN AN ECOLOGICALLY SUSTAINABLE DEVELOPMENT FRAMEWORK? A WESTERN AUSTRALIAN PERSPECTIVE

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Abstract

Ecologically Sustainable Development (ESD) requires consideration of the ecological impacts of activities, along with their social and economic costs and benefits, plus the governance arrangements employed. Many fisheries in Australia are now implementing ESD principles to meet a growing number of government and community requirements. Given claims about the importance of Marine Protected Areas (MPAs) to the management of marine resources it is important to analyse how these systems relate.

It is argued, given the wide variety of management tools needed to effectively manage most species within marine habitats, the main use of complete no-take MPA's in Western Australia will not be to meet sustainability objectives. Rather, they will mostly be used to meet the social and governance objectives of ESD, specifically, the allocation of access to stakeholders that want to protect areas from any exploitation. This includes eco-tourism operators and divers, who want direct access to such areas, plus conservation groups who may not require direct access.

Despite not catching these resources, effective regional management will require such no-take areas to have a specific allocation of "access" in addition to the catching sectors. Taking such an holistic approach will be necessary for the implementation of the integrated fisheries management initiative that has begun in Western Australia. Moreover, treating the debates about the establishment of no-take areas as allocation issues rather than about the best way to manage natural resources should make their resolution easier.

Keywords: sustainable development, no-take areas, fisheries management, governance, allocation of access, ecological benefits, social benefits, ecosystem management

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INTRODUCTION

During the past 10 years, a major initiative of many governments has been to implement the concept of sustainable development. Within Australia, this has been termed Ecologically Sustainable Development (ESD) and is defined as '*using, conserving and enhancing the community's resources so that ecological processes, on which life depends, are maintained, and the total quality of life, now and in the future, can be increased*' (CoA 1992). The main principles of sustainable development require the protection of biodiversity, maintenance of ecological processes, and provision for intergenerational equity.

Many fisheries agencies in Australia are now actively seeking to implement ESD principles within their management arrangements to meet a growing number of government and community requirements. A National ESD Reporting Framework has been developed for wild capture fisheries to assist in this process (Fletcher *et al.* 2002). The Framework recognises that sustainable development, within a fisheries context, requires the addressing of issues beyond the target species, by examining the impacts on any bycatch, the habitats where the fishery operates, and other ecological processes that may be affected. There is also a need to explicitly examine the social and economic outcomes of the activity and ensure that the elements related to the effective governance of the fishery are included. Therefore, successful management of a fishery to meet ESD principles requires integration of the environmental, social, economic and governance factors. This can be described as "beyond the triple bottom line".

Concurrent with these sustainable development initiatives, there has been a strong momentum by many environmental groups for the widespread establishment of no-take Marine Protected Areas (MPAs). These are said to assist with the protection of marine biodiversity and ecosystem functions and are being promoted as a necessary component for the effective management of fisheries (e.g. WWF 2002). Such endorsements often claim that traditional fisheries management has failed and that MPAs provide a more 'ecosystem based' approach to assist in the sustainability of these harvested stocks and general biodiversity conservation. However, the evidence to support such views largely relies on theoretical studies of potential benefits, and there are few empirical data at the spatial or temporal scales required that document fishery benefits (Ward *et al.* 2001).

Given that the objectives of the two concepts of ESD and MPAs have a high degree of similarity, it was considered appropriate to assess the overlap and relative effectiveness of MPAs across the three main elements of ESD (ecological, socio-economic and governance). Although there is a variety of MPAs, only complete no-take areas will be examined here because this type causes the most controversy and for many conservation groups is the only MPA worth having. Consequently, the relative benefits of implementing no-take areas across the different stakeholder groups also need to be identified.

Analysis of the relative benefits and beneficiaries of MPAs will be completed from the perspective of the fisheries resources and management arrangements currently in place (and planned) within Western Australia (WA). The situation within this jurisdiction is probably not typical of that present elsewhere in the world, but it is appropriate that an analysis of the benefits of no-take areas should occur within what is often described as a system of well-managed fisheries, e.g. the western rock lobster fishery, which was the first to obtain Marine Stewardship Council accreditation.

CURRENT PARADIGM VERSUS LIKELY OUTCOMES

The generally held assumption for implementing marine no-take areas is that most benefits would be apparent within the ecological components of ESD (Fig. 1). The proposition is that such areas will result in increase productivity and biodiversity and assist the sustainability of the harvested species outside the MPA (e.g. Dayton *et al.* 2000; WWF 2002). Only a small amount of attention has generally been paid to any potential social or economic benefits/costs that may accrue from these areas. The recent publication from WWF (2002) does, however, recognise that these areas may "*contribute to the social and cultural values of local communities*". The value these areas might contribute to the governance arrangements is, however, rarely acknowledged.

Within WA, the benefits that are likely to emerge from the formation of no-take areas will differ substantially from this paradigm. Although there will be some environmental benefits, it will be argued below that these will be relatively small compared with the effects on other categories of ESD. It is expected that one of the major benefits (if such a system were to be introduced), would be for governance, especially in assisting with the allocation of access amongst the various stakeholder groups (Fletcher and Curnow 2002).

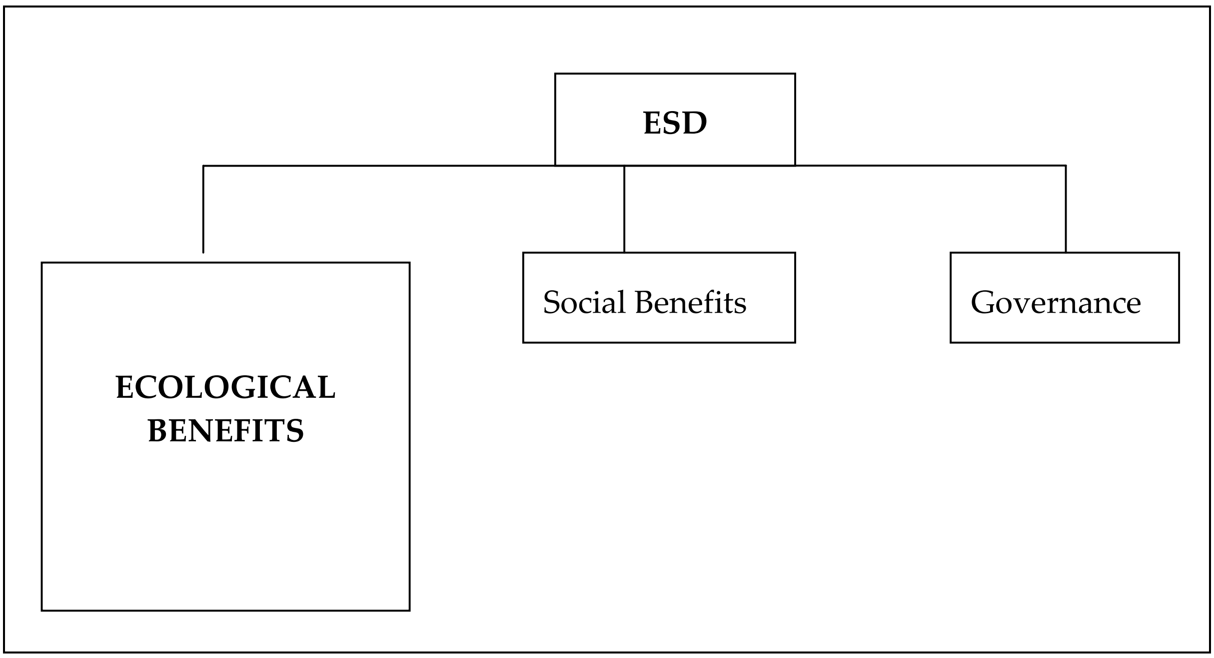

Fig. 1. Currently assumed level of relative benefits from no-take areas across the three main ESD categories.

Such a system could be valuable in providing social outcomes for those groups not normally accommodated by traditional fisheries management processes or allocations (such as passive users).

WHY IS THERE SUCH A DISPARITY?

There are three main reasons for the discrepancy between the presumed and likely benefits of MPAs across the components of ESD. The first comes from a general misconception about the way protected areas operate within marine systems compared with those in terrestrial systems. The second is a function of the local management arrangements already in place within WA. The third relates to the premise that there will always be an intrinsic value in establishing an MPA.

Misconception 1 – Terrestrial v. Marine systems

Terrestrial Systems - Much of the general population's assumption that a large ecological value will result from MPAs comes from their experiences within the more familiar management of the terrestrial environment. There are, however, a number of fundamental differences between these systems and their management, and these affect the level to which direct parallels can be made.

On land, most production is "benthically" derived, with the main source of nutrients coming from the soil. Most terrestrial communities, particularly the dominant plant communities, are heavily structured by the geo-physical properties of the terrain. Larger fauna that live in these resultant ecosystems may have relatively low rates of effective movement out of these areas. Thus, many of these ecosystems can be described as being relatively "self-contained" and can therefore be delineated relatively effectively by lines on a map. Moreover, in some cases these areas can be fenced to keep "things" in and keep unwanted things out.

Within the terrestrial environment, most human development relies on the removal or replacement of natural ecosystems, or at least their substantial alteration. For example, development of towns and cities requires removal of the natural environment to impose the houses, roads and other infrastructure. Moreover, agricultural production within most countries is not a 'natural' activity. Within Australia, this usually entails the removal of the native habitat and ecosystem and replacing it with an exotic (normally Northern Hemisphere) crop or ranching mono-specific system. Such activities have led not only to the removal of much of the native vegetation and associated faunal communities but also to significant damage to the surrounding ecosystems through agricultural run off and salinisation (NLWAA 2000).

Delineation of areas that lock out any extractive or destructive terrestrial activities is seen as the only effective way of maintaining parts of the natural terrestrial ecosystem that can continue to function in isolation from the disturbed areas outside. Consequently, terrestrial National Parks are mostly no-take and are often (but not always) successful in maintaining many aspects that people expect, including wilderness values, maintaining elements of biodiversity and natural ecosystems.

Marine Systems - Marine systems are much more "fluid" and three-dimensional than terrestrial systems. The basis for production is more "pelagically" derived, with oceanographic features being of much greater importance than the substratum present in an area. Much of the primary production in the oceans is not even linked to the substratum. Moreover, most marine animals and plants have inbuilt dispersal mechanisms during their larval phase. Overall, mobility is much greater than for terrestrial systems. Consequently, there is much greater leakage from any single area and a concomitantly greater dependence on other areas for the sustainability of many species. Organisms cannot be fenced in or out in the marine environment.

Given these relationships, merely having isolated sections of the coast where extractions are prevented will not, by themselves, ensure the sustainability of most of the biological components. Furthermore, any no-take area will not be self-contained unless it is very large or the objectives relate to only a few specific components (i.e. sedentary marine species without a planktonic larval phase).

Misconception 2 – Fisheries production and existing management arrangements

There is a general misconception within the community that all fishing activities are allowed to occur everywhere. In WA, and increasingly through all parts of Australia, this is not the case. For WA, there is already a comprehensive system of specific fishing closures and other regulations dating back to the 1960s that have been implemented to ensure the sustainability of the harvested resources and to guarantee their ongoing production. In addition, all fishing methods that may have direct effects on the seafloor and the broader environment are already heavily regulated. For example, dredging has not been permitted anywhere in WA since the 1970s, and the locations where otter-trawling can occur are heavily restricted such that this occurs in less than 5% of the total trawlable habitat.

This approach is consistent with the concept proposed by Walters (2000) that fishing be allowed to occur in only a relatively small number of places. By contrast, in many other regions of the world, trawling is allowed to occur everywhere except in a few regions.

Unlike terrestrial agriculture, fisheries production actually requires the natural habitat to be maintained and the ecosystem to continue to function. The species that are harvested are heavily dependent on the productivity of the ecosystem, and therefore if the ecosystem is altered substantially they will also be affected. There are numerous examples where this has occurred through direct impacts of over-fishing (e.g. Pauly *et al.* 1998), through other human-induced changes (e.g. impacts of land clearing and other activities on coastal waters (Cappo 1998) or through long-term natural changes in oceanographic conditions (e.g. the regime shifts between anchovies and sardines (Schwartzlose *et al.* 1999). Thus, when the natural ecosystem is affected, so too is the fisheries production (through catch levels and/or the species composition of the catch).

In conclusion, the large number of management arrangements already in place within WA greatly diminishes the ecological benefits that imposing additional complete no-take areas would have on the sustainability of all components.

Misconception 3 – Imposing a no-take area can't hurt"

The assumption that closing off an area must be good for the environment has its problems because no-take MPAs are not a universal panacea and in some circumstances may even exacerbate problems by the transfer or concentration of fishing activities. The unquestioned implementation of any hypothesis is unacceptable. The determination of whether (or how) a no-take MPAs should be established needs to be examined just as carefully as would any doctor before prescribing medicine, because there are dangers in assuming that there will be no contra-indications. A no-take MPA can result in anything from a beneficial to a negative impact depending upon the circumstances and the biological attributes of the components. Consequently, establishing no-take areas should only be considered as part of an overall scheme of management, not instead of, or in competition with, other arrangements operating in the region.

REAL ECOSYSTEM MANAGEMENT – FITTING MPAS WITHIN OTHER MANAGEMENT SYSTEMS

Complete no-take closures are a very coarse method of fisheries management. Being area based, they must compromise what is, or is not, protected given the vastly different 'footprints' that each species within the region will have. If they are used as a major method of fisheries management, they are unlikely to maximise the overall benefits to society because of the compromises that this will entail. Such an approach would not be consistent with meeting the principles of ESD. To achieve sustainability within marine systems requires a sophisticated suite of management arrangements that for fisheries will include species- or catch-based rules (such as size limits, seasonal and area closures and quotas), and a suite of gear- or effort-based rules (including the type of gear allowed, areas of operation, total effort, seasons of operation, etc.)[1]

The management systems that result from these considerations are likely to have different but overlapping boundaries that are not amenable to being defined within a single closed area. For example, within the Shark Bay region of WA, more than half the region is permanently closed to trawling and other parts have seasonal closures (Fig. 2). In addition to these trawl closures there are separate closures, bag and size limits for the recreational snapper fisheries in the Eastern Gulf and Western Gulf. In the offshore areas, there are quotas on the commercial catch of snapper, and size limits along with bag limits for the recreational sector. These snapper closures and regulations have different boundaries to the trawl closures because of the specific biological differences and processes being managed. Moreover, these boundaries are for only two of the fisheries; many others operate in this region.

The result is a multi-layered mosaic that helps maximise the benefits to society and deliver the best sustainability solution for the entire bioregion. The only way a no-take area alone could accomplish these sustainability outcomes would be the complete closure of the entire region, which would reduce the annual income generated by the region by about $A100million.

[1] This recognises that activities managed by other sectors (e.g. oil and gas, shipping, etc.) will also require suitable plans and there are already comprehensive sets of regulations.

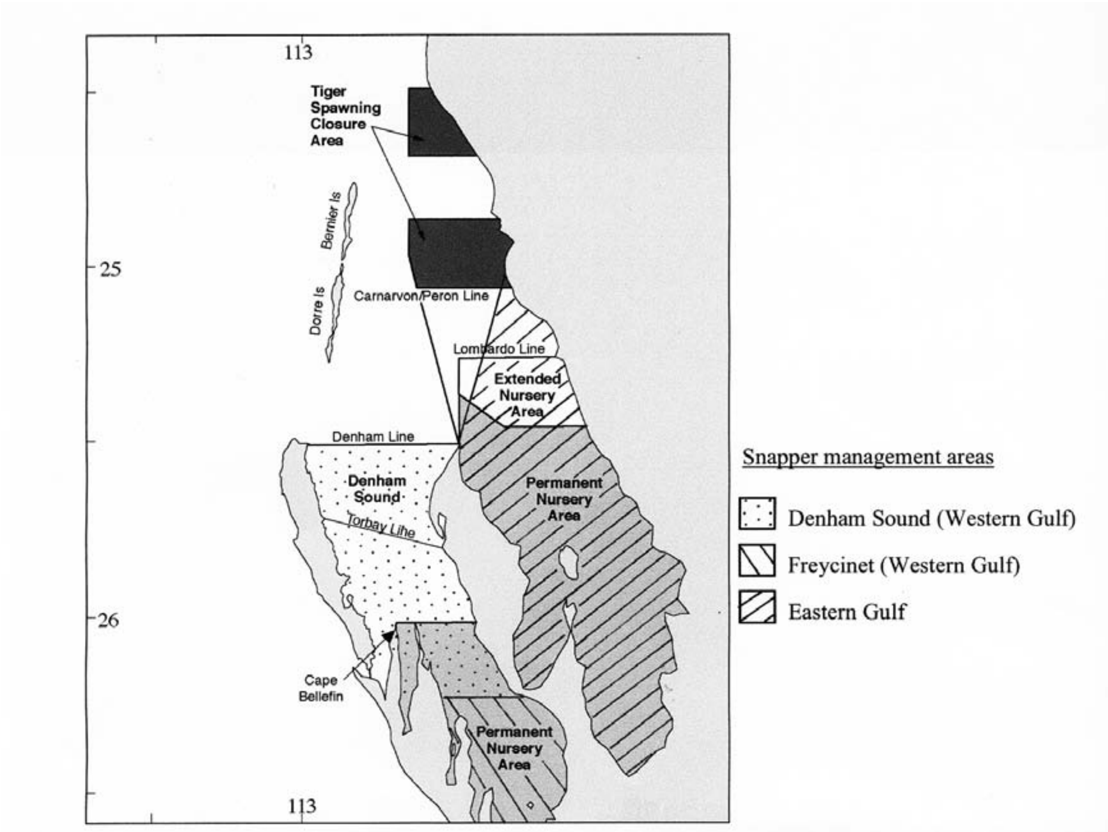

Fig. 2. Shark Bay, showing the management zones for prawn trawling and snapper fisheries.

WHO ARE THE BENEFICIARIES OF NO-TAKE MPAS?

Extractive users

In situations where the fisheries are being well managed, there will be very few cases where a no-take area will increase the total production of a range of target species from a bioregion. If the species is not being overfished (such that the spawning biomass is already above an appropriate threshold limit), any extra egg production that results from having more and/or larger individuals within the no-take areas will not increase subsequent recruitment levels[2] and, in some cases, e.g. abalone, it could cause a decrease (Ricker-type stock–recruitment relationship). So there will be no extra "productivity" provided by the MPA.

Spillover effects have been mentioned as being a potential benefit from no-take areas (Ward *et al.* 2001). If occurring at a significant level however, the rate of emigration of the species from the area must, by definition, be rendering the MPA largely ineffective. Furthermore, any "increases" in catch from spillovers are unlikely to be fully compensating for the total loss of catch from the excision of the entire no-take area. At best, this is not an increased benefit but a lessening of the impact.

Finally, for relatively sedentary species, if a no-take area were introduced, the total catch/targeted effort would have to be proportionately reduced to compensate for the areas no longer accessible. For abalone fisheries, where the rate of migration out of the areas is virtually zero, the loss of available catch will be directly related to the area of reef present within a no-take zone. Under such circumstances, not reducing the catch to account for this loss of area could have serious implications for the sustainability of the resource because of the increased exploitation rates that would be applied to the stocks outside the closed area (Haddon and Buxton 2003).

In situations where the stocks are not currently overfished, the lack of real benefits to the extractive users (commercial, recreational and indigenous fishers) means that they will not obtain any advantage from the introduction of complete (non-specific) no-take areas. Consequently, when the argument has been put forward that the main reasons for imposing a no-take zone are the benefits to them, e.g. the recent

[2] Unless this area is known to be a good source zone (*sensu* source–sink) and this pattern does not shift among years.

introduction of MPAs in Victoria, they object strongly.

Non-extractive users

The lack of significant benefits to extractive users within WA does not mean that there are no beneficiaries from the implementation of no-take areas. Instead, the beneficiaries will be the large number of non-extractive stakeholder groups, including:

- Eco-Tourism operators/Divers – who would benefit from the increased densities and likely larger individuals in these areas, plus the promotional value of this no-take concept;
- Local Community Groups – no take areas can be used to protect regions of high social value;
- Researchers – for some species/systems (but not all), having suitable no-take areas can assist in the determination of biological parameters and processes (e.g. natural mortality, reference areas);
- Conservation-minded people – who may not physically go to the areas but "like" the fact that there are areas where no exploitation is occurring; and
- Everyone – when the no-take area is protecting a truly unique area.

PROMOTING THE MAIN BENEFITS OF NO-TAKE AREAS IN WA

Within WA, it is possible that no-take areas will be used as one of the main mechanisms to provide resource access to the non-extractive stakeholder groups. These groups have traditionally not been considered in allocation debates, which have largely been restricted to the commercial or recreational fishing sectors. Area-based allocation tools, such as no-take MPAs, are likely to be the only effective approach to providing access shares to these "non-extractive" sectors (Fletcher and Curnow 2002).

An inclusive approach will be necessary for implementing the integrated fisheries management initiative that was begun recently in WA (FWA 2000). This initiative requires all stakeholders, including commercial, recreational and indigenous interests, divers, conservation groups, etc., to be included in the management arrangements that are developed for a bioregion. Consequently, if consultation on the identification and development of no-take areas is conducted as part of an overall allocation process amongst these groups, rather than using the argument that this will be necessary for the sustainability of fish stocks, this should reduce the volatility of each debate.

If no-take areas are implemented as part of an allocation process, this will require that there are appropriate compensation mechanisms for the commercial sectors in circumstances where they are required to give up access (either in the form of a reduction in TAC or in the level of effort or access to a fishery). This compensation is to reflect that there has been a shift in the level of allocation. The issue of compensation was the main area of contention for the establishment of the system of MPAs in Victoria. The first attempt to get the legislation (which did not contain any provisions for compensation) through the Victorian Parliament was blocked as a result of this perceived lack of fairness. It was eventually passed when the commercial fishers were provided with some level of compensation.

Unlike commercial fishers, it will be harder to "compensate" recreational anglers for any loss of their access because they do not have individual allocations akin to 'property/access' rights tradable on the open market. In many places, however, this group already has specific "recreational only" zones where commercial fishers are not allowed. Ultimately, this system should be seen as a means of sharing the common resource amongst competing user groups. The recreational fishers are, however, only one of the user groups; no-take groups also have some right of "access". Obviously, effective consultation on the size and positioning of any no-take zone for "observational" purposes would be required to ensure that the overall benefits amongst the groups and society as a whole are optimized – this is ESD in operation.

CONCLUSION

There is no doubt that complete no-take MPAs are going to play a role in the management of marine resources and the implementation of ESD. They will be, however, just one tool of many needed for an acceptable level of performance across the three main components of ESD to achieve the new triple bottom line (Fig. 3).

Within WA, ensuring appropriate ecological performance will be primarily based on the comprehensive system of fisheries management (including multilayered -Targeted Fishing Closures –see Newman *et al*. 2003) already in place or under development (which includes a large number of specific fishing closures). Such comprehensive and sophisticated systems of management are seen as providing the best outcomes for society and the demonstrated benefits for the longer-term sustainability of the resources. Complete no-take areas can only play a small part in achieving these ecological goals.

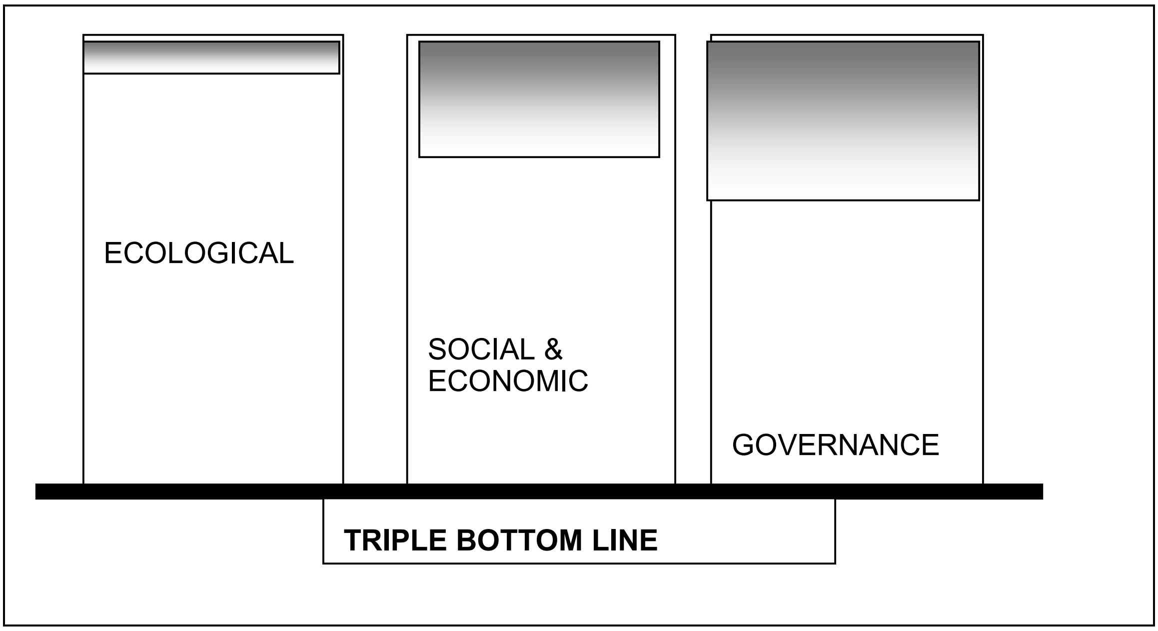

Fig. 3. Relative benefits of MPAs (shaded areas) for implementing ESD and achieving the new triple bottom line within Western Australia.

In other regions of the world where the management of fisheries activities may not be so comprehensive or effective, it is likely that no-take MPAs could play a greater role. Thus, most of the examples where large impacts have been seen from the establishment of an MPA come from countries where basic management arrangements are almost absent (e.g. Russ and Alcalca 1989) or generally ineffective (e.g. Dayton *et al.* 2000). Even in these situations, MPAs alone are unlikely to be sufficient in the longer term except to prevent growth overfishing on the fish stocks present in these areas.

The benefits that no-take areas can have within the social and economic elements of ESD are only just being recognised. For example, eco-tourism is a growing industry in many parts of the world, and having areas where there are minimal impacts from other users would be one of the main selling points of no-take MPAs (e.g. Williams and Polunin 2000). Their assistance with the governance of management arrangements for marine resources is, however, where they are likely to make the most impact over the coming decade. Changing demographics and community attitudes to the environment will require governments to consider the non-extractive users as legitimate stakeholders who require an effective and explicit allocation of resources. No-take zones should play a large role in accomplishing these objectives.

Finally, acknowledging the limitations of no-take areas as a fishery management tool and recognizing where their true advantages lie will assist in their wider acceptance amongst stakeholder groups. Thus, treating the debates about the establishment of no-take areas as allocation issues rather than a philosophical argument about the best way to ensure sustainability of resources should make their resolution easier.

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ABORIGINAL CULTURAL SUB-REGIONS AS SURROGATES FOR BIODIVERSITY MOSAICS IN CAPE YORK, AUSTRALIA – TOWARDS RECONCILIATION OF MANAGEMENT VALUES AND ON-GROUND REALITIES

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Abstract

Cape York is the ancestral and present home of numerous Aboriginal Australian groups, each having resource and cultural rights and obligations to particular geographical areas according to traditional law and custom. These laws and customs are recognized in the common law of Australia under the *Native Title Act 1993*, which provides a new basis for the legal recognition of Aboriginal domain and rights. The historical perception that the Cape represents an unspoiled 'wilderness' area is changing, and the change has brought with it pressure to include Cape York's marine and freshwater aquatic resources into reserve-system frameworks. The entire east coast of Cape York falls within the Great Barrier Reef Marine Park, the largest in the world, proclaimed under the *Great Barrier Reef Marine Park Act 1975*. The Park has world heritage status, carrying obligations that governments of Australia are required to address. Among these, 'no-take' protected areas have broad and far-reaching implications for indigenous peoples. Historically, biodiversity parameters have been employed on a worldwide basis as a fundamental axiom for the establishment of CAR (comprehensive, adequate and representative) protected areas. This paper puts forward a case for using subregional 'people' mosaics as the primary basis for identifying and negotiating protected areas. It discusses what might drive selections made within those areas. It is argued that the issue of sustainable use is central, not only to biodiversity protection, but also to the survival of cultures that are not homogeneous and that pose cultural CAR questions in themselves. Bioregional maps and maps of Aboriginal groupings are available. What can be made of this information and how can it support mutually acceptable outcomes?

Keywords: aboriginal, cultural, sea, biodiversity, bioregional

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OPENING NOTE

We acknowledge the traditional owners of this area, the Gimuy-Yidinji people, not only as a sign of respect, but as a reminder that all parts of Australia are spoken for in an indigenous traditional sense. Legal channels for indigenous rights and aspirations have only recently been opened through the Mabo and Croker Island Federal Court cases (Mabo and others *v*. Queensland [no 2] 1992 175 CLR 1 and The Commonwealth *v*. Yarmirr, Yarmirr *v*. Northern Territory (2001) HCA 56).

As you might appreciate, this is an unusual forum for us as an Aboriginal traditional owner organization to be speaking at, yet this ought not to be the case. Our absence from such meetings reflects a distance between indigenous and non-indigenous thinking and a resourcing gap in the ability of indigenous people to engage with government agencies and academia involved in protected-area planning and implementation. We hope to point to a substantial potential for collaboration between two differing but valuable knowledge systems, which can build protective area regimes that suit all parties. The indigenous presence here at the Congress also signals a new consciousness developing both nationally and globally. This new consciousness is coming to realize the damage that has been inflicted on our environments when social and cultural interpretations of country are under-valued in favour of economics. It is ironic that indigenous peoples of Cape York are for the first time getting the opportunity to develop economically, at a time when such development may be curtailed by environmental considerations. Mainstream managers are trying to wind back ecological impact while indigenous people are yet to start benefiting from commercial resource use. Protected areas are therefore a double-edged sword for indigenous people. On the one hand they protect land and sea from exploitation but they also limit the ability of indigenous people to make a living by developing economies that enable them to interact with broader society. The new consciousness we refer to is also belatedly beginning to recognize the existence of indigenous peoples, the relevance of this amazing cultural diversity, (some 6000 languages worldwide) and the need to depart from strictly biological methodologies to arrive at sustainable use of natural resources. The application of foreign farming methods, introductions of foreign species, and reluctance to take on board local knowledge have clearly caused cultural and biological mayhem in this 'Age of Development'. Endemic species, cultures, and natural interactions of these two are under serious threat in many locations around the world. These include a significant component of native foods, medicines and cultural resources under siege from patent-seeking biotechnology developers. There are some indications that the 'Age of realization' is dawning also – realization that globally we must take stock of our direction, that indigenous people have a great deal to offer, and that we need to resource the incorporation of western knowledge into indigenous management systems.

We presume your interests involve predominantly taxonomy, physiology, reef ecology, ecosystem modelling, bioregional planning and protected area management. We wish to volunteer some thoughts of our own that we think should become a greater part of the scientific psyche and recipe for protected areas planning, namely social and cultural units of management.

We believe that mainstream understanding of indigenous science, and the indigenous understanding of western science, are central to changing the indigenous predicament. If protected areas can offer indigenous people benefits we will support them. We have reached, indeed passed; the point where keeping cultures and people alive is as high a priority as protecting parts of their environment. This is a deadly serious matter for indigenous cultures. They cannot be expected to quietly die out while their domain is turned into "no-take" protected areas for the benefit of third parties. This sounds alarmist but it has happened in the past and continues today. There are balances to be struck and the resources required to achieve those are not equitable either in funding terms or in information terms. We, as a forum, are big on biology but small on social and cultural issues.

Frequently, committee structures provide governments with justification for policy and regulation. Policy then affects people on the ground: all people, those in cities and those on the land. Conservation policy is largely generated by population and education centres remote from the "saveable" environments in question. This is fraught with problems for people of the land and sea. Indigenous people have not yet successfully entered the decision-making loop (namely knowledge – advisory committee – government policy – legislation – then new knowledge). We hope to influence the way managers and politicians think about this major problem. Traditional owners, find themselves on the lower end of the priority list whilst they are out there on the spot and closest to the intended or proclaimed protected areas. Indigenous peoples must see the practical value of protected area legislation if protected areas are to gain support. Those formulating policy must not only understand the requirements of stakeholders and owners, it is crucial that the latter are developing the policies.

These are desperate times for indigenous people economically. We need to keep options open but also to protect country. We seek quality of life for the indigenous peoples of Cape York, and that includes developing economies and ensuring control of their social, cultural, economic and ecological environments. We seek to do this through integrated subregional planning. After several years of difficult and laborious discussions we have succeeded in building a framework of land and sea coordinators around Cape York. Subregions are a contemporary expression of the interests of traditional people and the issues pertaining to their lands and seas.

We seek to shift the 'protected area' rationale from a strong bioregional emphasis to a 'people catchment' or subregional framework that considers a triple-bottom-line outcome. It is important that we emphasize the 'triple bottom line'– positive outcomes must be environmental, economic and social. It is easy to pay lip-service to such a concept; however, when we compare the resources applied to each component, the social, cultural and spiritual criteria are usually undernourished, whereas we would argue that satisfaction of these is the most critical to successful protection. On Cape York, where 50% of the population is Aboriginal, practical and human realities might override purely environmental imperatives. The challenge is to recognize them and still provide for sustainable living and protection of both environment and cultures.

We as an organization are very much focused on outputs on the ground and hope the activities that you are involved in contribute to the development of effective, community-owned and community-driven management practices. We hope to steer what is currently called 'mainstream thinking and management' into addressing problems as they are seen by indigenous peoples, rather than solely to augment the western understanding of natural systems while preserving a narrow view of the world.

The recommendations of scientists in 1986 led to the declaration of the world's biggest 'green zone' over large tracts of Yadaigana and Wuthathi sea country in the far-northern section of the Great Barrier Reef. This decision reflected a poor understanding of indigenous reality and aspirations. The socio-economic implications of such declarations for indigenous peoples warrant serious consideration. The GBRMPA Representative Areas Program provides an opportunity to renegotiate the nature and extent of highly protected areas (HPAs).

The concepts in this paper can be applied to biodiversity-based planning generally. It is also important that protected areas on land are considered in discussion of aquatic protected areas because domains of indigenous peoples cover both land and sea. The intention is to stimulate thought and consider the principles that guide decision-making.

INTRODUCTION

It is pertinent here to note that our call for recognition and consideration has both moral and legal justification. It is as much about justice and governance as it is about protection of biodiversity. In 1992, the now-famous Mabo Case was heard and judged by the High Court of Australia. This was a claim to native title of the land component of the Islands of Mer (known as the Murray Islands to wider Australia) in the Torres Strait between Cape York and Papua New Guinea. The important outcome in this case was that prior to the case '*terra nullius*', the legal doctrine of an 'empty land', was considered to be a reality. If land and sea have not been recognized as belonging to anyone, or anyone belonging to it, it is easy to see how regulations and scientific principles have developed in isolation from the indigenous perspective. Indigenous peoples, already with highly developed knowledge of natural resource management, were simply not recognized and suffer the same dilemma even today, to varying degrees. Indigenous relationships and obligations to their country and its resources, as well as the application of indigenous learning and knowledge (dare we say science) have continued regardless of these legal developments.

The Mabo case overturned the myth of *terra nullius.* The court ruled that

1. the land was occupied and owned by Miriam people in 1788, and the Crown protected native title, and native title existed in common law;
2. many valid actions (under Crown law) have occurred since 1788, in dealings with the land by the State of Queensland and by Australian federal authorites. These actions have diminished the native title of Miriam people; and
3. what remains after diminution of these rights may then belong to Miriam people according to their particular law.

This case gave rise to the *Native Title Act*, which was passed in 1993 and thereby provided recognition of native title to all indigenous peoples of Australia. It is important to understand that native title already resided in existing law; the Mabo case exposed it. That is, native title was not invented to accommodate the situation, but was in the judgment of the court, 'embedded' in the existing legal position. It is also necessary to understand that each case is judged in relation to the traditional laws of the particular indigenous group involved and these are culturally and importantly spatially defined, giving rise to indigenous law for particular areas. Thus, the subregional approach we are taking on Cape York is a relevant one.

Native title has, is, and will always be claimed by indigenous people. In fact their 'native title' is much broader and more far-reaching than the limited descriptions provided for under Australian law. Native title is a legal concept based on a European understanding of transferable property. Indigenous peoples assert an underlying exclusive right to determine the use and future of their countries, including their sea component. This might seem unreasonable under a 'commons' principle generally held by Europeans, but it must be recognized as one end of the spectrum against which outcomes and concessions must be measured and negotiated. Being in control of country is central to the responsibility of indigenous peoples for their country. By 'country', indigenous peoples mean land, sea, sky, spirituality, culture, connection, everything (Langton *et al.* 1999 provide a useful description). For any group of indigenous people, this country is not Australia as a whole, it is the land and sea they belong to according to their law and tradition. It is the traditional owner's mandate and duty to speak for their land and sea, understood by indigenous people, and recognized by the *Native Title Act*. The traditional owners are the people who speak for certain geographic areas. The traditional owners might wish to delegate that authority but that is their decision.

2001 saw the first legal recognition of native title in the sea in the Northern Territory. Although it was a victory for indigenous peoples in principle, it provided for no exclusive use, no commercial use, and no limitation on fishing by outside interests. The decision did provide for subsistence use, access and recognition of native title limited by existing legal use of the sea by other parties (The Commonwealth *v.* Yarmirr, Yarmirr *v.* Northern Territory [2001] HCA 56).

It is relevant to acknowledge that the legal mechanisms for recognising the rights of indigenous people are very recent and some sectors of wider Australia are a little reluctant to accept the state of law that Australian courts are finding. The Yanner case for instance, found that indigenous peoples could take protected species (in this case crocodiles) by traditional right (Yanner *v.* Eaton [1999] HCA 53). Other cases of relevance exist and serve to indicate that important precedents continue to be set and that legal processes are revealing more and more indigenous rights. The danger for the management sphere is that actions may be taken that will be legally contestable and/or eligible for compensation.

We frequently find, however, that interests are not necessarily competitive and we offer the following advice:

- don't presume anything;
- keep an open mind;
- include indigenous people at the centre of planning from the beginning;
- find out what the aspirations of the parties are;
- provide adequate time and resources for each party to understand the other's motivations and positions;
- find out what the pressures and motivations are (sometimes they are political and/or research priorities with little consideration of social implications or relevance to commercial users and/or traditional owners); and
- be prepared to expand research proposals and thinking to include local peoples, to build capacity and to foster understanding of values and intent in both directions.

Can indigenous peoples offer a management unit concept that belongs to society as well as biology, a management unit that makes social, cultural and biological sense?

Indigenous peoples have been singing to some one else's tune for too long. Management practices and the knowledge informing them are so hugely skewed towards the western scientific understanding of natural processes that the spiritual components so central to indigenous thinking are hardly dealt with at all. The indigenous peoples' rationale for management and protection has not been seriously scrutinized by western science and has been marginalized. Management has not relied to any degree on 'Indigenist research', a new term. We draw your attention to the emergence of Indigenist research (see Martin 2000a for an introduction to the topic).

Australian and international law has recognized some indigenous rights and they must be accounted for in management, research and the planning and implementation of protected areas. Indigenous cultures have rules that must be respected if management is going to work. These rights and rules must be known to the planners. If there are issues with secret knowledge or intellectual property, the process must adjust by moving towards the information holders: the traditional owners must become the planners. This means allowing people with knowledge to devise management solutions that do not require them to divulge information to third parties,

THE SOURCE OF LAW AND PRINCIPLE

To simplify matters, we shall assume that there are just two visions: the western science and traditional ecological knowledge (Langton 1998; Langton *et al.* 1999; Posey 1999).

In the indigenous vision there are a number of central themes. For the purposes of this paper they might be seen as

- spirituality and connectedness with the earth,
- obligations to country under traditional law,
- kinship responsibilities,
- the vesting of decision-making authority with traditional owners of particular areas, and
- traditional ecological knowledge.

Western vision, it seems to us, is based on concepts of

- democracy,
- sustainability,
- economic growth,
- western-scientific endorsement of process, and
- a commons view of the sea.

We acknowledge that in reality there are overlaps and suggest that sustainability is a core aspiration of both visions. This sustainability, however, must include cultural sustainability or cultural survival and requires the involvement of expertise, research and information well outside present considerations regarding protected areas. People on the ground will not subscribe to any process that compromises identity, and it will be doomed to failure.

INDIGENOUS CONCEPTS OF IMPORTANCE IN MANAGEMENT AND PROTECTED AREAS

Spirituality. Although we are not experts on all indigenous peoples, the point we wish to make here is that if people are born, are initiated, live, and are buried on places the connection is powerful. Even if people are removed from their places, as they have been in many instances in Australia, bonds with country are still strong because this is what Aboriginal children are taught.

Obligations. Connection with country brings with it obligations to country. These include meetings on country, ceremony, speaking to land, sea and ancestors, teaching youngsters, respect for protocols, and so on. Paying respects to country and traditional owners is important. Indigenous people are obliged to follow certain procedures in certain places; certain people are not permitted at certain places. These are part of the religion and rituals of indigenous peoples. It is inappropriate for outsiders to know some of these things, but they are important considerations when gazetting protected areas, and must be taken into account if aquatic protected areas are to be relevant to indigenous people.

Kinship. Such relationships can be very difficult for non-indigenous people to understand, and they form the life's work of specialists in the field. Indigenous people live these relationships and appreciate them as a normal part of expressing connection among themselves, and between themselves and their countries (see Williams 1998, relating to the Yolgnu clans in east Arnhem Land, Australia; see also Langton 1998, recommended as prescribed reading for conservation managers wanting to deal with Aboriginal people).

In western society, there is an assumption of democratic process and an assumption that everyone has a right to comment on everything including a particular person's freehold property. However, freehold-property owners sometimes see interference by government and outside parties as an affront to their rights. Indigenous people have the same view of their country. Management plans created by outside interests potentially interfere with their property rights and their desired lifestyle. Even if a group of Aboriginal people had the wherewithal to purchase the land of another group, traditional rules would make it near impossible. The same applies to usage rights. Traditional owners must sanction the use of country by others or even by junior members of their own group. In our experience of Aboriginal order, even comment from non-traditional owners on someone else's country is frequently regarded as inappropriate. If we are not traditional owners, it is really none of our business. Conversely, everything that occurs on or in a traditional owner's countries is their business.

BIODIVERSITY AS A WESTERN CONSTRUCT

One of the present authors has been educated as a biologist and with very few exceptions it seems that everything experienced in mainstream biological scientific education is influenced by one pervading 'truth': Charles Darwin's theory of evolution and the various lines of enquiry that stem from it – speciation, radiation, variation, biodiversity, and the like. We are not denying the integrity of the theory but we are asking scientists to reflect on the other ways of knowing and understanding sustainability and coexistence with nature – that we need not be captured by a single explanation to the detriment of other, equally valid knowledge (Martin 2000b).

Western and Aboriginal worlds collide through the existing protected area processes defined through bioregions which, in turn, are based on an extension of Darwinian principles. The way western culture separates natural systems tends to be based on the plant and animal expressions of Darwinian process – ecosystems, gene pools, cohorts, biological units and sometimes just plain 'gravitational hydrodynamics' manifested in catchments. When such criteria are applied to 'making a living' on the planet, they are clearly inadequate. It is 'making a living' that is the primary issue for Cape York peoples and it is relevant that people's traditional country becomes a key, if not the principal, determinant in negotiation (if there is negotiation) of protected areas. But why does the debate on resource assessment and management stop short of including people and culture?

It can be strongly argued (if we are to take the biodiversity line) that cultures are an endpoint of a Darwinian evolutionary process, the outcomes of long-term trial and error. Is it not reasonable that the exquisite complexity of thousands of languages reflecting millennia of coexistence of people with their resources are born of survival of the fittest biologically and (importantly) socially and spiritually? To ignore this connection is possibly fatal to aspirations for protected areas. There are 5000–7000 languages spoken around the world (Posey 1999). We believe that languages are good indicators of cultural texture and warrant consideration as indicators of scale in community dealings. The process driving their evolution must be a combination of biophysical evolution and cultural evolution. These two elements must remain bound together; both are necessary considerations in practical management planning. People have evolved with their country.

In our view, the 'people' factor is critically important. As has been said many times, management is generally about managing people, not resources. We must progress beyond biophysical criteria and indicators, as is very slowly happening in matters regarding ecologically sustainable development (ESD), where social scientists are being asked to contemplate socio-cultural criteria and indicators. Are the natural ecosystems being considered by scientists in fact *human systems* in the indigenous view, with indigenous languages providing the deep-level interpretive tools and identifying integrated units of people and nature together? In view of the rich diversity of languages and knowledge held by indigenous peoples around the world, we believe it is crucial to learn from those peoples and account for their values in considering protected areas within their domains. We cannot reasonably expect a good result if we consult the knowledge in perhaps five major languages and are limited by one or two major theories, none of which are native to the country for which they purport to speak. Indigenous reasoning must be fully understood and or accepted before it is discounted out of hand or treated as (being provocative) an 'inferior' data set. In fact, we argue that indigenous interpretations are quite superior in the scheme of ESD, providing a holistic approach to looking after the land, the sea and their peoples and therefore sustainable futures.

BIOREGIONAL PLANNING

The first concession that needs to be made is that in the indigenous world certain people belong to certain country. This is sometimes a difficult concession to obtain.

It is clear that there is considerable biodiversity in Australia, particularly in the tropical north. Yet the people are isolated from their resources in so many ways: physically and economically (and the arrogant might say intellectually, but as we have discussed we need to define which intellectual framework we are talking about).

Biodiversity has been mapped during the IBRA (Interim Biogeographic Regionalisation of Australia) and the IMCRA (Interim Marine and Coastal Regionalisation of Australia) processes undertaken by Environment Australia (see Roberts and Tanna 1998 for comments on these regionalizations in relation to Aboriginal countries). The IMCRA provided the framework for an impressive bioregionalization of the GBR Marine Province, which provides the scientific rationale for the GBRMPA Representative Areas Program. We argue that a CAR (comprehensiveness, adequacy and representativeness) system is required for cultures on Cape York also. How can we protect these?

We now consider another depiction of diversity, this time cultural diversity – the AIATSIS MAP of Aboriginal Australia (Horton 1994). It has limitations and is distorted by loss of history but it gives an indication of the numbers of Australian languages. Clan or Aboriginal estates on the coast appear on the whole to be smaller than estates in arid country. This might imply that there are area-defined constraints in sustainable use, with coasts being 'richer' perhaps, as far as resources are concerned. Superimposed on biological constraints are complex social rules (Langton *et al.* 1999).

The IBRA offered 81 bioregions, and Aboriginal people could offer 350 coarse groupings (Roberts and Tanna 2000) – a higher-resolution framework for protected area planning if you like. But the truth of the matter is that the bioregions can be subdivided in the time it takes to select a couple of criteria on a GIS. However, all is not lost in the resolution argument, because indigenous people, too, can come down to families and even to the level of individuals as far as representing country is concerned. We might agree to call it a tie on the matter of sensitivity of the regionalization but defend our position that management based on cultural units might be more effective and lasting because it brings in the 'people' factor. It is the people who must live by the plan and therefore should be the ones engaged in its preparation.

The question we are asking here is, can a management unit based on cultural affiliations of indigenous groups be used as the basis for management and sustainable use – or 'caring for country' as it is called by Aboriginal people? Indigenous peoples want to look after their countries. 'White fellas' could also be said to be wanting to do the same thing through the ESD process and application of the precautionary principle. The affiliation of Aboriginal peoples with their countries is not a variable in the management equation; it is an immutable strong point around which their aspirations, rights, economy and culture pivot. Consequently, it follows that Aboriginal countries, in this case expressed as subregions, must be prioritized as a set criterion in negotiation of protected areas and indeed other regional initiatives on Cape York. The people that belong to a certain country demand the right to determine what is done with it and to it.

A SUGGESTION FOR A BLENDED APPROACH

It is clear that protected-area planning needs to be informed by different levels and types of knowledge. Individuals and agencies such as those present at the Congress are well qualified to provide the information on western models pertaining to protected areas, as specialists in this area. Indigenous people are well qualified to provide their knowledge both of biological systems and the social context into which they must fit; they too are experts on their country in both holistic and specialized ways. There is a requirement for intermediaries that can bring the two systems of knowledge together. We are attempting to do this in a small way with this paper. In our view the 'linkage people' are a crucial area that is neglected very badly in a funding sense – and let us say right now that linkage cannot be achieved from an office. On-site engagement is necessary, and operational expenses are significant.

Anthropological, ethnographic, historical, and social and art literature have an important place in land and sea management in Australia (Moore 1979; Sharp 1992; Sutton 1995; Langton 1998; Langton *et al.* 1999) and Indigenist research is becoming a new force on the intellectual front. The context of specialist research must be understood and directed to support social outcomes if we are to get serious about management and protected areas on Cape York.

As far as the seas and coral reefs in particular are concerned, we provide an illustration of a hypothetical situation for consideration (Fig. 1).

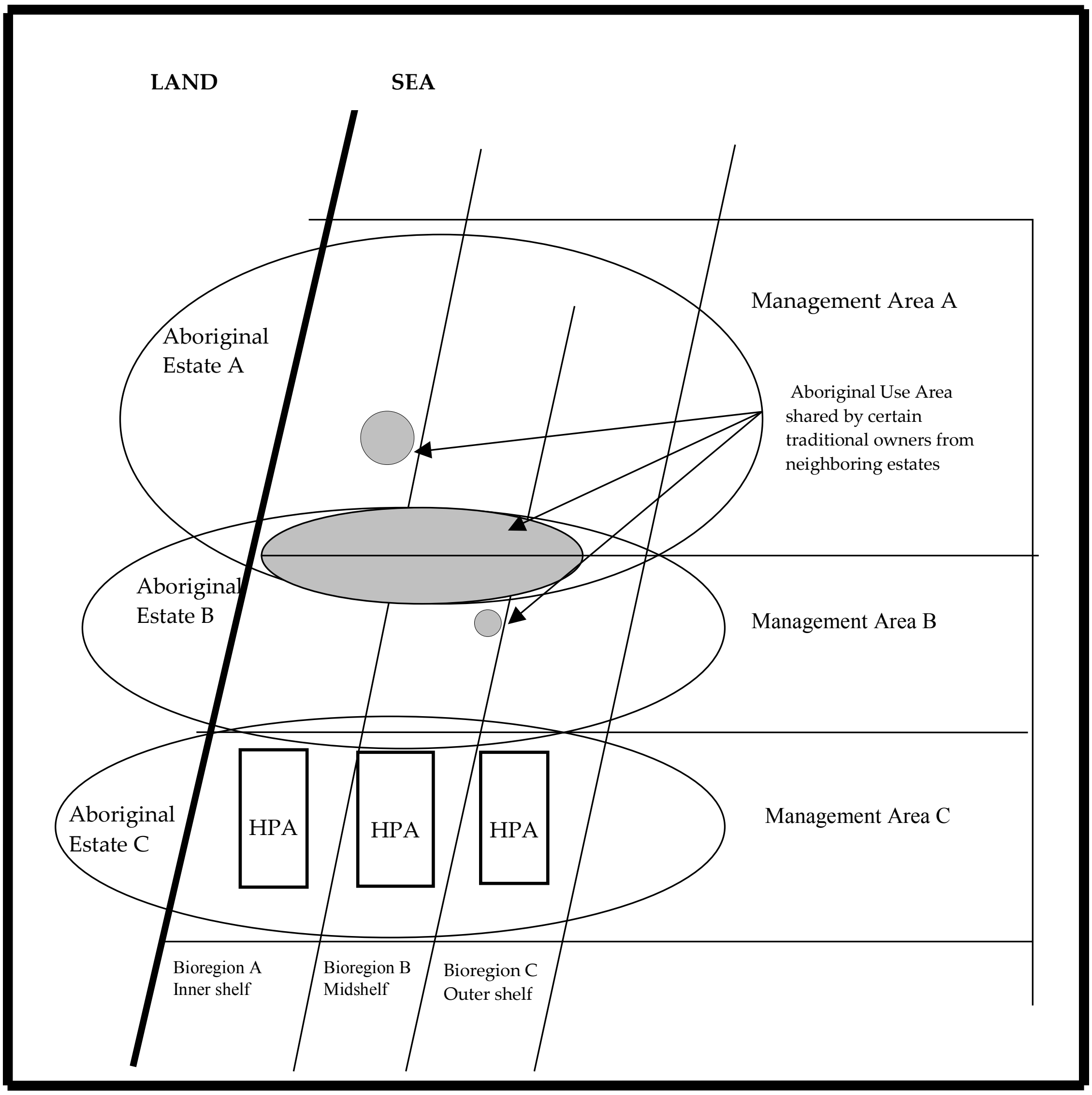

Fig. 1. Interrelationships on the east coast of Cape York, Australia. *Subregions are made up of one or several clan estates (oval shapes).* HPA = highly protected area or no-take reserves. *Shaded areas: areas traditionally shared by various people.*

Imagine the east coast of Cape York. In the diagram above HPA stands for "highly protected area" or "no take" reserves. Subregions are made up of one or a number of clan estates. In some cases the Aboriginal estates are clear in reality, in others they are not. Countries might cover land and sea.

Affiliations with places can be quite particular to individuals. Some areas are traditionally shared by various people (Fig. 1). The western management system has difficulties delineating the complex understandings that Aboriginal people have developed over millennia and which continue to change. In several cases in our experience, indigenous peoples have issues in dealing with broad-scale regional management and fitting themselves between existing regulations within foreign jurisdictional boundaries (Shirley Johnson, Wulgurukaba pers.comm.). So perhaps it is appropriate to investigate the practicalities and possibilities of using an artificial management-area system that does not deal with ownership directly but targets agreement on a management regime for a particular area. The boundary can roughly approximate the real thing but need not be the real thing. The issue here is about the right people agreeing to the management of a defined area rather than working out exactly where the traditional boundaries for each are. The latter concept, using a management boundary as a substitute for what could be very complex arrangements in reality. has been hinted at from a number of directions[1]. Indigenous people sometimes feel it would be easier to agree to some artificial representation of their interests where it might assist in getting on with the job. The reality is complex to explain to people unfamiliar with such concepts and it does not suit mainstream legal provisions or understanding. Consequently, the Aboriginal concept of country is deemed 'impractical' by many agency managers. Indigenous peoples themselves have been saying all the while that their lives are centred on their country, but the message has not been heard. Account for country, account for beliefs, account for people is the message. Management plans must be drawn up with the people concerned. In the case of management arrangements for areas that overlap, neighbours must be invited in.

In the area marked Aboriginal estate C (Fig. 1), we show what can happen when protected areas coincide with a clan estate. In this case, we assume that it is desirable to reserve an area in the inner-shelf bioregion, the mid-shelf bioregion and the offshore bioregion. If they all fall into the same Aboriginal estate or country, the ability for that Aboriginal group to use resources to make a living is seriously compromised where rules for the reserved areas constrain Aboriginal traditional and commercial use. This can be further complicated by a 'complementary' on-shore protected area, often a government aspiration.

This 'management area' arrangement offers a potential approach for delineating management regimes by cultural affiliation and getting involvement from relevant traditional owners, but it must be clearly explained to traditional owners, it must be done by particular people, and it must be well resourced to achieve lasting results. This is not a complicated principle for traditional owners to understand. The politics that attend it are what is complicated.

In simple terms we feel that the Aboriginal cultural management unit is one of the immutable realities for successful management and sustainable use. It must become a fundamental plank in management particularly if compliance with a management plan including protected areas is a required outcome. The attitude taken by all must be 'if we cannot accommodate indigenous people (at least to some reasonable and agreed extent) this plan is going to fail'. This means that managers and other agents need to identify a number of options within targeted protected areas or at least be aware of the complexities, that can be worked on with the relevant traditional owners to provide mutually agreeable outcomes. We believe that this principle is internationally applicable.

CONCLUSION

Indigenous peoples are traditionally tied to their country, whether it is land or sea or both. They are empowered under traditional law and Australian law to use the areas for getting resources and conducting their culture, and they have obligations and responsibilities for their countries. Indigenous peoples are obliged to protect and preserve their inheritance. They are to some extent bound by it. Consequently, Indigenous peoples' 'belonging' or ties to their country becomes a non-negotiable reality of a proper management plan. These attachments in ordinary traditional circumstances cannot be shifted for the sake of convenience.

It is clear that some management systems and protected-area placements can play havoc with

[1] Aboriginal peoples are aware that western thinking cannot understand their ways of doing things, and they have consistently made this known, attempting to fit their priorities into a foreign but dominant perception of their environment. Leanne Sommer, Dermot Smyth and David Epworth have also shared ideas on management boundaries and must be acknowledged.

this understanding, denying access and usage rights. Inappropriate management is resented and ignored not just by indigenous people but also by other parties. It is clear that indigenous peoples are in a good position to be on-ground managers of their places: they are there, they generally like being there and are frequently obliged to be there. These desires and synergies should be encouraged by program development.

The question is, 'Can indigenous geographic and traditional reality be conscripted to deliver biodiversity outcomes sought by most of the western scientific paradigm?'. We believe it can. A fixation on biodiversity as the primary criterion for management is an obstacle to free thought on many issues, particularly the idea that cultural diversity, cultural management systems and indeed languages might be the crowning glory of Darwinian process and worthy of consideration as higher-order criteria for protected-areas planning. Instead of having protected areas designed and managed with biodiversity as the principal determining factor, perhaps the clan or group estates of indigenous peoples expressed in practical sub-regions can provide an acceptable framework to do the same thing. The social implications are far-reaching, as are the compliance implications.

Western science informs policy and planning. Traditional knowledge and custom should do the same. Collection and interpretation of Aboriginal knowledge are specialist arts requiring experience and skills. Efforts must be made to gather what is left before it is lost and no longer available to management or research. This knowledge must be stored and taught in communities to and by indigenous peoples. This information is not merely of curiosity value but represents the oil for the machinery required for indigenous people to participate in – and direct – management. The aim of the exercise is to improve the lives of indigenous people through negotiation and manifestation of rights in management. Gathering information must not be seen as merely getting the information before it is lost and then not worrying about the people. The people, not the researchers, own the knowledge. Indigenous people must benefit from it in practical ways in the present as well as in the future.

The ideas suggested require significant investment and serious increases in capacity (within communities, indigenous organizations and research institutions). We have huge amounts of biological information but it must be expressed in terms of social outcomes. Expansion of protected-areas planning to become multidisciplinary projects into the arts and social sciences offers great potential in gaining a fuller understanding of indigenous cultural space in the sea and would no doubt be pertinent to non-indigenous sectors also (Jackson 1995). Finally, the process must be community owned, and research programs must be designed accordingly including at the budget level.

Scientists must consider the implications of their advice on regional and local cultural structure, not only the environment. For the taxonomists, that 'new species' you find will have been found on someone's country, and someone will therefore already be responsible for it. Ignoring this fact can cause problems for people on the ground.

The primary distinction between the western scientific view and indigenous views of management appears to be that 'biodiversity'-based management is an animal-and-plant-systems concept, whereas Aboriginal estates or countries are holistic systems with a strong human dimension. Even the most brilliant and comprehensive biodiversity-based management plan quickly fails if people do not understand it and therefore don't comply with it. Not just indigenous people but all people.

All interested parties might first consider whether cultural management areas can provide a mechanism for representing two worldviews. Secondly they might consider how these might be negotiated to account for the social, cultural, economic and political circumstances of indigenous peoples and provide a positive direction all around. Thirdly, they might consider how social sciences can be incorporated into the 'expertise' required for good management. This requires a meeting of hearts and minds, reassessment, strategic direction, funding of capacity building, time and (importantly) the will.

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INCORPORATING TERRESTRIAL AND UNDERWATER CULTURAL RESOURCES IN AQUATIC PROTECTED AREAS MANAGEMENT TO AID COMMUNITY DEVELOPMENT, ENHANCE TOURISM AND FACILITATE RESOURCE STEWARDSHIP

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Abstract

Aquatic protected areas are most often associated with protecting salt and freshwater areas for maintenance of aquatic biodiversity, as nursery areas, and for protection of aquatic ecosystems and habitats. Yet controversy often swirls around such areas, primarily due to single-purpose management of these areas in a world of multiple-use demands on and multiple values associated with those resources. These multiple values and use demands are imposed by humans, who historically have used these resources for food; as materials sources for clothing, fuel, construction; as inspiration for artistic expression, culture, and spiritual understanding of the world; and as a recreational resource. The water itself has been used as routes for transportation and trade. Associated cultural structures, artefacts (terrestrial and underwater), stories and expressions (foodways, traditional arts & culture, language) help link people with water and coastal environments. Consequently, aquatic resources management is impossible without also managing human responses to and interactions with them, and without also considering the cultural objects and structures associated with them. Occasionally, aquatic natural and cultural resources can be symbiotic, such as when submerged cultural resources (e.g. shipwrecks and docks) serve as habitat for aquatic species. This paper proposes that, to develop effective management strategies for protecting aquatic resources and expanding public support for them, the range of resources, uses and values should be integrated. Arguing for interdisciplinary approaches to aquatic protected area management, this paper uses selected case studies and research to illustrate the role of aquatic protected areas, including associated coastal resources, in community development, tourism enhancement, and use of interpretation and education to encourage resource stewardship. Recommendations include recognition of multiple resource values and uses, use of the "maritime cultural landscape" concept as a development and management tool, involvement of multiple stakeholders in both management decisions and management actions, increased use of collaboration and partnerships, and training of students and professionals via an integrated interdisciplinary approach.

Keywords: marine parks, marine protected areas, system-based management, multiple resource values, maritime cultural landscape

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INTRODUCTION

Aquatic and marine protected areas are most often associated with protection of water bodies to maintain aquatic biodiversity, as nursery areas, and for protection of aquatic ecosystems and habitats. Consequently, much of our scientific study focuses on the biological, physical and chemical conditions and changes associated with the water and things in it – living organisms, chemicals, sediments, diatoms, etc. However, most aquatic protected areas, by nature of their biophysical relationship with adjacent land structures, are impacted by what happens on land as well as in and on the water, thus suggesting that management incorporate terrestrial activities. Also, having the sole focus of marine protected areas on protection of aquatic ecosystems, whether real or perceived, often is the source of controversy by stakeholders who value and use the water for other purposes. Understanding human attitudes, values and perceptions as well as behaviors is as important to managing the resources as is understanding the relationships between geophysical, biological and chemical conditions of aquatic systems. Recognizing, understanding and minimizing "impact creep" and non-point-source pollutants are as important as managing the more visible catastrophic impacts. Thus, management of aquatic or marine protected areas is more than managing complex aquatic ecosystems. Education and involvement of the general public (as individuals and as members of various stakeholder groups), inclusion of aquatic-system impacts in land-use decisions, and integration of social science with geophysical and biological sciences are critical to the success of aquatic protected area management.

This paper presents five broad recommendations to work toward a more holistic, systems-based approach to managing aquatic and marine protected areas. Fundamental to success of the final four recommendations is the recognition of multiple resource values and uses of aquatic and marine resources.

MULTIPLE RESOURCE VALUES AND USES

Historically, humans have ascribed multiple values to aquatic resources and have used them for varied purposes: for food; as materials sources for clothing (e.g. whale baleen for corset stays, beaver pelts for hats), fuel (e.g. whale oil), building materials (e.g. limestone; sand for concrete); as inspiration for artistic expression, culture, and spiritual understanding of the world; and as a recreational resource. The water itself has provided easy-to-access routes for transportation and trade. Proximity to waterways has always been important in selecting areas to farm (water to irrigate and to transport goods), to develop industry (water to power mills and produce electricity), for protection (shore-based prominences for siting military forts and castles for strategic advantage and visual access to attackers), and for developing communities (access to water, water-based food sources, and easy transportation). In turn, the coastal environments and the water-based work – whether fishing or sailing, moving lumber rafts or rescuing shipwrecked sailors, servicing sailors on shore leave or working on oil rig platforms – influenced the development of foodways, traditional culture and arts, and language.

Historically, sea shanties helped sailors perform a variety of tasks in unison, from hauling lines on sails to raising anchors. The arts – such as carving on baleen and ivory, turning brass goblets in the engine workshop, and sketching cartoons and women on calendars – have helped sailors express themselves creatively while keeping boredom at bay. Tall tales and souvenirs from exotic ports have helped bring the lives and cultures of people from far-flung corners of the world to other places via ship. Marlinespike (knot-tying, sometimes called fancy work) not only served specific nautical functions, but was used to decorate almost anything within a sailor's physical world. Our modern English language – through still-used nautical terms and phrases – has been influenced significantly by nautical life and language, even if most of us never know the sources. (For example, we warn children to "toe the line" if we want them to behave. The term derives from two uses aboard ship: when warship crews were ordered to line up in formation on deck, with their toes aligned along the lines between planks that were packed with oakum; when young sailors committed an infraction, they might be asked to stand for hours along one of the lines. Other expressions with nautical roots are "chewing the fat," "clean bill of health," and "foot loose and fancy free.") (International Marine Educators, Inc., 2002; Naval Historical Center, 2002) Maritime cultural structures and artefacts, both coastal and underwater, still function or serve as reminders of lifestyles dependent on water. Lighthouses and lifesaving stations have a special allure to tourists. The once-functional Fresnel lenses now are considered artistically beautiful as well as technologically sophisticated. Submerged remnants of piers, docks, and cribs mark places of former industry and transportation. Shipwrecks and associated artefacts serve as time capsules for social, technological, political and economic history.

Today, while many fewer people's lives are tied directly to the sea or Great Lakes than even 100 years ago, we are all still heavily influenced by the water. From El Niño and La Niña to typhoons and hurricanes, from tidal changes to lake-effect snowfall, global weather patterns are strongly influenced by the sea and lakes. Despite advances in technology and navigation, the most cost-efficient way to transport bulk cargo still is via large freighters. Ships, ranging in size from huge factory ships to small shrimp boats, still are used to harvest food from the lakes and sea. Although whale oil has given way to petroleum as an energy source, aquatic areas are still important. Oil rigs have long broken the surface of salt water, from the North Sea to the Gulf of Mexico; today, there is increasing pressure to drill under the North American Great Lakes. Although often less directly connected to water-based industry, residents of coastal communities build monuments to recognize the lives and contributions of family members, community heroes, and laborers who were tied closely to the sea.

During the Age of Sail, waterfronts and coastal areas were considered the "front yard," the prime areas of real estate and often the location of homes of sea captains and wealthy merchants. During the Industrial Revolution, when industry became a dominant coastal land use and the water became the primary dumping ground for industrial and urban waste, the many coastal areas became the "back yard", the necessary but unattractive "dump" and an industrial "eyesore." In recent decades, massive clean-up efforts have been triggered by visibly obvious and highly publicized environmental disasters such as the burning of the Cuyahoga River, identification of 418 toxic chemicals in the air, water and soil of the Love Canal region of Niagara Falls (Bertuca 2002), the devastating impacts on the Great Lakes fishery by invasive lamprey eels, and the scientific work presented in Rachel Carson's *Silent Spring.* Thanks to these clean-up efforts, coastal areas have once again become attractive – and expensive – real estate. Pressures on coastal areas – for residential, commercial and recreational development – continue to increase. In fact, nearly half of all the building construction in the United States of America (USA) during the 1970s and 1980s occurred in coastal areas. Currently in the USA, 54% of the entire population lives in the fringe defined as "coastal counties" (NOAA 2002). Some projections indicate that, by the year 2010, the United States' coastal population will swell to more than 127 million, reflecting an increase of more than 60 percent since 1960. (NASA 2002) Much of the coastal development is on coastal dunes and barrier reefs, environments that are subject to erosion, longshore movement of sand, and hurricanes. In efforts to protect this attractive and expensive property, people haul sand or pump it ashore from deeper water, then build seawalls, jetties and other structures that, in the long run, often increase erosion and transport of sand.

Lakes, seas and associated coastal areas also remain magnets for recreation and tourism. Beaches, ranging from isolated, pristine sandy beaches to heavily developed boardwalk areas such as Coney Island and Atlantic City, draw millions of visitors annually. Grand hotels and small fishing cabins, all-inclusive resort complexes and intimate ecotour cottages line coastal areas. Cruise ships, having grown popular in salt-water environments, are returning to the Great Lakes as a component of lake-based tourism. High-speed ferries carry tourists to Mackinac Island, where motorized vehicles are not allowed. As the Coast Guard gives up responsibility for structural complexes of its lighthouses, public and nonprofit organizations are assuming responsibility for maintenance of many of these and opening them to residents and tourists for tours and as museums. Other recreational uses of the water include motorized and non-motorized boating, sport fishing, SCUBA diving and snorkeling. Boat tours, scenic and glass-bottomed, provide tourists a new perspective on coastal, island and underwater environments. Dinner cruises and on-board gambling offer value-added experiences for visitors engaging in common activities by providing unique venues. New types of recreational equipment are constantly being developed to further take advantage of water-based and coastal areas, including parasails, windsurfers, personal water craft, surface-supplied dive helmets, mini-submarines, and submerged tourist lodging.

Economic impacts almost always are one category of values associated with resource use decisions, with attempts made to express other values also in economic terms, such as willingness to pay, and costs of lost opportunities. Values more often conceptually associated with preservation of terrestrial and aquatic wilderness areas include existence value, biodiversity value, and future value. (Payne et al. 1992). Those arguing for use values typically associate direct economic impacts with specific resource uses. In aquatic environments, examples include boating, diving, shipping, and commercial and sport fishing. Estimated and projected economic impacts based on primary data collection are relatively limited for non-industrial and commercial uses, but some examples include the following for recreational uses of aquatic resources:

- User studies show that recreational SCUBA diving was responsible for $US5.8 million in direct spending in the Alger Underwater Preserve (Michigan, Lake Superior) in 1984 and 1985, and an estimated $US3.7 million in the Leamington Marine Heritage Area (Ontario, Lake Erie) (Vrana 1997);
- In 1994, owners of Michigan's 555,000 active boats contributed 13.4 million boat days and over $US200 million in boating-related expenditures at marinas and vicinity (Talhelm et al. 1998).
- In 1991, Michigan's 1.76 million anglers contributed over $US1.073 billion in direct equipment and trip expenditures (Fedler and Nickum 1991).
- Although they acknowledged the challenges of estimating potential impacts of a marine sanctuary prior to its designation, based on secondary data from similar areas and probable uses, researchers projected a possible $US2.5 million in direct visitor spending for the Thunder Bay National Marine Sanctuary and Underwater Preserve (Mahoney et al. 1996).
- Although development of a maritime heritage landscape in a small Michigan community, anchored by a large historic vessel (a rail car ferry), focused on multiple resource values, a critical component of the business plan was the potential economic impact (Vrana 1999).
- Although not specifically an economic impact study, a survey of Great Lakes maritime attractions indicated that these tourism attractions charged entry fees ranging from

free to $US13 per person, with over 75% charging $US5/person or less. Operating budgets ranged from "less than $US5000" to "more than $US1 million," with nearly 60% having gross annual operating budgets of less than $US150,000 (Tolson 2000).

- An study on economic impacts of cultural tourism (museum-based sampling) is currently under way in Michigan. Maritime-themed sites are included in the study.

Associated with and underlying the wide range of aquatic resource uses are the values and beliefs ascribed to the resources and their uses. Frequently, conflicts arise over the uses, and which use has priority. Sport anglers blame factory ships for damaging fish nursery grounds and over-harvesting fish, including spawning stock, and for by-catch kill. Often, commercial fishermen blame sport anglers for wielding political power to limit their commercial catch. Both fear being locked out of quality fishing areas by competing uses (shipping, sport diving), negative impacts of invasive species transported by freighter ballast, and exclusion from prime fishing areas by designation of marine preserves or sanctuaries. Quiet water sports enthusiasts (sailors, canoeists, kayakers) dislike the intrusion of power boaters and jet skiers. "Environmentalists" fear oil spills from tankers and below-lake drilling. Coastal home-owners blame neighbors' jetties for triggering beach erosion in front of their homes. Some stakeholder groups welcome designation of aquatic protected areas – as attractions for tourism, for protecting the value and aesthetics of personal property, as components of economic development – while others fear that their personal rights and preferred uses of the water, both economic and recreational, will be unjustly prohibited. Therein lies the larger, more complex set of issues facing managers of aquatic and marine protected areas.

The remainder of this paper argues for interdisciplinary approaches to management of marine and aquatic protected areas. Selected case studies and research are used to illustrate the role of aquatic protected areas, including associated coastal resources, in community development, tourism enhancement, and use of interpretation and education to encourage resource stewardship. Based on the existence of multiple resource values and uses, recommendations include use of the "maritime cultural landscape" concept as a development and management tool, involvement of multiple stakeholders in management decisions and actions, increased use of partnerships in managing coastal and marine protected areas, and training of students and professionals in an integrated interdisciplinary approach.

MARITIME CULTURAL LANDSCAPES AS A DEVELOPMENT AND MANAGEMENT FRAMEWORK

In the biological and other natural sciences, "ecosystem management" has been gaining increasing attention as a way to more holistically manage natural resource systems rather than managing individual species. One definition describes an ecosystem as "a community of different species interacting with one another and with their nonliving environment of matter and energy" (Miller 1998, p. 97). Definitions of ecosystem management usually expand the management process beyond just the physical and biological components of the system, also recognizing human values and actions as important components. Additionally, it incorporates desired management goals or outcomes. Thus, ecosystem management can be defined as "the integration of ecological, economic, and social principles to manage biological and physical systems in a manner that safeguards the ecological sustainability, natural diversity, and productivity of the landscape" (Wood 1994, p. 6).

Although humans and their needs and impacts are incorporated in the concept of "ecosystem management", the label may be unfamiliar or uncomfortable to scientists, historians and managers grounded in the social sciences (e.g. history, archaeology, economics, sociology, anthropology), folkways and the cultural arts. Often, the term "ecosystem" is perceived as ignoring human values, such as economic health, community development, and cultural and aesthetic values. Originating in the domain of historic preservation, a similarly holistic, yet more human-centered, concept evolved as a way to preserve and manage historic resources. This concept, termed cultural landscape, was initially single-discipline-focused with the primary goal of preserving historic structures and communities (this is comparable to single-species management in the natural sciences). The concept has expanded to include other disciplinary approaches (e.g. archaeology, anthropology) and generally describes geographical settings impacted by and that now reveal relationships, past and present, that have developed among humans and the land or sea, through their interactions over time. A formal definition, provided by the United States National Park Service, describes cultural landscapes as "geographic area[s], including both cultural and natural resources, and the wildlife or domestic animals therein, associated with a historic event, activity, or person or exhibiting other cultural or aesthetic values" (Birnbaum and Peters 1996; NPS 1992, p. 107)

Of four general types of cultural landscape described by the National Park Service, one – historic vernacular landscapes – best meets the needs of using a systems approach to managing historic, cultural and natural resources. By the NPS definition, historic vernacular landscapes are those "whose use, construction, or physical layout reflects endemic traditions, customs, beliefs or values; in which the expression of cultural values, social behavior, and individual actions over time is manifested in the physical features and materials and their interrelationships, including patterns of spatial organization, land use, circulation, vegetation, structures, and objects; in which the physical, biological, and cultural features reflect the customs and everyday lives of people" (NPS 1992, p. 4). Vrana and Vander Stoep (2003) argue that the cultural landscape (or heritage landscape, a term used more commonly outside the USA and intended to include natural resources) framework can be applied to maritime contexts. Such applications include both the resources and human activities occurring on and in the water and those impacting or occurring on adjacent terrestrial (coastal) areas. Thus, all types of resource uses and values (discussed in the previous section) are considered when developing comprehensive management plans for such areas.

The complex of water-based human activities *(coastal trading and commerce, shipping, commercial and sport fishing, boat building, exploration and discovery, immigration via water routes, transportation, naval operations, navigation and marine safety, lumber transport, and maritime/coastal recreation and tourism)* and associated resources on, under and adjacent to the water *(fish and other aquatic species, dunes, beaches, bluffs, coastal vegetation, remains and artefacts from prehistoric native peoples, lighthouses, lifesaving stations, large vessels, small water craft, shipyards, warehouses, docks, piers, industrial structures, locks and canals, oil platforms, naval facilities, water and power generation stations, fish houses, boat houses, canneries, coastal farms benefiting from temperate coastal climates, coastal communities – historic and contemporary – and other historic sites)* all are part of the maritime heritage landscape.

Using the concept of maritime cultural landscape as a research, planning and management framework allows not only consideration of a wide range of resources and activities, but also of cultural values and aesthetics. It facilitates the study and understanding of historic and current interrelationships between human actions /resource uses and the health and sustainability of those resources. It facilitates consideration of economic and quality-of-life sustainability as well as ecosystem sustainability and historic resource preservation. The concept has not been widely accepted or applied, but examples in the Great Lakes Basin do exist. Not surprisingly, some of the first are in regions where the National Park Service has a role: 1) Sleeping Bear Dunes National Lakeshore and the adjacent Manitou Passage Underwater Preserve (Vrana 1995), located in Lake Michigan in the northwest corner of Michigan's lower peninsula (historically along a major shipping route; a place of fuel-wood replenishment and harbor of refuge; a major agricultural area dependent on the climate moderated by the lake); 2) Isle Royale National Park, an archipelago in the northwest portion of Lake Superior (having a long history of water-based human activity, including copper mining, both Native American and European; commercial and recreational fishing, both Native American and European; historic and current tourism; recreational activities).

A small coastal community in Wisconsin is currently developing a community-wide tourism product and experience, incorporating historic maritime themes; historic structures and landscapes within the community and along the waterfront; current recreational waterfront uses; and development of a maritime museum and harbor-walk interpretive trail. The maritime themes are being used to integrate the community's interpretive stories, to develop a tourism brand and logo, to preserve the historic maritime architecture, and create multi-component maritime tourism experiences for visitors. Concurrently, economic development, community enhancement, and education of residents about their history are incorporated in the planning. (Vrana et al. 2000).

Another example is along the shores of Lake Huron, where the Thunder Bay National Marine Sanctuary and Underwater Preserve was officially designated in 2000. The Sanctuary and Preserve is "the first [NOAA (National Oceanic and Atmospheric Administration)] national marine sanctuary to focus solely on a large collection of historic shipwrecks and other underwater cultural resources, and the only sanctuary located entirely within state waters." The co-management team (State of Michigan and NOAA), in cooperation with the adjacent community of Alpena, Michigan and other stakeholders, is considering use of the maritime cultural landscape model, as recommended in the pre-designation business plan (Vrana and Schornack 1999).

Theoretically, such comprehensive planning approaches, guided by a maritime cultural landscape framework, should facilitate development of win-win management decisions, but challenges still exist. Use of the landscape framework does not guarantee lack of conflict about resource uses and values, does not immediately develop trust where distrust and antagonism may have developed in a community over time, and does not mean every stakeholder will actively or willingly participate. However, it does at least provide input and participation by stakeholders representing industry, commercial development, resource protection, historic preservation, residential interests, and scientific evidence. Results of relevant biogeochemical, economic, historical and social research should be considered in developing plans based on a cultural or heritage landscape framework. As is consistent with a holistic systems planning approach, the interrelationships and impacts of alternative plans on the resources and social variables should be predicted and considered. Finally, if a maritime heritage landscape framework is to work at all, stakeholders must be involved in the process, and partnerships of various kinds can more effectively facilitate implementation of components of a resulting heritage landscape plan.

STAKEHOLDERS' INVOLVEMENT IN MANAGEMENT DECISIONS AND ACTIVITIES

In addition to the wide range of uses and values that different stakeholder groups and individuals ascribe to aquatic resources, stakeholder groups are often quick to place blame elsewhere for negative impacts on the resources. Also, most are outraged by catastrophic events affecting resources (often blamed on others) and much less aware of "impact creep," a process that includes the full range of less immediately obvious and often slower changes resulting from cumulative effects of non-point-source pollutants and relatively small changes in land use. The general public expresses shock when media reports tell of manatees or dugongs being sliced by boat propellers or when tankers spill large quantities of crude oil. Sport anglers decry the reduction in numbers and size of sport fish, often accusing commercial fishing interests of greedy harvesting. Residents and tourists are angry when sewage overflow from municipal water treatment facilities necessitate beach closure. However, the general public is less likely to be aware of or concerned about how "common" actions on land – personal behaviors as well as those of agriculture, industry, and urban development – affect the water resources, including designated aquatic protected areas. These include things such as dumping of leftover paint and used motor oil down gutter drains, construction of jetties or groynes along property boundaries of private waterfront homes, runoff of residual lawn pesticides and herbicides, and the gradual transition from vegetative to hard surface cover through paving and building construction. Yet, people engaging in these actions often are not aware of their impacts, they believe their impacts are miniscule compared to those of industry, they are not overly concerned about impacts if they are not immediately obvious or relevant to themselves, or they believe that someone else should take care of any problems. Their involvement as stakeholders in a comprehensive landscape planning process can both provide an opportunity for them to express their preferences, values and resource-use priorities *and* provide opportunities for them to learn about impacts of water- and land-based actions as well as the interrelationships between various resource uses and impacts.

Stakeholder input and involvement can occur in a wide range of structures, including stakeholder surveys, focus groups and other social research; use of advisory committees or special task forces; public open houses and public meetings; management "alternatives workbooks" (Sleeping Bear Dunes National Lakeshore, NPS 2001); community charettes and planning teams. Each of these techniques has associated advantages and disadvantages, and varying characteristics; some are more appropriate early in a planning process, others as ways to maintain dialogue with stakeholders throughout a planning process and to use as evaluation and monitoring tools. Selection of the proper process depends on the objectives (Arnstein 1969; Glenn 1978; Glass 1979; Fazio and Gilbert 1986). (Benefits and challenges of the processes of public stakeholder input, including as applied to natural resource management, are covered extensively in the literature, so are not fully developed here.)

Because marine and coastal resources (including aquatic protected areas) are so attractive as tourism resources and community assets, it makes sense to include decisions regarding use, and development and management of these resources, within a broader framework of community and economic development that includes tourism. Additionally, resources and amenities developed for tourism should also specifically benefit residents. Process models have evolved in different disciplines, including community development, rural sociology, museum and cultural arts, and tourism. Most contain similar components, principles and process steps that incorporate stakeholder involvement, use of collaboration through partnerships and other structures, principles of authenticity and historic preservation, a focus on natural resource protection, and inclusion of economic factors. Underlying the processes are goals of environmental, economic, and social sustainability.

- A nation-wide partnership of museums, other cultural institutions and business interests, called "Partners in Tourism: Culture and Commerce," identifies five principles of heritage tourism: "focus on authenticity and high quality; preserve and protect historic, cultural and natural resources; make sites come alive; find the fit between community values and tourism; collaborate" (Partners in Tourism 2002, p.1)

- Michigan State University (MSU) Extension has conducted a series of community-based tourism development workshops, focusing on principles of authenticity and collaboration (Vander Stoep and Schaffer 1996-2000). Using the Kretzmann and McKnight (1997) asset-based community development model as a planning structure, programs contained case studies, tools and workshops to illustrate their five major process components: mapping community assets (inventory), building relationships (with stakeholders, developing partnerships), mobilizing for economic development and information sharing, community visioning, and leveraging outside resources to support locally driven development. These principles and processes can be applied in any environment, but they are particularly useful in marine and coastal environments because such areas are prime tourism destinations, and because the terrestrial-and-aquatic systems are complex.
- The MSU Tourism Resource Center uses multi-disciplinary teams of experts to work with community stakeholders and task forces to assess tourism readiness and opportunities.
- Similarly, the Glywood Center, through its Countryside Exchange Program, convenes multi-disciplinary professional teams with representatives from North America and Europe, to work with community stakeholders to assess and develop recommendations for community and economic development. Coastal communities, which usually incorporate tourism and water uses in their plans, have used such teams. (Glynwood Center 1999)
- NOAA has recently produced a mini-CD to encourage and facilitate community involvement in marine and coastal management. The CD, "Engaging Communities: Participatory Strategies for Coastal Managers," describes and provides case studies for three methods to enhance public participation in aquatic and coastal resource management decisions (NOAA 2002).

In addition to planning process involvement, stakeholders can be involved also in management activities. This has added benefits of building community pride and identity, educating local residents and visitors about the resources, developing attitudes and behaviors regarding stewardship of natural and cultural resources, and facilitating long-term resource monitoring and management. Examples associated with marine and coastal environments include:

- using citizens to monitor water quality;

  Throughout the USA, numerous programs involve citizens in collecting water quality data. Data from numerous watershed monitoring stations are entered into integrated databases for analysis.
- citizen participation in river, lake, and beach cleanups and dune grass planting;
- management of cultural resources by nonprofit organizations, either as sole or co-managers;

  In Michigan, the Great Lakes Lighthouse Keepers Association has restored and now manages an island lighthouse in the Mackinac Straits area, where it conducts educational programs. Additionally, as more lighthouse complexes are no longer needed by the Coast Guard, many are being transferred to nonprofit groups, which develop a range of uses for the sites, including adaptive re-use as museums, offices, tourist lodging, and restaurants; for public tours; and as educational centers for youth-at-risk.
- avocational involvement in shipwreck historical research, documentation and monitoring

  Building on a 1989 course, MSU has offered, since 1999 and in partnership with Canadian and USA managers of submerged resources and state historians, two courses in avocational underwater archaeology and maritime historical research, resulting in Level I certification of participants by the Nautical Archaeology Society, an international organization supporting efforts to research and preserve submerged cultural resources (Vander Stoep 2001).

INCREASED USE OF COLLABORATION AND PARTNERSHIPS

As stated in the "stakeholder involvement" section, collaboration and creation of long- or short-term partnerships facilitate involvement of stakeholder organizations and agencies, both those having similar goals and programs and those that may complement each other. Potential benefits include sharing of information and better understanding of other organizations' management goals and priorities; pooling of funds to finance joint projects, products and services that might not otherwise be possible; coordination of information, tourism experience packages, and reservation systems to facilitate tourist use of aquatic and coastal resources; coordination and reinforcement of stewardship messages; and sharing of varied skills, expertise and research data for mutual benefit. Coordination of specific aquatic resource management actions across governmental and political jurisdictions is critical, especially for watershed systems that do not "respect" geographic and political boundaries. For managers of aquatic protected areas, involvement in broader community and economic development efforts (e.g. being a member of the Convention and Visitors Bureau, a city development authority or community economic development team) can assure that aquatic resource protection issues are considered in more comprehensive planning and development efforts. Such participation can also help build trust between the managers of aquatic protected areas and other community leaders and residents. Although challenges do exist in maintaining successful partnerships and co-management efforts, the potential benefits, in the long term, are worth the effort. Some examples include:

- The International Joint Commission was created in 1909 to resolve disputes over use of water resources that cross international boundaries, with the Great Lakes being a major system within their jurisdiction; in the Great Lakes, efforts have focused on scientific studies and advice to governments about problems affecting the Great Lakes Basin (Government of Canada and the United States Environmental Protection Agency 1995).
- In the Alger Underwater Preserve region of Michigan's Upper Peninsula, as in increasingly more locations, multiple management agencies co-operatively operate a visitor information center that serves as a visitor gateway to a range of land- and lake-based experiences.
- Increasingly, aquatic resource managers are developing watershed-based approaches to resource management rather than staying confined to political boundaries.
- In some areas, particularly in Canada, First Nation professionals are co-managing resources with federal agencies; this is particularly true for fisheries management.
- In the recently (2000) designated Thunder Bay National Marine Sanctuary and Underwater Preserve, the State of Michigan and a federal agency (NOAA, Marine Sanctuary Program) created a formal agreement and procedures for co-managing the Sanctuary/Preserve. Still in its infancy, the co-management team also regularly involves a Sanctuary Advisory Committee, a group of non-agency local residents who represent the views of various local stakeholder groups.

INTEGRATED INTERDISCIPLINARY TRAINING

Successful implementation of stakeholder input and involvement, as well as collaborative management of resources, requires time, patience, trust, listening skills, open minds . . . and understanding of both the current knowledge and philosophical foundations of specialists working in a wide range of disciplinary areas. In the management of aquatic protected areas, this includes geologists, physicists, chemists, aquatic toxicologists, limnologists, fisheries biologists, historians, archaeologists, anthropologists, sociologists, climatologists, and many others. Traditionally, our educational system has been structured so that students must become increasingly specialized as they advance through degrees. Also, most professionals tend to attend conferences and to read and write for journals in their own fields. However, natural resource systems are inherently complex, and global economies and their impacts on natural resources are increasingly interrelated and complex. Wise decisions rarely can be made unilaterally (in terms of a single discipline). Increasingly, use of interdisciplinary teams is being encouraged for research as well as resource management. Often, however, professionals in varied fields speak "different languages," based on their professions. Interdisciplinary collaboration and joint decision making would be considerably easier if scientists and managers had some basic understanding of the content, research, language and priority issues of those in other fields. Therefore, this paper argues for development of more interdisciplinary educational, training and professional development opportunities (including conferences). This does not mean that everyone should be a generalist, or that no one be a specialist. It does mean exposing students and professionals to other disciplinary content.

As an example, for maritime and coastal resource management, a course at Michigan State University has been taught since 1996 that attempts to do just this. Major "clusters" of content include:

- descriptions and characterizations of varied natural and cultural aquatic, maritime and coastal resources;
- history of maritime environments and human occupation, including traditional culture, folklife, foodways and cultural relationships;
- commercial, industrial and recreational uses of maritime and coastal resources;

- research methods and techniques applicable to maritime areas (e.g. natural sciences, archaeology /anthropology, underwater archaeology, documentary and oral historical research, social research, artefact conservation and historic preservation as part of research);
- education, public outreach, interpretation and museums as channels through which to formally and informally educate and develop stewardship in residents and visitors to aquatic and coastal areas;
- legislative and legal (statutory and case law) structures underpinning maritime resource use and management; and
- management and planning approaches, to include public involvement.

Students then develop an integrated management plan for an aquatic or maritime resource/ region of their choice. Field trips to coastal communities, conversations with local stakeholders, and class presentations by specialists from many disciplines are all incorporated into this interdisciplinary set of experiences (see Figure 1).

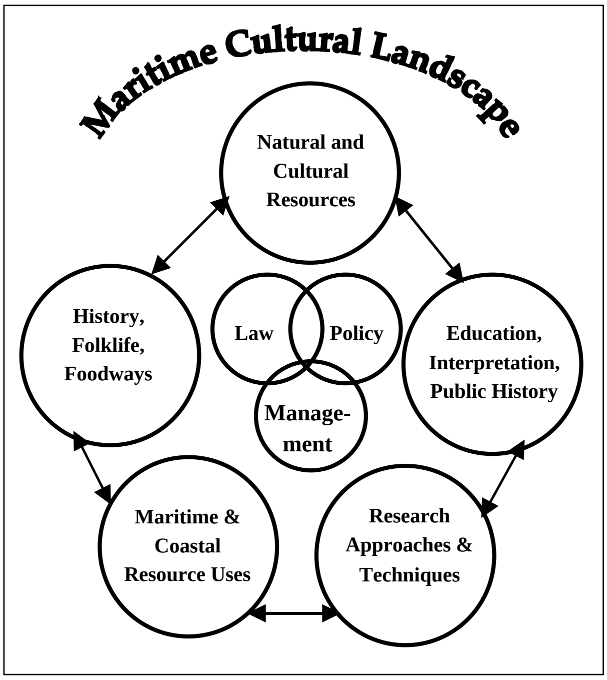

Fig. 1. Maritime and coastal resources management conceptual course model

A second recommendation is for professionals to encourage creation of periodic professional conference opportunities that cross disciplines, and for professionals to consciously attend non-home-discipline conferences. Examples of interdisciplinary conferences do exist, that to greater or lesser degrees facilitate interdisciplinary interactions: International Symposium on Society and Resource Management; the International Symposium on Trends in Tourism and Outdoor Recreation; the World Congress on Coastal and Marine Tourism. Sometimes professional organizations hold joint or coordinated conferences, but these often are in similar disciplines and participants often attend sessions only in their "relevant" tracks. These are all positive efforts, and they should occur more often. Challenges, however, are great, because conference funding and professional reward structures usually serve as disincentives rather than incentives for cross-disciplinary conference participation.

A third recommendation is to provide interdisciplinary learning opportunities for lay people, both youth and adults. Such opportunities for students and adults can occur, often through nonformal settings and opportunities. Examples include a recent *Ecology of the Great Lakes* workshop for K-12 educators. The primary focus was on understanding the biological, chemical and physical properties of Lake Superior, and the impacts of human actions on the lakes, but several sessions did deal with human values and lake resource uses, with lifestyles and cultures associated with lake living and labor, and with coastal and underwater cultural resources. Another example is the one-week Great Lakes Camp for Michigan teenagers who want to explore career opportunities associated with Great Lakes resources. Sea Grant agents and faculty from multiple disciplines participate as camp instructors and leaders. Students are involved in a range of experiences that include physical and natural science experiences, ecosystem management, recreational uses of aquatic resources, and cultural components.

CONCLUSION

Ultimately we are all connected to the seas, both salt and fresh water, and global survival depends on wise use of the resources. If we focus only on managing the resources themselves, and ignore the human factors affecting resource use, we will never be successful. Efforts to wisely use and manage these complex resource systems ultimately will be much more successful and efficient through use of integrated, interdisciplinary, and collaborative approaches in which citizens and professionals share their knowledge, expertise and values in seeking win-win strategies for sustainable resource use and protection.

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Marine Reserves: Time for a Global Perspective

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Abstract

Research on marine reserves, be it field research, management evaluations or modeling, has primarily focused on local or regional scales. This is illustrated with reference to published work during the 1990s at Lizard Island, Great Barrier Reef, on patterns of movement of commercial fishes in relation to marine reserves using visual census, tagging, freeze branding and ultrasonic telemetry. It is now time, however, to address overfishing of ecosystems on the scale at which it is happening in our globalized society, that is on a global scale. There is broad consensus that the world has to deal with massive fisheries overcapacity, fuelled by direct and indirect subsidies, and enhanced by technology creep. The results are globally declining catches and biomass, 'fishing down marine food webs', and ecosystem destruction. Increasingly, studies are demonstrating that marine reserves can play a successful role in fisheries management, but clearly their use needs to go hand in hand with serious and sustained efforts to reduce the overcapacity of all fishing fleets. This effort reduction, however, has to be accompanied by the creation of 'insurance policies', in the form of areas that will be permanently closed to extractive uses, i.e. marine reserves. Ultimately, this needs to be addressed on a global scale, and requires us to reconsider our currently unsustainable approach to marine resources and their use.

***Keywords**:* common pool resources, global perspective, overcapacity, overfishing, subsidies

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Introduction

Marine Reserves (MRs), also referred to by a variety of names such as Marine Protected Areas, Fisheries Closures or No-take Zones, are here defined as 'areas permanently removed from all extractive uses', and are in contrast to Marine Parks, such as Australia's Great Barrier Reef Marine Park, which uses multiple-use zoning as well as marine reserves. The issue of MRs has been reviewed repeatedly (Roberts and Polunin 1991, Dugan and Davis 1993, Halpern and Warner 2002, Russ 2002). Traditionally, much of the global MR research was heavily focused on local (Russ and Alcala 1996a, 1996b, 1998a, b, Zeller and Russ 1998, McClanahan and Mangi 2000) or, at the most regional scales of investigation (Mapstone *et al.* 1996a), focused on fundamental, empirical research topics such as abundance patterns, size distributions, or the essential questions of spillover or recruitment effects of target species. A small component of studies used modelling approaches to investigate the effects of marine reserves (De Martini 1993, Russ *et al.* 1993, Attwood and Bennett 1995, Man *et al.* 1995, Walters *et al.* 1999) or to evaluate reserve management (Alder *et al.* 2002). Few investigations have considered marine-reserve issues on larger or global scales (Russ and Alcala 1999, Roberts *et al.* 2001, Pauly *et al.* 2002).

Marine reserve example

Research that I have been associated with during the 1990s may serve as a typical example of this localized focus, as would the work of many others (see review by Russ 2002). My work focused on coral reef fishes of significance to fisheries in the Indo–Pacific, mainly serranids and lutjanids. The focal species were the common coral trout (*Plectropomus leopardus*, Serranidae), which forms the main target species of the Great Barrier Reef commercial line fisheries (Mapstone *et al.* 1996b), small serranids such as *Cephalopholis cyanostigma*, and small lutjanids such as *Lutjanus carponotatus*. The main topics of investigation related to home ranges and basic patterns of movement and activity (Zeller 1997, 2002), population size estimation (Zeller and Russ 2000), spawning aggregations (Zeller 1998) and patterns of adult fish movements in relation to established marine reserves (Zeller 1996, Zeller and Russ 1998). The field component of this research was carried out at Lizard Island, northern Great Barrier Reef, and the primary methods consisted of a range of marking techniques, including standard external tags, Passive Induced Transponder tags, and freeze branding, combined with capture–recapture tools such as fish traps, hook and line fishing and underwater visual census (Zeller 1996, Zeller and Russ 1998, 2000, Zeller *et al.* 2003). In addition to the more traditional techniques, these studies were the first to successfully use ultrasonic telemetry, a remote-tracking technique, on the Great Barrier Reef (Zeller 1996, 1999), including the application of an automated, remote-tracking system (O'Dor *et al.* 2001).

These studies showed that the non-pelagic reef fishes investigated have relatively small home ranges (e.g. *P. leopardus*, ~10,000–18,000 m^2 Zeller 1997) and limited ranges of movement and activity (Zeller 1997, 2002), with the exception of spawning aggregation activities (Zeller 1998). A related study found no differences in large-scale movements of coral trout inside and outside marine reserves (Zeller and Russ 1998). These results influenced a subsequent investigation of the potential and likely scale of adult spillover across MR boundaries, under conditions of experimentally induced density gradients, and using a range of mark–recapture techniques similar to those used in the previous studies (Zeller and Russ, unpublished data). More than 1300 fish from the three main target families of commercial and recreational fisheries (Serranidae, Lutjanidae and Lethrinidae) were tagged, with recapture rates of 25% over the three-year study period. Preliminary examination of the resulting data indicated that approximately 60% of all recaptures were from the same small spatial area of initial captures (spatial resolution of 50 x 30 m), indicating that the distances moved between recaptures were very limited. Very few fish seemed to move more than 100 m from their area of capture (Zeller *et al.* 2003). These results confirm experimentally what has been becoming increasingly evident (Russ 2002), and that is that many reef fishes have a limited range of activity and movement as adults (here I specifically exclude spawning activities from this generalization).

This, of course, has implications for MRs on coral reefs (and likely elsewhere), with regard to both the scale of their perceived effectiveness as fisheries management tools (if one ignores the potential significance of the recruitment effect (Russ 2002)), as well as the scale of much of the research that is being conducted, because adult spillover on coral reefs is limited in the spatial extent at which it can influence local fisheries.

GLOBAL PROBLEM

Why then is there a need for scientists, resource managers, and especially policy makers, to consider a shift in focus from local or regional pre-occupation to a more global standpoint on MRs? Few studies have attempted to step back and take the bigger picture into consideration (Russ and Alcala 1999, Roberts *et al.* 2001, Pauly *et al.* 2002). I feel that in many cases the level of focus of research has influenced the level of focus of policy and decision making, limiting policy 'vision' to too small a scale. Although this might be understandable in developing countries with their often huge underlying social, economic and political problems, it does not address humanity's larger problem. Fundamentally, the reasons why scientists, managers, and policy makers should be concerned with the global picture are simple, and are outlined below.

In general, fisheries are known to be in trouble around the world (North Atlantic cod, bluefin tuna, Patagonian toothfish, North-eastern Pacific rockfish stocks etc.), but often are still not perceived by the general public as having strong, or even any, impacts on the structure of underlying ecosystems. One reason for the relatively 'benign' perception of fisheries is that their impacts (e.g. declining landings or even individual stock collapses) are usually seen as local issues or problems, and rarely is the complete ecosystem picture considered. Yet, fisheries are a global industry, with fish products forming one of the world's most globalized commodities (Sumaila 1999). This industry works, deals, trades, and reacts at the global scale. Thus, the way scientists, managers and advisors to policy makers have to think and act is at the global level.

The Food and Agricultural Organization of the United Nations (FAO) maintains the only global database of fisheries landings (data reported by member countries). Based on this data the consensus was that global fisheries catches reached a plateau during the 1990s at around 80 million tonnes. These data ignore uncertainties regarding levels of discarding and 'IUU' catches (illegal, unreported and unregulated) (Alverson et al. 1994, Agnew 2000). However, a recent study, correcting for massive over-reporting of catches by the People's Republic of China, showed that the reported world fisheries catches have actually been declining slowly since the late 1980s (Watson and Pauly 2001). That study alone should dramatically change our perspective of the status of global fisheries, and should drastically alter our policy position as well as investment decisions by industry and lending institutions. As long as global catches seemed to be growing, or at least stable, and thus managing to meet global human demand, there seemed to be little public concern, much less national or international intervention. However, if, as that study shows, there is a general decline in global catches, then we have to act.

But not only are we seeing declines in catches, but we are also witnessing changes in the composition of catches that are of great concern, most clearly illustrated by what is now known as 'fishing down marine food webs' (Pauly et al. 1998). This exemplifies itself through declining mean trophic levels of catches, indicating that fisheries, after reducing top-level trophic species, are increasingly targeting fish further down the food web. A good example of this trend is the North Atlantic where it has been shown that the biomass of predatory fishes has declined by approximately two-thirds over the past 50 years (Christensen et al. 2002).

Yet, despite the declining trends in catches and lowering of trophic levels of landings, the global fishing-fleet capacity had grown by over 400% between 1970 and the late 1980s, while at the same time the landing rate had declined from over 6 tonnes to 2 tonnes per registered vessel tonne (Garcia and Newton 1997). According to the FAO, the growth in capacity has slowed during the 1990s, although continuous technological improvements are resulting in ongoing increases in effective catching power (Garcia and Moreno 2001). Thus, it is now generally agreed that a characteristic of many fisheries today is the existence of significant overcapacity in the range of 30–50% of current capacity (Garcia and Newton 1997, Garcia and Moreno 2001).

Thus, we are faced with declining catches and a shift in catch composition to increasingly lower trophic levels, while at the same time we are not succeeding in halting and reversing the growth in effective fishing power. How are we going to address this problem?

GLOBAL SOLUTION

Simplified, the driving forces behind global over-fishing developments are threefold:

1. Fisheries function under the underlying historical concept that marine resources are common pool property (Gordon 1954, Clark 1990);
2. Fisheries are heavily influenced by direct and indirect subsidies to essentially all fishing sectors and fleets around the world (Milazzo 1998, Munro and Sumaila 2002); and
3. There is a continuous technology creep, which increases the effective catching power of fishing fleets (Garcia and Newton 1997, Pauly *et al.* 2002).

How are we going to deal with these global problems? In a review, Pauly *et al.* (2002) suggest two key mechanisms to help us stop the destructive downward spiral of over-exploitation with associated fisheries failures and ecosystem degradation (Christensen *et al.* 2002). Both mechanisms will cause pain in the short term, but seem the only logical solution in the long term. Interestingly, both mechanisms have also been recently acknowledged as key issues during the 2002 Global Summit on Sustainable Development in Johannesburg.

The first mechanism is a massive reduction of subsidies to the fishing sector, to enable market forces to better control unsustainable over-capacity of existing fleets. The required reductions in fishing effort will involve effective decommissioning of a large fraction of the world's fishing fleets, going hand in hand with implementation of fisheries regulations that apply a strong form of the precautionary principle. Although the conceptual elements for this are in place, e.g. in the FAO Code of Conduct for Responsible Fisheries (Anonymous 1995, Edeson 1996), the political will to act has so far been lacking (reflected in the growing number of fisheries collapses throughout the world). Whether the recent agreement calling for placing global fisheries on a sustainable basis by 2015 and eliminating subsidies, arrived at during the 2002 Global Summit on Sustainability in Johannesburg, will provide better impetus for action remains to be seen. However, it is clearly a useful step forward.

The second mechanism is the creation of large-scale marine reserves based on zoning of the entire ocean areas. This would remove large areas of fishing grounds from exploitation, to permit stock and ecosystem rebuilding by effectively setting an upper limit to fishing mortality. This is where I return to the Theme of this Congress. MRs essentially fulfil the function of 'insurance policy', a role that, historically, was performed by natural refuges unavailable owing to distance or to technologically more limited fishing gears (Roberts 2002). This role of natural refuge has been eliminated through technological and capacity expansion of the world's fishing fleets. To undertake this endeavour at the scale ultimately required (ocean-basin scale and globally) will require humanity to contemplate a change in the historically entrenched notion of marine resources as common pool and essentially open access, towards a more applied and enforced notion of long-term global heritage, and to implicitly re-consider the long-held notion of *Mare Liberum* (Russ and Zeller 2003). This approach, if implemented and enforced thoroughly, will permit our societies to finally develop sustainable fisheries with catches likely in excess of present levels, based on resources that are embedded in functional and diverse ecosystems.

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REVIEW OF GENERIC NO-TAKE AREAS AND CONVENTIONAL FISHERY CLOSURE SYSTEMS AND THEIR APPLICATION TO THE MANAGEMENT OF TROPICAL FISHERY RESOURCES ALONG NORTH-WESTERN AUSTRALIA

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Abstract

A large number of generic no-take (sanctuary) areas have been gazetted or planned along the coast of Western Australia. Although the principal purpose of these zones is to conserve biodiversity, they are also considered to be a useful tool to conserve stocks of economically important species. The application of generic no-take areas and species or fishery-specific closure systems, are reviewed in relation to available information from the tropical fishery resources of north-western Australia. The marine environment of north-western Australia supports significant commercial and recreational fisheries for both pelagic and demersal finfish. This review takes a top-down approach, whereby the available biological and fishery abundance data for key commercial and recreational fish species is collated and assessed to determine whether generic no-take areas and/or targeted fishery closure systems, both spatial and temporal, can provide an effective contribution to the management framework needed to ensure sustainable harvest strategies. The key exploited species examined occupy a range of marine environments from the pelagic to inshore demersal reef areas. This review and assessment of the anticipated performance of different closure strategies for each species type utilises data from a series of comprehensive studies undertaken to support ongoing management of commercial and recreational fishing across more than 2000 km of coastline. Outcomes from this review suggest that the generic no-take approach is a relatively inefficient mechanism for maintaining breeding stock levels of economically important finfish species, relative to specifically designed closures and associated fishing effort control systems designed to protect key target species, particularly those with greater mobility. The relative merits of the no-take areas and fishery closures in relation to the sophistication of overall fisheries management controls and the mobility of the target species concerned is discussed.

Keywords: No-take areas, targeted fisheries closures, tropical fisheries management, north-western Australia

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INTRODUCTION

The terms marine reserves, marine protected areas, marine harvest refugia or sanctuaries are often used interchangeably to refer to no-take or no-fishing areas in the marine environment. The principal purpose of these no-take areas (NTAs) is often to conserve biodiversity and important habitat features, and their establishment is often driven by both conservation and social concerns. Social values such as the preservation of seascapes and wilderness, and cultural values such as maritime history, indigenous heritage and use, are all important considerations in the development of NTAs. However, NTAs are also often assumed to have significant benefits for the management of exploited fish stocks (Ballantine 1997; Ward *et al.* 2001). The basis for this assumption has generally been through observations in small-scale studies on closures (Ballantine 1997) or larger-scale studies carried out in developing countries with little capacity for the more complex fisheries management systems operating in Australia and New Zealand (Russ 2002 and references therein).

Recent reviews of the fisheries management benefits of NTAs (Baelde *et al.* 2001; Ward *et al.* 2001) have, however, found few tangible examples of a direct overall benefit to fisheries management. Typically, studies within NTAs all do show the expected local increases in the size and abundance of relatively sedentary species, e.g. lobsters, abalone and wrasses, with the cessation of fishing (Edgar and Barrett 1999). However, these studies generally have not taken into account the redirection of fishing effort on to other sections of the stocks concerned, and hence the overall or net effect of the NTA on the stock as a whole. For temperate waters, a study (C Buxton *et al*., *pers. comm*.) is seeking to model the impacts of effort displacement and loss of production from the Tasmanian NTAs (Edgar and Barrett 1999) to assess the benefits of these closures to the stocks as a whole.

For tropical waters, one of the main bodies of work available is from Russ (2002 and references therein) in the Philippines, where finfish stocks are severely overfished and area closures are the sole management strategy. However, few studies have examined the benefits of NTAs in regions where the finfish species are being maintained at long-term sustainable-harvest levels through input and/or output controls. The tropical finfish stocks of the North West Shelf off Australia, which are subject to a tight management regime and have extensive databases, therefore provide a relatively unique opportunity to assess the likely benefits of NTAs and compare them to targeted fisheries closure (TFC) systems that are currently operating within a sophisticated management regime.

This paper examines the relative applicability and effectiveness of biodiversity-based, but static, NTAs in comparison to TFCs in maintaining the productivity of targeted finfish populations. The distributions of selected species are compared with the spatial extent of the NTAs and TFCs. The species considered were chosen to represent species with a range of depth distributions and varying degrees of mobility, from the highly mobile and pelagic Spanish mackerel (*Scomberomorus commerson*) to the relatively site-specific species such as red emperor (*Lutjanus sebae*). These species provide a basis for predicting the applicability of the various closure systems to species with different life history and mobility traits.

DATA SOURCES

The data used in this paper are predominantly derived from compulsory monthly commercial catch-and-effort statistics (CAES) returns completed by all Western Australian licensed fishing vessels. These data are well validated by landings and transportation data for catches, noting that there is a very limited local market where 'leakage' can occur. Effort data, both spatial and in aggregate for the major licensed fisheries, are now managed and validated through a satellite-based vessel-monitoring system (VMS). Additional biological sampling of the individual species in the catch from the major fisheries has been provided by research studies that have included on-board observations of commercial vessel catches and factory catch sampling.

In addition, fishery-independent research vessels have collected more detailed data including those from a comprehensive trawl survey of inshore waters from which the commercial finfish fishery is excluded. For the Kimberley region, studies have generated detailed distributional and biological knowledge of the exploited stocks (Nowara and Newman 2001; Newman *et al.* 2001; Newman and Dunk 2002, 2003). In the Pilbara, studies have included manipulation of a significant trawl fishery over a two-year period (Stephenson and Dunk 1996; Stephenson and Mant 1999; Newman *et al.* 2000; Newman 2002a, 2002b, 2002c). For the pelagic fisheries sector, a large study of the Spanish mackerel stocks has provided comprehensive data on this species (Mackie *et al.* 2003; Buckworth *et al.* unpublished).

For the recreational sector, opportunistic surveys were undertaken at boat ramps and aboard charter vessels in the Pilbara region in the early 1990s (Moran *et al.* 1995), and a 12-month survey of recreational fishing activities was undertaken in the Pilbara region from December 1999 through to November 2000 (Williamson *et al.* unpublished). A detailed biological study has also been completed on a key recreational target species (*Epinephelus rivulatus*) not covered by the various comprehensive fishery-based studies (Mackie and Black 1999; Mackie 2000, 2003).

In the region between North West Cape and the Northern Territory border, three major marine reserve systems have been established at Ashmore Reef, Cartier Reef and the Rowley Shoals. However, in total 15 marine parks have been proposed throughout the region including the Montebello/Barrow Islands proposed marine Park and the Dampier Archipelago/Cape Preston proposed Marine Park, following the report by the Marine Parks and Reserves Selection Working Group (CALM 1994).

FISHERIES OF THE NORTH WEST SHELF

The North West Shelf region of Australia has been subject to fishing since the 1960s, when a small Japanese fishery was operating. Through the 1970s and 1980s an extensive Taiwanese pair-trawl fishery operated over most areas outside the then 12-mile limit (Sainsbury 1987; Sainsbury *et al.* 1997; Nowara and Newman 2001). Catches from this fishery, particularly that of the larger more valuable species, declined from a peak of close to 40,000 tonnes to less than 10,000 tonnes and it was phased out in 1990. Following a recovery of the stocks, a number of smaller Australian vessels were encouraged to explore the area, leading to the development of several managed fisheries under Western Australian jurisdiction, with associated research programs.

The North West Shelf study site is an ideal location for this review; it has a full fishing history, several comprehensive studies have been completed since the mid 1990s on all the major commercial target species, and the magnitude of the recreational catch has been assessed and is to be published in the near future. The commercial fisheries of north-western Australia are managed under new and innovative fisheries management involving total allowable effort/individually transferable effort regimes using modern satellite-based technologies for compliance. The implementation of marine parks in the region is under way, and their application will benefit from this work on the relevance of NTAs (sanctuaries) to fisheries stocks.

The key exploited species examined in this review occupy a range of marine environments from the pelagic to inshore demersal reef areas; the key species are Spanish mackerel, *Scomberomorus commerson* (pelagic), goldband snapper, *Pristipomoides multidens* (deepwater demersal), red emperor, *Lutjanus sebae* (offshore demersal), Rankin cod, *Epinephelus multinotatus* (offshore demersal), blue-spot emperor, *Lethrinus hutchinsi* (inshore–offshore demersal) and spangled emperor, *L. nebulosus* (inshore–offshore demersal). Troll-based line fisheries target Spanish mackerel, and primarily fish traps and fish trawls target the species associated with demersal reefs across their range. These six target species drive the main fisheries of north-western Australia. Catches of these six species in 2000 comprised approximately 1500 tonnes out of a total of 3000 tonnes landed annually by the commercial fleet (Penn 2001). The recreational catch for the Pilbara and West Kimberley region totals about 320 tonnes (Williamson *pers. comm.*), mostly from nearshore waters, and are therefore not a significant factor in exploitation at this stage.

Blue-spot emperor (*L. hutchinsi*) is distributed throughout north-western Australia, with the highest catches being landed from the central Pilbara region (Fig. 1). It has rapid initial growth and attains a maximum age of only 14 years (Table 1). Maturity is reached within the first two years of life and spawning occurs only in September throughout the range (Table 1). Juveniles occupy nearshore reef habitats. Trawling and trap fishing is prohibited through this inshore zone and, hence, the juveniles are not subject to significant exploitation pressure. A legal minimum length (LML) of 280 mm total length (TL) is applied to this species, allowing most fish to attain maturity before becoming vulnerable to capture. The stock structure is not known, however, the adult fish are found offshore in deeper waters, suggesting cross-shelf movements of this species. As in most reef fish, longshore movement patterns after recruitment to the fishery are likely to be limited. Pilbara and Kimberley populations are therefore considered separate for the purposes of fisheries management.

Spangled emperor (*L. nebulosus*) is distributed throughout north-western Australia, with catches variable throughout the region (Fig. 2). It is a slow growing and long-lived fish with a maximum age of at least 27 years (Table 1). Maturity is achieved at approximately 3.5 years of age, after which spawning occurs from October to January throughout the range with peak spawning in December (Table 1). Juveniles are found in habitats similar to those of adults around reefs. A LML of 410 mm TL is in place for this species. Exploitation pressure on juveniles is considered to be limited. Although the stock structure of spangled emperor has not been investigated, the Pilbara and Kimberley populations are considered separate for the purposes of fisheries management.

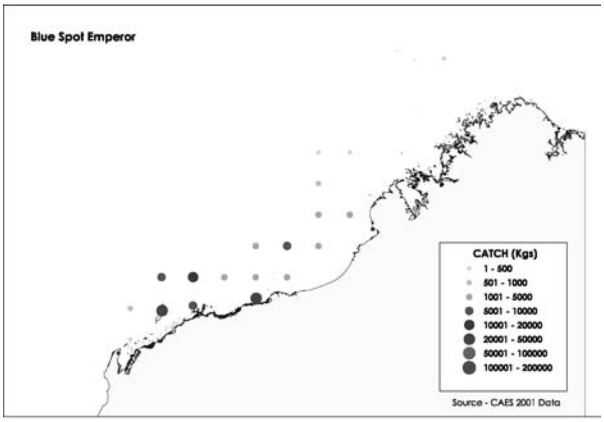

Fig. 1. Spatial distribution of the catch of blue-spot emperor, *Lethrinus hutchinsi* across northwestern Australia.

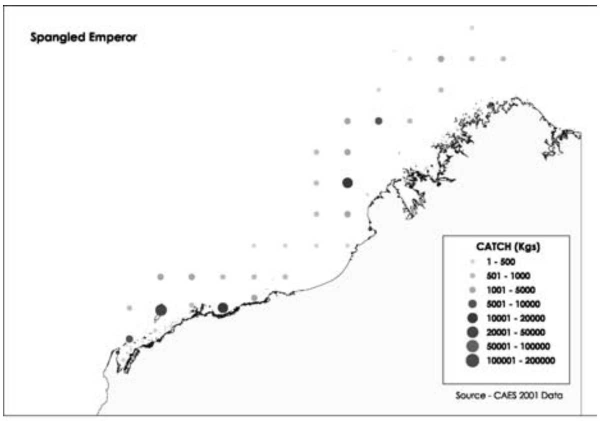

Fig. 2. Spatial distribution of the catch of spangled emperor, *Lethrinus nebulosus* across northwestern Australia.

Table 1. Growth and mortality parameters, reproductive season and critical habitats of each selected species in northwestern Australia.

Species	Growth and Mortality Parameters				Longevity	Spawning Season	Peak Spawning	Sexual Maturity	Proportion Female	Depth range	Juvenile habitats	Adult habitats
	L_∞ (mm)	K	t_0 (yr)	M	(yrs)			(FL mm, yr)	(%)	(m)		
Lethrinus hutchinsi [1]	315	0.716	-0.472	0.300	14	Sept.	Sept.	240, 1.8	50	5-100	Inshore reef	Inshore-Offshore reef
Lethrinus nebulosus [2]	568	0.221	-1.49	0.155	27	Oct.-Jan	Dec.	380, 3.5		0-100	Offshore reef	Offshore reef
Epinephelus multinotatus [1]	666	0.221	-1.835	0.180	23	Aug.-Oct	Oct.	391, 2.2	50	20-150	Inshore reef	Offshore reef
Lutjanus sebae (male) [1]	699	0.165	-1.496	0.100	40	Sept.-Mar.	Oct.			5-140	Cross-shelf reef	Cross-shelf reef
Lutjanus sebae (female) [1] (Pilbara population)	549	0.235	-1.57	0.110	37	Sept.-Mar.	Oct.	392, 3.8	50	5-140	Cross-shelf reef	Cross-shelf reef
Lutjanus sebae (male) [3, 4]	628	0.151	-0.60	0.122	34	Oct. & Mar.	Oct.	457, 8.0		5-140	Cross-shelf reef	Cross-shelf reef
Lutjanus sebae (female) [3, 4] (Kimberley population)	483	0.271	0.07	0.122	30	Oct. & Mar.	Oct.	429, 8.2	60	5-140	Cross-shelf reef	Cross-shelf reef
Pristipomoides multidens [3, 5]	598	0.187	-0.17	0.139	30	Jan.-Apr	Mar.	470, 8.2	45	60-200	Deepwater – sand	Deepwater – reef
S. commerson (male) [6]	1155	0.692	-0.293	0.23	22	Sept.-Jan.	Oct.-Nov.	809,<2	55	5-100	Inshore-structure	Inshore-Offshore reef
S. commerson (female) [6] (Pilbara population)	1259	0.631	-0.285	0.19	18	Sept.-Jan.	Oct.-Nov.	628,<2	45	5-100	Inshore-structure	Inshore-Offshore reef
S. commerson (male) [6]	1067	0.847	-0.211	0.23	12	Oct.-Jan.	Oct.	809,<2	55	5-100	Inshore-structure	Inshore-Offshore reef
S. commerson (female) [6] (Kimberley population)	1219	0.646	-0.262	0.19	11	Oct.-Jan.	Oct.	628,<2	45	5-100	Inshore-structure	Inshore-Offshore reef

Data sources
1 Stephenson and Mant (1999)
2 Moran *et al* (1993)
3 Newman *et al* (2001)
4 Newman and Dunk (2002)
5 Newman and Dunk (2003)
6 Mackie *et al* (2003)

Rankin cod (*E. multinotatus*) is distributed throughout north-western Australia, but catches are concentrated in the Pilbara region (Fig. 3). It is a slow growing and long-lived fish with a maximum age of at least 23 years (Table 1). Rankin cod are protogynous hermaphrodites, with spawning occurring from August to October throughout the range with a peak in October (Table 1). Maturity of female fish is achieved after the first two years of life, with the larger and older fish in the population being all males. Juveniles inhabit nearshore reef habitats, but exploitation pressure on juveniles is considered to be limited. The stock structure of Rankin cod has been investigated, and adult assemblages have been shown to remain separate throughout their life history (Stephenson *et al.* 2001). Hence, Pilbara and Kimberley populations are separate for the purposes of fisheries management.

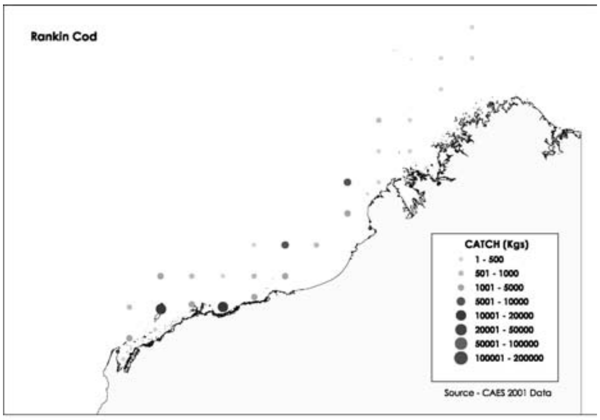

Fig. 3. Spatial distribution of the catch of Rankin cod, *Epinephelus multinotatus* across northwestern Australia.

Red emperor (*L. sebae*) is distributed throughout north-western Australia, with catches consistent throughout the region (Fig. 4). It is a slow growing and long-lived fish with a maximum age of at least 40 years (Table 1). Female fish mature at 4–8 years of age and spawning occurs from September to March throughout the range, peaking in October (Table 1). Juveniles are found across the continental shelf. Hence, red emperor populations are vulnerable to capture below the size and age at maturity. Since adult assemblages remain separate throughout their life history (Stephenson *et al.* 2001), Pilbara and Kimberley populations are considered separate for the purposes of fisheries management.

Goldband snapper (*P. multidens*) is distributed in depths between 60 and 200 m across north-western Australia, with catches consistent throughout the region (Fig. 5). It is a slow growing and long-lived fish with a maximum age of at least 30 years (Table 1). Maturity of female fish is achieved at approximately eight years of age, after which spawning occurs from January to April throughout the range, with peak spawning in March (Table 1). Juveniles are found in featureless sandy habitats in deep water, separate from sub-adult and adult fish. Goldband snapper are, however, vulnerable to capture below the size and age at maturity. The stock structure of goldband snapper shows that adult assemblages remain separate throughout their life history (Newman *et al.* 2000). Hence, Pilbara and Kimberley populations are separate for the purposes of fisheries management.

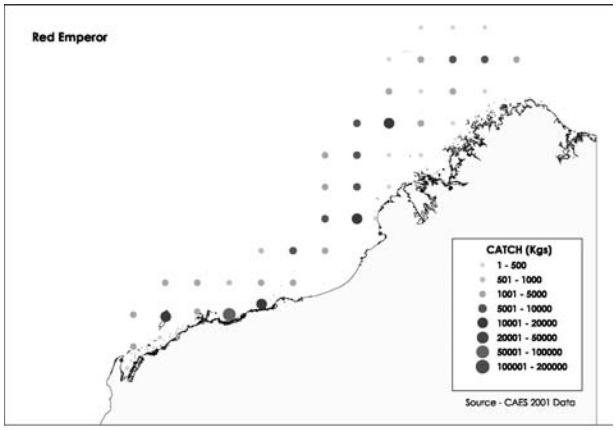

Fig. 4. Spatial distribution of the catch of red emperor, *Lutjanus sebae* across northwestern Australia.

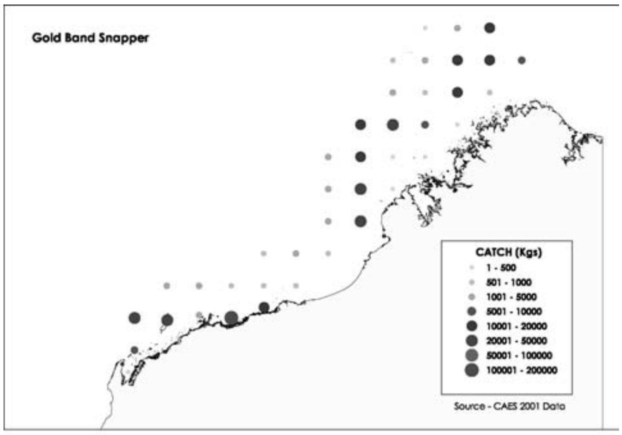

Fig. 5. Spatial distribution of the catch of goldband snapper, *Pristipomoides multidens* across northwestern Australia.

Spanish mackerel (*S. commerson*) is distributed throughout north-western Australia, with the highest catches being from the Kimberley region (Fig. 6). It displays rapid initial growth and can attain a maximum age of at least 22 years (Table 1). Maturity is reached within the first two years of life, and thereafter spawning occurs from September to January throughout the range, with peak spawning in October–November (Table 1). Juveniles occupy nearshore coastal habitats with some form of physical reef structure. Early life-history stages are not subject to significant exploitation, with most fish attaining maturity before becoming vulnerable to capture.

Adult assemblages remain separate, with a limited amount of gene flow among adjacent populations. Since adult assemblages mix over scales of approximately 100 km in this region, the Pilbara and Kimberley populations can be considered separate for the purposes of fisheries management.

It is important to have a good biological understanding of the exploited species and the distribution of fishing to determine the scale and type of management that is required. For each of these species we know their distribution and relative abundance and more importantly what life-history stages are affected by exploitation. From the biological data we have an understanding of their production potential and the requirements for maintenance of the spawning-stock biomass as well as the expected time frames for stock recovery from depletion events. Timing of spawning and the length of the spawning season (Table 1) are important factors when considering temporal closures.

The overall aim of fisheries management in the region is to limit the exploitation rate to maintain spawning stocks above their biological reference points across the distribution of each species (Tables 2 and 3).

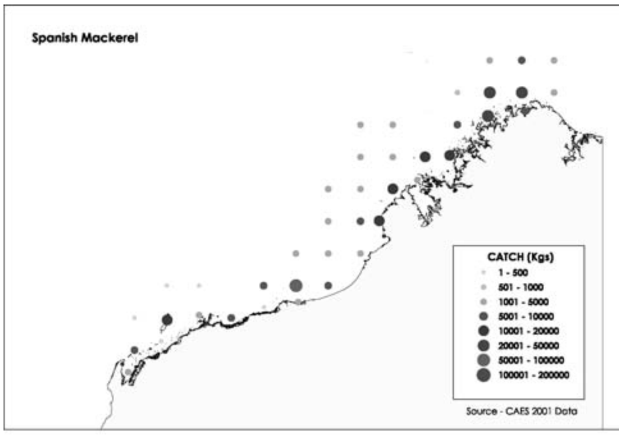

Fig. 6. Spatial distribution of the catch of Spanish mackerel, *Scomberomorus commerson* across northwestern Australia.

Table 2. Annual harvest rates for each species as a proportion of stock size and spawning biomass target level as a proportion of virgin biomass.

Species	Annual Harvest Rate (% of stock size)	Spawning Biomass Target Level (% of virgin level)
Blue spot emperor	20-25	30-40
Spangled emperor	10-15	> 40
Rankin cod	10-15	> 40
Red emperor	10-15	> 40
Goldband snapper	10-15	> 40
Spanish mackerel	15-25	30-40

Table 3. Current fishery management control mechanisms applied to target northwestern Australia finfish resources.

Species	Fishery Management Control Mechanisms							
	Commercial						Recreational	
	Limited Entry	Gear Limits	ITEs	VMS	TFC	NTA	Size Limit	Bag Limit
Blue spot emperor	✔	✔	✔	✔	✔		✔	✔
Spangled emperor	✔	✔	✔	✔		(✔)	✔	✔
Rankin cod	✔	✔	✔	✔	✔			✔
Red emperor	✔	✔	✔	✔	✔		✔	✔
Goldband snapper	✔	✔	✔	✔	✔			
Spanish mackerel	✔	✔	✔	✔			✔	✔

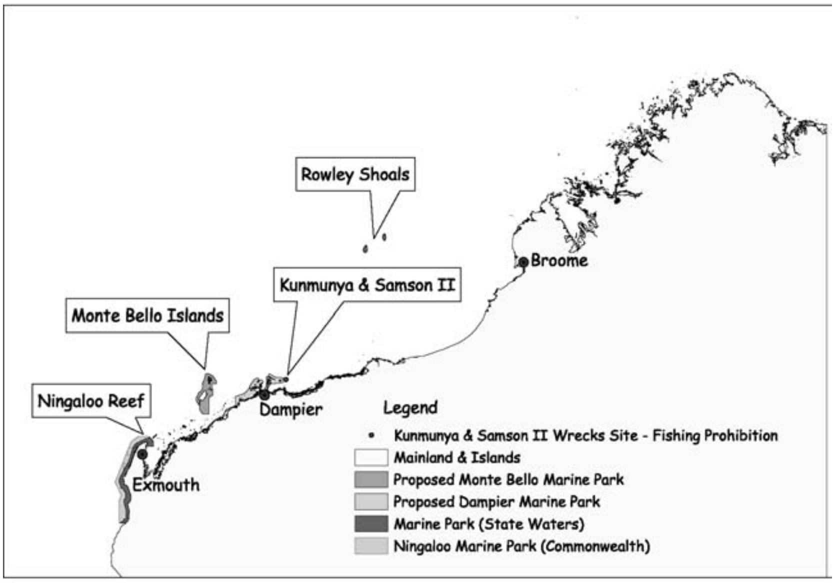

Fig. 7. Current and proposed marine parks and sanctuary areas across northwestern Australia.

The Ningaloo Marine Park is on the North-West Cape (Fig. 7), where ~1420 km^2 is closed to commercial fishing and ~220 km^2, comprising mainly fringing coral reef, is closed to all types of fishing (sanctuary zones). Two additional marine parks are proposed: the Montebello/Barrow Islands and Dampier Archipelago/Cape Preston Marine Parks (Fig. 7), which are both likely to encompass ~2500 km^2, with a few small NTAs. These proposed marine parks are based around offshore or nearshore islands encompassing coral reef habitats.

The fish-trawl fishery is confined to a specific area on the broad continental shelf with a large-scale targeted spatial closure in the centre of the fishery to protect the spawning-stock biomass of red emperor and Rankin cod (Fig. 8).

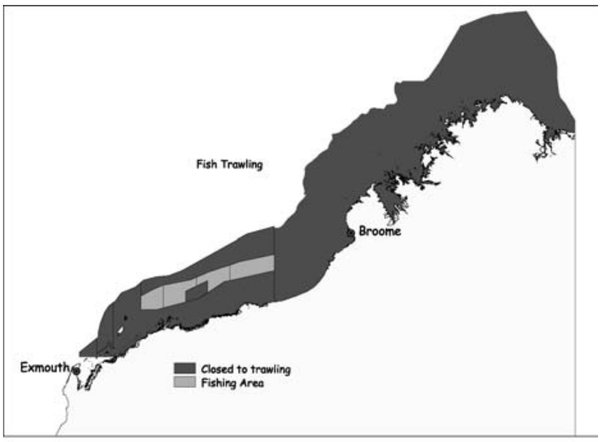

Fig. 8. Location of the Pilbara Fish Trawl Fishery indicating the area along the coast of northwestern Australia that is available to the fish trawl fishery and that area of the coast in which fish trawling is prohibited.

This area was set aside because it was an area that had historically yielded high catch rates of both species. This spatial closure applies to trap and fish-trawl fishing only, with other fishing activities such as pearling being allowed in the area. Moreover, the offshore zone of the Fish Trawl Fishery from 100 to 200 m depth is closed, which facilitates protection of a portion of the spawning-stock biomass of goldband snapper. In addition, no fish trawling is allowed inshore of the 50 m depth contour, which prevents conflicts with other user groups and protects the juvenile stages of a number of species.

The Pilbara Trap Fishery and the Northern Demersal Scalefish Fishery (NDSF) are both principally fish-trap fisheries (although line fishing is optional for fishers in the NDSF) that are widely distributed along the continental shelf of north-western Australia. Each of these fisheries has an inshore closure to trap fishing that extends along the entire coast and thus prevents conflicts with other user groups (Fig. 9).

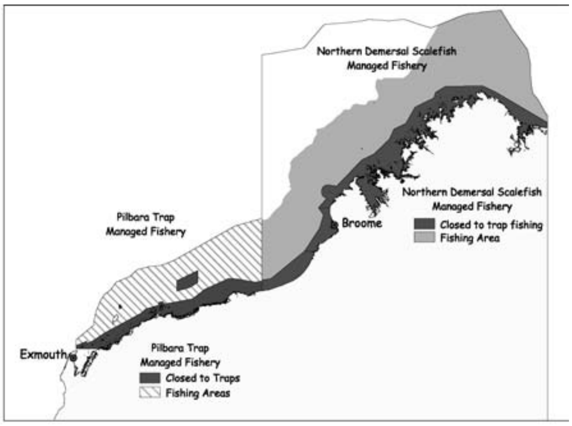

Fig. 9. Pilbara Trap Fishery and the Northern Demersal Scalefish Fishery indicating the area along the coast of north-western Australia available for fish trapping and coastal area where fish trapping is prohibited.

DISCUSSION

All the key species examined are widely distributed throughout the fishing areas in north-western Australia and their catches are relatively consistent from year to year. Spawning occurs across the fishery with no known single large spawning aggregations for any of the species examined in this review. All these species have a pelagic larval stage that may facilitate gene flow and larval mixing among distinct adult assemblages. Therefore, there is uncertainty as to what part of the breeding population of each adult stock is most important to the overall stock. Thus, the key element for fishery managers is to ensure that breeding-stock levels are maintained across all sectors. This strategy also seeks to ensure that indirect effects of fishing on adjacent adult populations through reduced recruitment are limited.

Spanish mackerel, goldband snapper, red emperor, Rankin cod and blue spot emperor all have wide distributions across the North West Shelf and tend to occupy and spawn in habitats that are expansive throughout the region. Since the NTAs in existing and proposed marine parks in the region make up a very small proportion of the overall area in which these species occur and spawn, they are unlikely to provide significant protection to the spawning stock of each of these species. For the spangled emperor, however, the generic NTAs may be more effective, because its distribution is limited to coral reef habitats (Table 4). These habitats are relatively restricted in their distribution along the North West Shelf and are highly represented in the extant and proposed NTAs. In the case of Ningaloo Marine Park, the restriction of fishing to recreational activities immediately outside these zones would further limit fishing pressure in this region. However, the small size of the existing NTAs creates uncertainty regarding their effectiveness in providing a net benefit to maintain the spawning stock of this target species. Furthermore, any net benefit is likely to be localised. The NTAs depicted in Figure 7 contribute little to the effective management of the fish resources described above because they are small and isolated relative to the overall stock distributions.

In Western Australia, NTAs are established as part of the marine reserve system under the Conservation and Land Management Act 1984. These NTAs are created for the conservation and restoration of the natural environment, the protection, care and study of indigenous flora and fauna, and for the preservation of features of archaeological, historic or scientific interest. Therefore, the objective of NTAs differs from that of TFCs. Although fisheries management systems in Western Australia focus primarily on limiting fishing mortality and fishing capacity through input controls, spatial area closures are also used where appropriate to protect parts of the fishable stock (adults and/or juveniles) and also to prevent or limit conflict among competing user groups. TFCs have most of the benefits of NTAs, but they are used in a more practical manner for fisheries management because they relate specifically to the exploited component of stocks rather than the unexploited component.

In order to protect the juveniles of a number of species and to limit conflict among user groups (commercial, recreational, indigenous), the nearshore area extending out from the shoreline to a line approximating the 50 m depth contour (up to ~30 km offshore in some areas) has been closed to trap fishing along the 2000 km coast encompassing both the Pilbara and Kimberley management zones. Since the juveniles of species such as blue-spot emperor and Rankin cod occur predominantly in the nearshore areas, these species are likely to significantly benefit from such closures. However, such closures are unlikely to substantially protect the juveniles of species such as spangled emperor and red emperor whose juveniles have a wide cross-shelf distribution or occupy similar habitats to their adults.

Prior to 1998 the Pilbara Fish Trawl Fishery operated in an area of ~26685 km^2. However, in 1998, a TFC with an area of ~3000 km^2 was implemented to protect the spawning stocks of red emperor and Rankin cod from fish trawling and trapping in this area. This large area of protection no doubt protects a large part of the spawning stock of these species. TFCs are more likely than generic NTAs to benefit those species under exploitation and to meet the needs of fisheries management in Western Australia. Targeted spatial and temporal closures as part of the overall fisheries-management package are considered to provide the optimum benefit to the exploited stocks by protecting key components of the spawning-stock biomass and protecting juvenile nursery areas in many cases. However, the use of these TFCs in isolation would not be an effective fisheries management strategy, because effort could expand and concentrate on the remaining parts of the accessible stock, such that the overall level of exploitation would exceed the biological reference points.

Fisheries management systems therefore require flexibility and should include a range of harvest-control strategies, such as input controls that limit the amount of available fishing effort, limited entry, TFCs and so on, in order to allow for effective sustainable management of fisheries resources. For the trap fisheries in north-western Australia, a total allowable effort (TAE) allocated as individual transferable effort quotas (ITEs) serves as a basis to restrict productive inputs. These ITEs limit the number of traps fished per vessel and the number of days allocated per year to fish in the fishery to match a predetermined notional TAC. Initial access to each fishery was limited following a formal consultative process that included input from all relevant sectors such as recreational fishers, commercial fishers and conservation groups. Defining the eligible commercial fishing vessels was a critical step in the TAE/ITE process and has allowed an equitable allocation of effort quota among all licensees. This allocated-effort model also enables trading in fishing rights to encourage economic efficiency. This allows individual fishers to optimise the amount of access (fishing time gear units) required to maintain or enhance their fishing operations. The basis of the ITE quota is the allocation of a finite number of days to be fished in the fishery.

This procedure can be applied to other fisheries and provides a mechanism for limiting effort, assuming a satisfactory level of compliance and enforcement. The demersal trap and fish-trawl fisheries in Western Australia are regulated via a satellite-tracking system that obtains detailed spatial data on the distribution of fishing effort and monitors the use of fishing days. The boundaries for these fisheries are set on the basis of our knowledge of the stock mobility (from otolith isotope analysis) of the major species, red emperor, Rankin cod and goldband snapper (Stephenson *et al.* 2000; Newman *et al.* 2001). Adult populations of these demersal species do not move across the fishery-management boundaries, or the movement is negligible (Stephenson *et al.* 2000, Newman *et al.* 2001).

CONCLUSIONS

Fishery management systems in Western Australia have the overall objective of limiting fishing mortality to ensure that the breeding stock levels of key target species are maintained above their biological reference points. In north-western Australia, this is achieved through controlling fishing capacity by utilising a total allowable effort (TAE) control system that is allocated through individually transferable effort units (ITEs). Effort is allocated on a spatial basis, with substantial areas closed to the trap or trawl fishery, but not necessarily closed to all other fishing activities such as pearl oyster collection. These spatial closures, focus on the exploited species, keeping the overall harvest levels relatively low while protecting representative fish habitats and minimising impacts on biodiversity, community structure and ecosystem processes.

Ultimately the credibility of management agencies that promote and administer spatial area closures will be judged by the relevance of the areas chosen for closure in relation to those species that require protection of some portion of their spawning stock. Moreover, the stock structure of each species needs to be determined in order to match the spatial extent of any closures that may be needed to control the overall exploitation rates relative to the mobility of the target species.

The use of spatial area closures in isolation is not a panacea, but it is an important component of fishery management systems, as closures offer both conservation and sustainable exploitation benefits. This review has shown that, for the assessed species, the existing and proposed generic NTAs as part of the Western Australian marine park system are unlikely to provide any significant benefit within the context of the overall fisheries management of these species. This conclusion is based on the wide distribution and different habitat affinities of each species and the small size of the generic NTAs and the inability to design a single NTA on a large scale that could be relevant to more than a few species. Any large NTA would automatically constrain other fishing activities, for example pearl oyster fishing, which is unlikely to have an impact on either habitat or biodiversity. Such NTAs are therefore difficult to justify in terms of meeting fisheries management objectives, whereas they meet the objectives of different industry sectors, such as tourism. In contrast, the specific TFC, such as those applying to red emperor and Rankin cod in the fish trawl fishery, specifically control the exploitation of the species concerned without unnecessary secondary impacts on other benign fishing activities. These TFCs do allow for the cascading effects of red emperor and Rankin cod eggs and larvae, which is one of the main attributes used to justify NTAs.

Therefore, targeted spatial and temporal fisheries closures as part of an overall fisheries management package are considered to provide the most practical solution to protecting key components of the spawning stock biomass and/or protecting juvenile recruitment areas that ultimately underpin spawning biomass. NTAs in isolation are a relatively blunt fisheries management tool, however, they are very applicable in the absence of other management tools.

Any cascading benefit or flow into other areas from NTAs will also depend on the mobility of the species and the level of exploitation and fishing controls outside the NTAs. This possible positive cascading effect outside of the NTA has to be compared with the negative impact of lost production from the stock locked into the NTA. There is, therefore, a need to undertake detailed quantitative assessments of the impact of generic NTAs on the dynamics of key finfish stocks and to assess the net changes in spawning biomass and egg production of fish stocks for each species relative to their degree of mobility. The outcomes from this review suggest that the generic NTA approach is likely to be a relatively inefficient mechanism for maintaining breeding stock levels across a range of species, relative to specifically designed closures (TFCs) to protect key target species, particularly those with greater mobility within complex fisheries management systems.

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LAW REACHES NEW DEPTHS: THE ENDEAVOUR HYDROTHERMAL VENTS MARINE PROTECTED AREA

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Abstract

The establishment of the Endeavour hydrothermal vents Marine Protected Area (MPA) within Canada's Exclusive Economic Zone (EEZ) constitutes a significant step forward in the conservation of deep-sea biodiversity. It is one of only a few deep-sea MPAs that have adopted an ecosystem approach. It attempts to reconcile the conflicting objectives of deep-sea conservation and continued access to hydrothermal vent ecosystems for scientific research. The development and management of the MPA has been characterised by a high level of stakeholder involvement. Two issues, the potential for exploitation of the genetic resources of hydrothermal vents and aboriginal title to the seabed, appear not to have been addressed. Nonetheless, it may act as a model for other MPAs including a proposal for a similar MPA in Portugal's EEZ.

***Keywords**:* deep-sea biodiversity, hydrothermal vents, deep-sea marine protected areas, aboriginal title to the seabed, bioprospecting

INTRODUCTION

Hydrothermal vents, or deep-sea submarine fissures, support unique ecosystems with high levels of biodiversity and high levels of endism. Despite extremes of depth and pressure, these remote ecosystems are increasingly under threat from human activity. This paper examines a proposed marine protected area (MPA) for several hydrothermal vents within Canada's Exclusive Economic Zone (EEZ) that have been subject to intensive scientific research. The MPA is proposed through draft regulations pursuant to Canada's *Oceans Act, 1996.* It is anticipated that formal proclamation will occur in September 2002 (*pers comm* Canadian Department of Fisheries and Oceans (CDFO)). The paper briefly examines the ecology of hydrothermal vents and the threats to their associated ecosystems. The paper then examines the policy objectives underlying the MPA's development, stakeholder involvement, and the proposed Management Plan. The failure of planners to examine issues associated with the potential for the exploitation of hydrothermal vent genetic resources and the possibility of aboriginal title to the seabed is also noted. The paper concludes with a brief overview of the World Wide Fund for Nature (WWF) proposal for an MPA for the *Lucky Strike* hydrothermal vent field under the Convention for the Protection of the Marine Environment of the North East Atlantic (OSPAR Convention), and with some observations on how the Canadian experience may guide the development of that MPA.

THE ECOLOGY OF HYDROTHERMAL VENTS – UNIQUE ECOSYSTEMS

To date, more than 100 hydrothermal vent sites have been identified around the world (Ré 2000). The most recently discovered are twelve on the Gakkel Ridge (which runs under the Arctic Ocean from north of Greenland to Siberia) and 13 sites identified within New Zealand's EEZ.

The discovery of hydrothermal vents in 1977 was arguably one of the most significant developments in oceanography of the 20th Century (Dando and Juniper 2001). Whereas the ocean floor had been regarded as akin to a desert, devoid of life, it is now known that hydrothermal vents support amazingly diverse and rich ecosystems. Of the approximately 500 species discovered around hydrothermal vents to date, nearly 80% are endemic to hydrothermal vents and new to science (Dando and Juniper 2001). Species include giant clams, mussels, the giant tube worm (*Riftia pachyptila*), brachyuran crabs, galatheid crabs, turrid gastropods, limpets, polychaetes, pink bythitid vent fish, barnacles, brittle stars, sea stars, anemones, sponges, soft corals and hairy snails (*Alviniconcha hessleri)* (Lutz and Kennish 1993).

Significantly, hydrothermal vent areas host one of the highest levels of microbial diversity and animal abundance on earth (CDFO 2001b). Key species are hyperthermophilic Bacteria and Archaea (Butler *et al.* 2001), which thrive in the extremes of heat and pressure and in the unusual chemistry of the hydrothermal vents. Bacteria oxidise sulfides, together with other chemicals released from hydrothermal vents such as hydrogen, iron or manganese. In so doing they serve as the base of the food chain. Many of these bacteria have formed symbiotic relationships with several other species.

The discovery of hydrothermal vents and their ecosystems driven by chemosynthesis has reanimated debate within the scientific community as to the origin of life on earth and the search for life elsewhere in the Universe. The theories as to the origin of life include those that argue that the first life on earth may have emerged at hydrothermal vents. It is theoretically possible (albeit highly improbable) that research into hydrothermal vents may ultimately yield an answer to this question. This of itself, quite apart from arguments in favour of the need to conserve biodiversity, is strong justification for a strict application of the precautionary principle (Leary 2001).

The threats to hydrothermal vent ecosystems that have been identified are marine scientific research, bioprospecting for genetic resources, deep-seabed mining and deep-sea tourism (Dando and Juniper 2001). We also have little understanding of the impacts of pollution and the introduction of alien invasive species on deep-sea habitats. By far the most immediate of these threats is marine scientific research. Impacts identified to date include the following: habitat loss and organism mortality as a result of removal of chimneys and rocks for geological investigations or chemical sampling; environmental manipulation, such as drilling, which can change fluid flow pathways and shut off the supply of fluids to colonies of vent organisms; clearing fauna for experimental studies; transplantation of fauna between locations; placement of instrument packages that disturb fauna and change water flows; and the use of submersibles and remotely operated vehicles (including the impact of light from submersibles on photosensitive organisms) (Dando and Juniper 2001). The impact of scientific research is further compounded by the fact that most research is confined to only a few sites that are visited repeatedly.

CANADA'S OCEANS ACT

The main objective of the *Oceans Act 1996* is the establishment of a framework for oceans resource management and marine environmental protection in Canada (CDFO 1997b). Underlying that objective are three fundamental principles: (*1*) sustainable development; (*2*) integrated management of activities in estuaries, coastal waters and marine waters that form part of Canada or in which Canada has sovereign rights under international law; and (*3*) the precautionary approach. Although such principles sound good in the abstract, the *Oceans Act* does not clearly define what they mean nor does it provide guidance as to how they are to be implemented. There are no clear definitions of key terms including integrated management and the precautionary approach (Hatcher 2002). Interpretation of these terms is made more problematic by the fact that the legislation was enacted and implemented some five years before Canada had formulated its Oceans Policy. To a considerable extent many of these principles and the way they are to be implemented have, however, now been defined in two recently released documents: (1) *Canada's Ocean Strategy* (CDFO 2002a); (2) *Policy and Operational Framework for Integrated Management of Estuarine, Coastal and Marine Environments in Canada* (CDFO 2002b).

However, it is clear that MPAs are a key component in achieving the stated objectives of the *Oceans Act*. The *Oceans Act* mandates the development and implementation of a national system of marine protected areas. Under section 35(1) Canadian MPAs are defined as

"an area of the sea that forms part of the internal waters of Canada, the territorial sea of Canada or the exclusive economic zone of Canada and has been designated…for special protection for one or more of the following reasons:

1. *the conservation and protection of commercial and non-commercial fishery resources, including marine mammals, and their habitats;*
2. *the conservation and protection of endangered or threatened marine species, and their habitats;*
3. *the conservation and protection of unique habitats;*
4. *the conservation and protection of marine areas of high biodiversity or biological productivity; and*
5. *the conservation and protection of any other marine resources or habitat as is necessary to fulfil the mandate of the Minister*."

The establishment of MPAs has proceeded in the absence of any formal articulated policy as to how they are to be created. Although an MPA Policy (CDFO 1999) was issued in March 1999 this document does not contain details as to the policy and procedure to be adopted by Canadian Department of Fisheries and Oceans (CDFO) in establishing MPAs. In the case of Canada's west coast a draft discussion paper (CDFO 1998) was released in 1998. Despite public consultation and input into this document, "a revised strategy that has considered the extensive public input has still not emerged" (Canada House of Commons 2001). The recent parliamentary review of the *Oceans Act* endorsed views expressed before it of "a major policy vacuum" (Canada House of Commons 2001). To date, no regulations establishing MPAs have been proclaimed. If the *Endeavour Hydrothermal Vents Marine Protected Area* is formally established, it will have taken nearly 6 years for the provisions of the *Oceans Act* to have been implemented.

THE ENDEAVOUR HYDROTHERMAL VENTS

The Endeavour Hot Vents Area is part of the Juan de Fuca Ridge System and lies in water 2250 m deep ~250 km south-west of Vancouver Island off Canada's Pacific Coast, within Canada's EEZ (CDFO 2001a). It is considered to be the most biologically productive and diverse hydrothermal vent site along the Juan de Fuca Ridge. At least 60 distinct species are native to the Juan de Fuca Ridge and the Endeavour Hot Vents Area and at least 12 of those species have not been found anywhere else in the world (Canada Minister for Public Works and Government Services 2001). Whereas the deep ocean floor near the Endeavour hydrothermal vent field is characterised by sparse animal abundance of about twenty worms and brittlestars per sq m, the area immediately surrounding the diffuse hydrothermal vent flows supports an abundant web of life that can range up to half a million animals per sq m (CDFO 2001b).

It is this abundance of life that has in part been responsible for the intense scientific interest in the area. Scientists using the USA submersible *Alvin* and the remotely operated vehicle *Jason* have undertaken more than a dozen missions in the area. In addition, four joint Canada–USA studies have made use of the Canadian Remotely Operated Platform for Ocean Studies (ROPOS) in the area (CDFO 2001b). On occasions this research has been highly invasive and destructive. For example, in July 1998 the American Museum of Natural History contracted the University of Washington to recover parts of several chimneys for display and specimen study. This joint project of American and Canadian scientists removed upper sections of four chimneys, parts of which are now on display in museums in the USA.

Given the intense scientific interest in the area, it is clear that the involvement of Canadian and USA scientists in the establishment and management of the MPA is crucial, a point that appears to have been recognised at all stages of the development of the MPA so far.

STAKEHOLDER INVOLVEMENT IN ESTABLISHING THE MPA

The process that has been followed in establishing MPAs under the *Oceans Act* is similar to processes elsewhere in the world (CDFO 1997a). Potential MPAs are identified, evaluated, selected, established and managed. The identification and initial screening of Areas of Interest (AOI) were performed by CDFO. Shortly after this the Endeavour field was designated as a pilot MPA under the *Oceans Act.* In 1999 a Planning team (supported by an Advisory Team) was established to study the feasibility of an MPA at the Endeavour site, to develop recommendations and an action plan and to develop and implement a consultation plan for the MPA (Canada Minister for Public Works and Government Services 2001).

Experience of MPAs to date suggests that stakeholder involvement is a key factor in their successful establishment and the implementation of their associated management plans and zoning arrangements (Gubbay 1995). A characteristic of the process leading to the establishment of the Endeavour MPA has been the consultation process, which has engaged a wide range of stakeholders such as scientists. The Planning Team comprised a range of interested stakeholders including representatives from CDFO, Natural Resources Canada, Universities, Canadian Non-Government Science, InterRidge, International Science and RIDGE. The Advisory Team included representation from CDFO and universities (Canada Minister for Public Works and Government Services 2001). Prior to concluding that the MPA was feasible, the Advisory and Planning Teams also consulted representatives of the mining and deep-sea fishing industries (CDFO 2001b) and a broad range of interested parties, including representatives from Heritage Canada, the Canadian Parks and Wilderness Society and other non government organisations (Canada Minister for Public Works and Government Services 2001).

A significant part of the process was the preparation of the AOI Evaluation. The AOI Evaluation process involved collecting and compiling an Ecosystem Overview. The Ecosystem Overview brought together information on the Endeavour Vents Area including: (1) an ecological assessment (documenting what was known about aspects of the natural environment of the proposed MPA including geology, physics, chemistry and biology of the area); (2) a technical assessment (covering factors relevant to the establishment of the MPA such as jurisdiction and enforceability); and (3) a socio-economic assessment (which explored issues arising from human activities and interests in the area such as fishing, mining and scientific research) (Institute for Pacific Ocean Science and technology 1999).

A draft of the Ecosystem Overview was subjected to scrutiny by stakeholders who participated in a workshop in March 1999. The workshop presented and gathered feedback on the Ecosystem Overview and other management issues (Canada Minister for Public Works and Government Services 2001). Participants included representatives of federal and provincial governments such as Parks Canada Agency, Natural Resources Canada, British Columbia Ministries of Energy and Mines, of Environment, Lands and Parks, and of Fisheries, and the Information, Science and Technology Agency, US National Oceanic and Atmospheric Administration, academic institutions, museums, oceanographic groups, and the mining industry (Institute for Pacific Ocean Science and Technology 1999). The most significant concern that was raised and addressed was the impact on ongoing Canadian and foreign scientific research within the MPA. Concerns were raised that restrictions on access to the MPA and a complicated bureaucratic permit process might make it difficult to attract funding for ongoing scientific research. With limited funding available for this type of scientific research, the point was made that competitors for funding would inevitably ask the question "Why should research be funded in an area where continued access is uncertain and where the Canadians may raise all kinds of obstacles to foreign scientists?" Significantly, these concerns appear to have been recognised in both the draft regulations proposed to establish the MPA and the Management Plan, which are examined below.

The issue of access to mineral resources was also raised. The mining industry had argued that before an area of Canada's territory [sic] "is alienated forever from public access an assessment of lost economic opportunities should be made" (Institute for Pacific Ocean Science and technology 1999). However, a technical and economic feasibility assessment conducted in the area in February/March 2001 by Natural Resources Canada concluded that estimates of mineral tonnage in the area were too small to be economically viable (Canada Minister for Public Works and Government Services 2001). However, that conclusion ignores the potential economic value of genetic resources associated with hydrothermal vents, a point that is examined below.

IMPLEMENTING THE OCEANS ACT ON THE DEEP-SEA FLOOR: THE PROPOSED MPA REGULATIONS

The proposed regulations are known as the *Endeavour Hydrothermal Vent Marine Protected Area Regulations*. The regulations are to be read in conjunction with the proposed Management Plan, which provides that the principal objective in establishing the MPA is to contribute toward "the protection and conservation of a representative portion of the Endeavour segment of the Juan de Fuca Ridge, its dynamic submarine ecosystems, unusual hydrothermal features, specialised biota and habitats, high biodiversity and enhanced biological productivity" (CDFO 2001b).

Under Regulation 1 the MPA, to be called the *Endeavour Hydrothermal Vents Marine Protected Area*, is defined as

> *"The area of the Pacific Ocean --- the seabed, the subsoil and the waters superjacent to the seabed – that is bounded by a line drawn from a point at 47º54′N, 129º02′W, from there west to a point at 47º54′N, 129º08′W, from there north to a point at 48º01′N, 129º02′W, and from there south to the point of beginning".*

For the purposes of the Regulations this is defined as the "Area". In all, the the Area is 93.48 km². In terms of the requirements of the *Oceans Act* the establishment of the MPA has been justified under three criteria: (1) the conservation and protection of a unique habitat in terms of Section 35(1)(c); (2) conservation and protection of a marine area of high biodiversity in terms of Section 35(1)(d); and (3)conservation and protection of a marine habitat necessary to fulfil the mandate of the Minister of Fisheries and Oceans under Section 35(1)(e). The designation as an MPA under the third criterion appears to be unnecessary, given that the Endeavour area already clearly falls within other provisions of Section 35.

Although the regulations have not yet been proclaimed, CDFO has already commenced implementing key provisions of the Regulations and the Management Plan, including aspects of the access authorisation process, the use of observers, outreach and education, and the governance structure (to be outlined in more detail below).

Draft Regulation 2 prohibits certain activities within the Area that threaten ecosystem integrity. Specifically, it provides

> *"No person shall:*
>
> a. *disturb, damage or destroy, in the Area, or remove from the Area, any part of the seabed, including a venting structure, or any part of the subsoil, or any living marine organism or any part of its habitat; or*
>
> b. *carry out any underwater activity in the Area that is likely to result in the disturbance, damage, destruction or removal of anything referred to in paragraph (a)".*

Of the activities that have been identified as threatening hydrothermal vent ecosystems, it appears that only deep-sea mining is prohibited. Deep-sea tourism appears to be unaffected provided that it does not involve any of the activities prohibited under draft Regulation 2, although to date there have been no tourist dives in the Endeavour area. Marine scientific research will be permitted to continue. This is because the prohibition on activity in the Area under draft Regulation 2 is qualified by exceptions noted in draft Regulation 3(1), which permit disturbance, damage, destruction or removal for scientific purposes. Research is permitted provided a research plan is submitted to CDFO no later than 90 days before the start of such research and provided all licences, authorisations or consents required by law have been obtained.

Draft Regulation 3(2) defines the information required in a research plan to be submitted as required by Regulation 3(1). This includes the following: details of each ship to be used in the research; details of the scientists involved in the research; proposed commencement date, duration and itinerary of the research; a summary of the proposed research to be carried out, including the data to be collected, sampling protocols to be used, techniques to be used (such as those involving explosives, radioactive labelling or remotely operated vehicles); equipment to be moored, the method of mooring, and any substances that will be discharged.

There appears to be nothing onerous in the information required. This is all information that can easily be collated and would be compiled anyway as part of the normal planning process for such research programs.

Existing procedures for issuing licences or permits for foreign scientists will be maintained. All foreign vessels wishing to conduct marine scientific research in Canadian waters are already subject to the Foreign Vessel Clearance Request Process (FVCRP) pursuant to the *Coasting Trade Act, and the Coastal Fisheries Protection Act.* Under this process marine scientific research within any area up to the edge of Canada's continental shelf is subject to approval by the Canadian Minister of Foreign Affairs. Canadian Department of Foreign Affairs and International Trade (CDFAIT) forwards requests to relevant government departments for their comment (Canada Minister for Public Works and Government Services 2001). Under existing procedures, these requests are vetted by CDFO on behalf of CDFAIT. Section 44 of the *Oceans Act* also authorises CDFO to attach a condition to a foreign ship's approval, namely that it must supply CDFO with the results of the marine scientific research. Since 2000, CDFO has been informally monitoring research activities in the area through the FVCRP process, and requests for clearance have been vetted by the Planning Team to monitor compliance with the spirit of the proposed MPA.

Research by Canadian Scientists will possibly be subject to the grant of licences under the *Fisheries Act* and the *Coastal Fisheries Protection Act.* However, the legislative authority for such licences appears unclear and further amendment to these Acts, or alternatively amended regulations under the *Oceans Act*, may subsequently be required. For the time being, CDFO relies on voluntary submission of cruise plans by Canadian researchers. The vast majority of Canadian researchers use research vessels of the Canadian Coast Guard, which is part of CDFO. Hence, the department responsible for regulating marine scientific research within the MPA also takes part in such research itself; whether there is any conflict of interest is yet to be seen. Overall, the procedures do not appear to involve any new measures, and USA scientists' concerns to avoid a complicated bureaucratic permit process appear to have been met.

MPA MANAGEMENT PLAN

The proposed Management Plan (CDFO 2001b) for the MPA divides the MPA into four zoned management areas as follows: (1) *The Main Endeavour Field* (approximately 400 m long by 150 m wide); (2) *The Mothra Field* (a vent field approximately 500 m long on the Western Wall of the Endeavour Segment); (3) *The High Rise Field* (400 m wide and 400 m long in the Axial valley of the Endeavour Segment); and (4) *The Salty Dawg Field* (several hundred sq m in the Axial Valley of the Endeavour Segment).

Different types of activity are to be permitted in each of these zones, in large part reflecting past activities. Few activities have previously taken place in the area of the *Salty Dawg* vent field and management of this area "will prioritise activities using observation-based or other less intrusive study techniques" (CDFO 2001b), leaving it as a "relatively pristine portion of the Endeavour area". Activities in the *Salty Dawg* field will be limited to (1) infrequent water sampling and annual visits to monitoring instruments in areas on or near the seafloor, (2) acoustic imaging of the field, (3) investigations of the water column that have no impact on the seafloor or benthic/near-bottom ecosystems, and (4) activities in the area that otherwise contribute to the knowledge and understanding of environmental impacts of human activities on hydrothermal vent ecosystems.

To date, the *High Rise Field* has been of only moderate interest for research activities. Its impressive relatively unspoilt natural features, combined with its more pristine environment, make it suitable for projects focussed on education. The *High Rise Field* will become a site for research associated with long-term monitoring and an important component of the education/outreach strategy of the MPA (CDFO 2001b). Most scientific research will be confined to the *Mothra* and *Main Endeavour* fields. To date, most research has focussed on these fields, including both observational and intensive sampling operations. These activities will continue to be permitted "provided they are consistent with the regulations". Presumably all this means is, that provided all authorisation procedures are adhered to, any type of scientific research, including the most invasive or destructive activities, will be permitted.

ENFORCEMENT/POLICING

It has been argued that enforcement is an essential component in the management of MPAs (Causey 1995). The policing of any MPA is often difficult (for example as a result of lack of resources such as personnel), but enforcement or policing on the deep seabed presents unique difficulties. The extremes of pressure and temperature and the total darkness mean that conventional measures such as regular patrols by fisheries officers or rangers are impossible. Nonetheless, the Management Plan does provide some indication of how these difficulties may be overcome and how activities within these zones will be policed.

Throughout the MPA, and in particular in the *High Rise* and *Salty Dawg* areas, Marine Environmental Quality protocols and indicators will be developed and implemented to prevent and minimise anthropogenic impacts (CDFO 2001a). Specific policing measures also include a requirement that before and after images of a sample site be submitted with cruise reports for activities involving sampling. Also, all submersible and dive operations (be they research, tourism or otherwise) will be required to record and document complete, continuous videotapes of the entire period on the seafloor. These tapes must be retained and may be subject to auditing by CDFO. In addition, all organisations conducting activities in the area will be required to submit cruise reports that account for all time at sea and that describe the activities and procedures undertaken (which must be submitted within two months of completion of each cruise). Finally, all vessels carrying out activities in the area will be required to reserve a berth for an observer. In 2000 and 2001 CDFO sent two observers on different vessels and is planning to send at least one observer in 2002.

EDUCATION

In keeping with the theme of allowing acess to the MPA for the pursuit of knowledge, the Management Plan also proposes the development and implementation of an education and outreach strategy CDFO 2001b). It is anticipated that this outreach strategy will be developed and implemented to focus on agencies responsible for granting funding for research in both Canada and the USA, including an emphasis on building further co-operation between researchers and funding agencies already involved in research in the Endeavour area. This, again, appears to be a recognition of the concerns raised by scientists discussed above. The Management Plan also proposes developing interest in hydrothermal vents and the MPA through the development of education modules suitable for delivery in Canadian schools and the development of educational material for delivery via a variety of media such as videotapes and the World Wide Web.

GOVERNANCE STRUCTURE

Overall management of the MPA is to be executed through a management committee chaired by CDFO. The management committee will act as adviser to CDFO, which retains legislative responsibility for the MPA. The most important role of the management committee will be to review proposed plans for research and other activities within the MPA, including making recommendations to CDFO with regard to the appropriateness of the activities and any recommended conditions imposed as part of the approval process CDFO 2001b). Reflecting the inclusive attitude to stakeholders exhibited to date, it is proposed that the management committee will be composed of a cross section of stakeholders and representatives of national government agencies. It is proposed that the management committee will be composed of representatives from CDFO-Oceans Directorate, CDFO-Science Branch, Natural Resources Canada, environmental NGOs and Canadian private sector (one member each), Canadian academic science (three members), foreign science (two members: one USA Ridge, one InterRidge), public education/ outreach (two members: one kindergarten to grade 12, one from a public awareness group).

Given the conclusion that there are no viable mineral resources within the MPA it is curious that representatives from Natural Resources Canada and the Canadian private sector will be appointed to the management committee. The management committee is also weighted heavily in favour of the interests of stakeholders from the Canadian and foreign scientific community. This contrasts with the single representative of environmental NGOs. It will be interesting to see how this mix works, particularly given the management committee's role in vetting plans for research within the MPA.

UNRESOLVED ISSUES: BIO-PROSPECTING AND ABORIGINAL TITLE RIGHTS.

Although little is known about the biodiversity of microbes at hydrothermal vents, there is mounting evidence that these organisms have a range of applications from molecular biology to the food processing, fabric and chemical industries (Butler *et al.* 2001), and the bioremediation of toxic wastes (*WWF/IUCN.* 2001). The most promising area of research relates to enzymes. Extremophiles and their enzymes (which can survive extremes of temperatures, pH and pressure) have great potential for wide commercial use (Allen 2001). Examples include microbes that can degrade crude oil and an enzyme that may provide a means of producing a hydrogen fuel source from glucose. Some specialised compounds produced by vent organisms also have potential medical applications (Dando and Juniper 2001). Many survive under highly radioactive conditions. Study of these may lead to discovery of new DNA repair mechanisms with possible applications in fighting cancer.

The full economic potential of the genetic resources of hydrothermal vents is unknown and unrealised (WWF/IUCN *et al.* 2001). But, with only a fraction of the projected thousands of hydrothermal vent sites worldwide so far identified, it is reasonable to speculate that hydrothermal vent sites promise a wealth of biotechnologically useful microorganisms (Jannasch 1995). Although the value of the genetic resources of hydrothermal vents is unknown, experience with bacteria in terrestrial environments provides some indication of their potential value. For example, the annual market for the *Taq* DNA polymerase enzyme (produced from the *Thermus aquaticus* bacteria isolated from terrestrial hot springs in Yellowstone National Park) is approximately US$500 million per year (Butler *et al.* 2001).

The potential economic value of the genetic resources appears not to have been considered in the process leading to the establishment of the MPA. There is nothing in either the draft regulations or the management plan to regulate bioprospecting. There is no obligation on bio-prospectors to share the proceeds of the commercialisation of the genetic resources of hydrothermal vents under Canadian Law.

Under International Law Canada is only required to permit marine scientific research (as distinct from bioprospecting) within its EEZ. Under *UNCLOS,* and arguably under Customary International Law, Canada can withhold its consent to bioprospecting or alternatively could make its consent conditional on the sharing of the benefits of such research. Similarly, under the provisions of the *Convention on Biological Diversity* , to which Canada is a party, bioprospectors are required to obtain the prior informed consent of Canada, as a condition of access for the investigation of genetic resources (Dando and Juniper 2001). Canada would be well within its rights to prohibit such activity or permit it on condition of benefit sharing such as through the payment of royalties, subject, of course, to its enacting enabling legislation under Canadian domestic law.

It is a difficult issue to resolve and few nations have attempted to come to terms with establishing a suitable legislative regime. However, given the potential economic value of these resources it is surprising that this issue appears not to have been canvassed. The conclusion reached at an early stage of the planning process that there was no economic interest at stake seems to have been somewhat premature.

ABORIGINAL TITLE

A second issue that appears not to have been considered in any great detail is the potential for aboriginal title rights to exist in relation to the seabed. Aboriginal title has been given constitutional recognition in Canada pursuant to Section 35(1) of Canada's *Constitution Act*. Ginn (2002) has suggested that Section 35 "appears to recognize that if a provision of the *Oceans Act* were to conflict with a judicially recognized aboriginal right, it would be possible for the right to take precedence over the Act. Potentially, this could mean that certain aspects of the ocean policy as articulated in the Act might not be applicable to or enforceable against a particular First Nation."

Given the location and depth of the proposed MPA, it is unlikely that aboriginal rights such as fishing rights will be affected. But what about aboriginal title rights to the area surrounding the MPA and in particular the seabed? The answer to this question depends ultimately on what aboriginal title rights First Nations hold in relation to the seas and the seabed. There has been no definitive ruling on this point in Canada. But this may not be far away. A summons has recently been filed by the Haida Nation that claims aboriginal title to an area of "land, inland waters, seabed and sea" described as "Haida

Gwaii". The case will inevitably reach the Canadian Supreme Court. While the decisions of Australian and other common law jurisdictions which have considered this issue, such as the *Commonwealth v Yarmirr; Yarmirr v Northern Territory; Western Australia v Ward; Attorney-General (NT) v Ward; Ningarmara v Northern Territory*; may be persuasive for the Canadian Supreme Court, it is not clear what the final outcome will be. Differences in the way aboriginal title is characterised under Canadian law and, in particular, the Constitutional recognition of aboriginal title in Canada, mean, recognition of title to the seabed is possible.

What would be the consequence of recognition of aboriginal title rights in the seabed? Would First Nations be required to comply with provisions of the *Oceans Act*? What of MPAs established under the authority of that legislation? What would be the implications for bio-prospecting? Do aboriginal title rights also extend to genetic resources? Would research and exploitation of the genetic resources of the seabed require First Nations consent? To what extent would First Nations be entitled to share in profits to be gained from exploitation of such resources? Given what appear to be significant economic interests at stake (particularly with respect to genetic resources), has the limited consultation with First Nations been adequate? All these issues must await a decision on the question of aboriginal title to the seabed under Canadian law. But it is surprising that such important issues appear not to have been considered by CDFO in any great detail in developing this and other MPAs under the *Oceans Act*.

LUCKY STRIKE AND OSPAR

Following the *Sintra Statement*, parties to the *OSPAR Convention* are committed to promoting "the establishment of a network of marine protected areas to ensure the sustainable use and protection and conservation of marine biological diversity and ecosystems" (WWF 2000) Considerable work is now being carried out by parties to the *OSPAR Convention*, and other interested parties such as WWF, to design mechanisms to implement these obligations, including developing an overall framework for MPAs within the context of the *OSPAR Convention*. In part, this mirrors what is already being done within the context of implementing the European Communities *Habitat Directive 92/43/EEC.*

WWF has suggested that *Lucky Strike* should be proposed as an MPA to *OSPAR* by Portugal. *Lucky Strike* is a hydrothermal vent area south-west of the Azores, within Portugal's EEZ, first discovered in 1993. The extensive scientific research at this site is largely unregulated. Geological and biological sampling poses a very real threat to its ecosystem. There is therefore an urgent need for some form of regulation of activities in its vicinity.

The process adopted in establishing the Endeavour MPA re-confirms what was already known about the importance of stakeholder involvement in the establishment of MPAs. Any MPA regime for *Lucky Strike* must involve key stakeholders such as scientists in its design and ongoing management. Similarly, given the history of scientific research in and around *Lucky Strike*, the MPA will need to permit scientific research to continue in one form or another. A zoning scheme permitting research in some areas and not in others, as in the case of the Endeavour MPA, might also be appropriate for *Lucky Strike*. Given the difficulties of enforcing compliance in the deep sea, the use of measures such as before-and-after imaging, continuous videotaping of submersible and other activities within the MPA, on-board observers and cruise reports all appear to be suitable tools in implementing any management plan to be developed for *Lucky Strike*. There also appears to be no reason why similar management regimes and tools to assist with ensuring compliance might not also be suitable for the conservation of other deep-sea habitats.

CONCLUSION

Measures for conservation of deep-sea habitats are long overdue. Although scientific research is a major threat to the continued integrity of hydrothermal vent ecosystems, at least at the Endeavour site it is now a managed threat. Previously unobserved and unregulated invasive, and at times destructive, scientific research is now subject to some scrutiny and is to some extent controlled and confined to specific areas. Adjacent sites will, to some extent, be preserved in their pristine condition. The MPA attempts to strike a balance between conservation of deep-sea biodiversity and continuation of scientific research. The high level of stakeholder involvement in its establishment and ongoing management points the way for future efforts at conservation at other hydrothermal vent sites and for other deep-sea habitats in general.

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BALANCING NATIVE FISH DIVERSITY, EXOTIC FISH IMPACTS, AND ANGLING IN NEW ZEALAND, NORTH ISLAND DUNE LAKES

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Abstract

The galaxiid fish *Galaxias gracilis* now occurs in 11 lakes on the west coast of the North Island of New Zealand. It was once abundant but is now rare in 5 of these lakes and is extinct in at least 2 others. Its conservation is therefore a priority. Studies of its ecology and life history indicated that juveniles are pelagic and planktivorous, whereas adult fish depend on larger invertebrates for food, so feed in the littoral zone. A six-year experiment was carried out in two lakes where this species is rare to determine whether predation by rainbow trout (*Oncorhynchus mykiss*) and/or impacts by mosquitofish (*Gambusia affinis*) were responsible for its population decline. Angling organisations halted trout stocking in the experimental lake, but not in a reference lake, and remaining trout were then removed from the experimental lake. Trout removal increased the abundance of juvenile galaxiids in the experimental lake relative to the reference lake, but adult galaxiids remained rare. The reason for this was the increase in *Gambusia* over summer and autumn in the experimental lake. *Gambusia* was observed attacking and killing large numbers of adult galaxiids as these fish attempted to feed in the littoral zone. As the increase in *Gambusia* was thought to be caused by trout removal, and was a greater threat to the galaxiids than trout predation, trout stocking was resumed to restore the balance between native fish, exotic fish, and trout angling. This experiment demonstrates the complexity involved in managing fish diversity in protected areas.

Keywords: galaxiids, *Gambusia*, trout, lakes, fish interactions, protected areas

AQUATIC PROTECTED AREAS AND MULTI-USE LAKES

The Kai Iwi Lakes Recreation Reserve comprises three lakes that, among other native and introduced fish, contain a rare and threatened species of native fish called the dwarf inanga (*Galaxias gracilis*). International treaties, to which New Zealand is now a party, require conservation of biodiversity, and this applies particularly to endemic, lacustrine species that are rare and endangered, such as the dwarf inanga.

Although 'faunistic reserves' or 'aquatic protected areas' can be formed to maintain biodiversity in lakes and to protect rare species in National Parks, rare and threatened species are not always restricted to such areas. In the case of the Kai Iwi lakes, creation of a faunistic reserve would not be realistic because of the historical use and the importance of the lakes for recreation, angling and tourism. Therefore, other means are required to conserve dwarf inanga in such environments. In this paper, I outline studies to identify the management requirements for conserving dwarf inanga in the Kai Iwi lakes. The results illustrate the difficulty of managing fish interactions in lakes, and the complex management structures that exist and that provide barriers to the integrated management needed to maintain biodiversity in lakes, while balancing this with existing uses.

THE KAI IWI LAKES RECREATION RESERVE

The Kai Iwi Lakes Recreation Reserve is on the west coast of the North Island of New Zealand and comprises 3 lakes: Lake Waikere; Lake Taharoa; and Lake Kai Iwi (Fig. 1). These lakes were formed in sand dunes some 10,000 years ago (Lowe and Green 1987) and are still relatively pristine and unmodified environments (Table 1). They are therefore popular recreational resources. Their water quality is relatively high compared with many other lakes in this Northland region of New Zealand, and they are comparatively deep (Table 1). There is no outlet to the sea, and there are no permanent tributary streams in the catchments, so water supply is mainly from direct rainfall on the lake surface and underground soakage.

Lake Taharoa is the largest lake in the series (Table 1) and is characterised by azure-blue waters and large sandy beaches. Small areas of sandstone are present on the northern and southern shores, and patches of rush beds (mainly *Baumea* sp. in the shallows and *Eleocharis acuta* in deeper water) occur at places around the lake edge. These patches are scarce and collectively occupy less than 10% of the lake edge.

Table 1. Main physical characteristics of the three lakes in the Kai Iwi Lakes Recreation Reserve.

	Lake Taharoa	**Lake Waikere**	**Lake Kai Iwi**
Altitude (m)	70	79	70
Area (km^2)	2.37	0.35	0.31
Max. depth (m)	35	29	14
Secchi disc (m)	10	6	8
Max temperature (ºC)	23	24	24

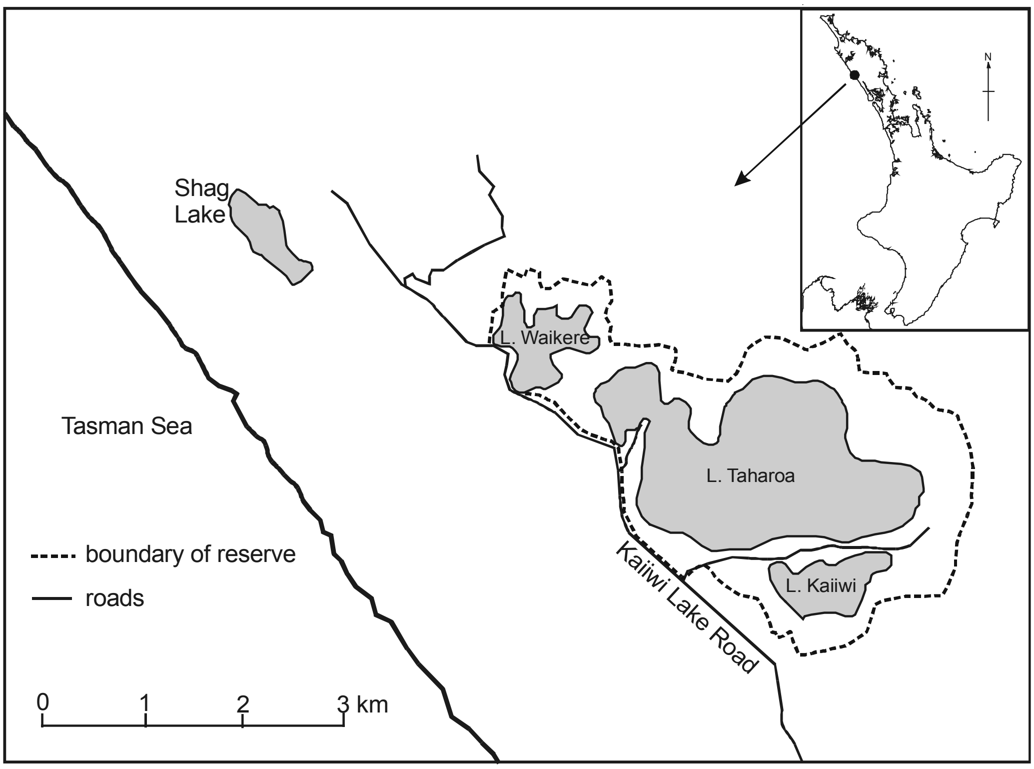

Fig. 1. Location of the Kai Iwi Lakes Recreation Reserve and the three lakes (Taharoa, Waikere, Kai Iwi) in the Reserve.

Lake Waikere and Lake Kai Iwi are similar in size but differ in that no boating is allowed on Lake Kai Iwi, whereas Lake Waikere is popular for water skiing and is a venue for national competitions. The shoreline of Lake Waikere contains a number of sandy beaches as well as sandstone outcrops. Relatively long stretches of shoreline are occupied by rush beds, but collectively these would occupy less than 25% of its length. In comparison, rush beds dominate the shoreline of Lake Kai Iwi, occupying more than 80% of its length. Lake Kai Iwi is at times connected to Lake Taharoa by a small drain; however, this connection occurs only when lake levels are high and provided the culvert has been recently cleared of vegetation and debris.

Exotic macrophytes have invaded a number of Northland dune lakes but are not present in the Kai Iwi lakes (Tanner *et al.* 1986). However, as with most lakes in New Zealand, the native fish fauna has been supplemented by the addition of exotic species. The native fish species comprise the common bully (*Gobiomorphus cotidianus*), dwarf inanga, and two species of eel (*Anguilla dieffenbacchi*, *A. australis*). The eels cannot be considered true inhabitants, because there is no access between the lakes and the sea. Most, if not all, the eels present are therefore stocked into the lake by commercial fishers who return some 10–20 years later and harvest them. Rainbow trout (*Oncorhynchus mykiss*) were introduced into the lakes in 1968 by the Northland Acclimatisation Society (now the Northland Fish and Game Council) to see whether a trout fishery could be established. This proved successful, but as the trout cannot breed in the lakes, annual stocking is required to maintain the trout populations.

Mosquitofish (*Gambusia affinis*), henceforth called 'gambusia' to remove association with the control of mosquito larvae, was unofficially introduced into the lake around 1971.

DWARF INANGA (*GALAXIAS GRACILIS* MCDOWALL, 1967)

The dwarf inanga was first described by McDowall (1967) who distinguished it from the riverine (catadromous) and landlocked forms of inanga (*Galaxias maculatus*), termed 'jollytail' in Australia and 'puyen' in Argentina. Although derived from landlocked inanga, the dwarf inanga was classed as a species mainly because of distinct morphometric and meristic differences. Dwarf inanga are smaller than landlocked inanga and have a larger eye and optic lobe. They have fewer vertebrae and more gill rakers, the latter being longer and more elongate. These adaptations indicate a tendency towards limnetic planktivory. Dwarf inanga mature at a much smaller size (30–40 mm) and as there are no streams (even during heavy rainfall) in many of the lakes where this species occurs, they spawn within the lake. The main spawning habitat for the riverine form of inanga is inundated river or stream banks. Flooded tributary streams entering lakes are also utilised by landlocked *G. maculatus* (Pollard 1971; Frankenberg 1969), but dwarf inanga no longer require this habitat. Taken together, these adaptations indicate divergent evolution away from the riverine ancestral form and towards a fully lacustrine form.

Between 1991 and 1993, studies were carried out to determine the conservation status of dwarf inanga. These revealed that this species was present in 11 dune lakes, including the 3 Kai Iwi lakes, all of which lie within an 80 km stretch of coastline on the north-west coast of the North Island (Rowe and Chisnall 1997a). Comparative data on the abundance of adult fish (> 40 mm long) revealed that the dwarf inanga were only abundant in 2 lakes, were common in 2 others, but were rare in 5 lakes, and had become extinct in another 2 (Rowe and Chisnall 1997a). Dwarf inanga were rare in all 3 of the more northern Kai Iwi lakes, but were less threatened in the Poutu lakes that lie further south. As a consequence of these studies, the dwarf inanga was described as a 'threatened' fish of the world (McDowall and Rowe 1996), and in 1996 it was added to the IUCN Red List as a vulnerable species.

Later taxonomic studies of the mitochondrial DNA of dwarf inanga populations indicated that the Kai Iwi lakes populations were genetically distinct from the Poutu lakes populations (Ling *et al.* 2001). It is apparent that the Kai Iwi populations were not derived from the Poutu populations through stocking by pre-European Maori. The Kai Iwi lakes populations were also further removed from the ancestral riverine inanga than the Poutu lakes populations and are therefore likely to be older. The Kai Iwi lakes populations are therefore an evolutionarily significant unit (ESU) requiring even greater care than other populations.

In addition to surveys to assess its conservation status, the basic biology and ecology of dwarf inanga was described. Studies of the distribution and feeding of the various size/age groups present within lakes indicated that the larvae (9–25 mm long) were planktonic and present in the water column of lakes down to at least 10 m in spring (Rowe and Chisnall 1996). Once these larvae were longer than about 25 mm, they became pigmented, and the juvenile fish formed schools at depths of 0–5 m in the limnetic zone. Here they fed exclusively on zooplankton. At a length of about 40 mm (at which size they were capable of breeding and were therefore adults), dwarf inanga moved to the shallows in the littoral zone, and larger, littoral invertebrates dominated their prey. During the day, the maximum size of these adult fish in the littoral zone was around 60 mm. However, at night, larger fish (60–80 mm) occurred in this zone. These larger adults schooled in the deeper waters (10–15 m deep) of the hypolimnion by day, returning to the littoral zone to feed at night (Rowe and Chisnall 1996). This behaviour was attributed to the daytime risk of predation by visual predators such as shags (*Phalacrocorax* spp.). Thus, the juveniles were planktivores but the adults depended on littoral foods, with small adults feeding in the littoral zone by day, and the larger adults foraging there only at night.

Additional studies to identify the potential environmental factors responsible for the decline of dwarf inanga indicated that eel stocking, trout predation, some water chemistry variables, and the invasion and modification of lake ecosystems by exotic plants and fish were all potentially involved, with the main factors varying among the lakes (Rowe and Chisnall 1997b). The rarity of dwarf inanga in the three Kai Iwi lakes was attributed mainly to predation by the stocked rainbow trout. However, in 1986, dwarf inanga were stocked into a dune lake outside their natural geographical range to provide a forage fish for the stocked trout population (Thompson 1989). This stocking succeeded and dwarf inanga quickly became abundant in the lake, despite on-going predation by the trout. The role of trout predation in the Kai Iwi lakes was therefore questioned and the potential impact of gambusia on dwarf inanga raised (Rowe and Chisnall 1997b). Gambusia are omnivores and occupy the littoral zone of lakes, on which adult dwarf inanga also depend for feeding. Gambusia also prey on the eggs and larval stages of other fish and have been implicated in the decline of a number of indigenous fish species in both the USA and Australia (Courtenay and Meffe 1989; Arthington and Lloyd 1989). It was therefore possible that they had compounded the effect of trout predation on dwarf inanga.

This uncertainty over the respective role of trout and gambusia resulted in a Department of Conservation (DOC) funded study to determine the cause(s) of the decline in dwarf inanga so that management actions to restore populations could be recommended. In the Kai Iwi lakes, this took the form of a long-term (6-year) trial to determine the effect of trout removal on dwarf inanga. The basic design for this trial was to remove trout from one of the Kai Iwi lakes leaving the other as a control lake. The abundance of dwarf inanga would then be monitored in both lakes and there was an expectation that dwarf inanga would quickly recover in the troutless lake thereby confirming the overriding effect of trout predation.

EFFECTS OF TROUT REMOVAL ON DWARF INANGA

The Northland Fish and Game Council agreed to suspend trout stocking in Lake Waikere (experimental lake) over the trial period and to continue stocking in Lake Taharoa (control lake). The remaining trout were removed from Lake Waikere over the summer of 1993/1994 by gill netting. Finclips given to each year class of trout provided data on the relative abundance of each, and on the maximum longevity of trout in the lake, which was 4 years. In September 1994, several juvenile trout were caught in the lake, indicating that an un-authorised stocking had occurred. Regular gill netting was therefore continued until 1998. No further trout stocking occurred and more than 92% of the trout were removed by September 1995.

Dwarf inanga were monitored in both Lake Waikere and Lake Taharoa twice a year (spring and autumn) from 1993 to 1997. The abundance of schooling fish was assessed by high-frequency (200 kHz) echosounding, which readily detects the presence of schools of fish in the limnetic zone of lakes (Rowe and Chisnall 1996). Transect lines were established down the main axis of each lake and echosounding was carried out each spring. The echograms were inspected and the number of schools of fish present was visually assessed on a 4-point scale (none, rare, common, abundant). In addition, mini-fyke nets (length 2 m, mesh size ~5 mm) were used to measure the relative abundance or catch per unit effort (CPUE) of adult dwarf inanga in the littoral zone. Two nets were set in each of the four quadrants of the lakes overnight, and catch rates for each species of fish present (including crayfish) determined the following morning. Mean CPUE was calculated for each lake and compared with the scale of abundance developed in other lakes to determine the conservation status of this species. In the other lakes, a mean CPUE of <10 was deemed to reflect a population that was rare, whereas a mean CPUE of 10–20 indicated that dwarf inanga were common, and a catch rate of >20 that they were abundant (Rowe and Chisnall 1997a).

No schools of limnetic fish were apparent in the echograms from either Lake Waikere (Fig. 2) or Lake Taharoa in spring 1993 or spring 1994 when trout were still present and before dwarf inanga populations could increase through natural reproduction. However, a large number of echoes were apparent in both 1995 and 1996 in Lake Waikere and not in Taharoa. The presence of these schools indicated good recruitment of juvenile dwarf inanga in the absence of trout. A further increase was expected in 1997, but this failed to materialize (Fig. 2). Removal of trout therefore temporarily increased the recruitment of juvenile dwarf inanga. However, it did not increase the abundance of adults.

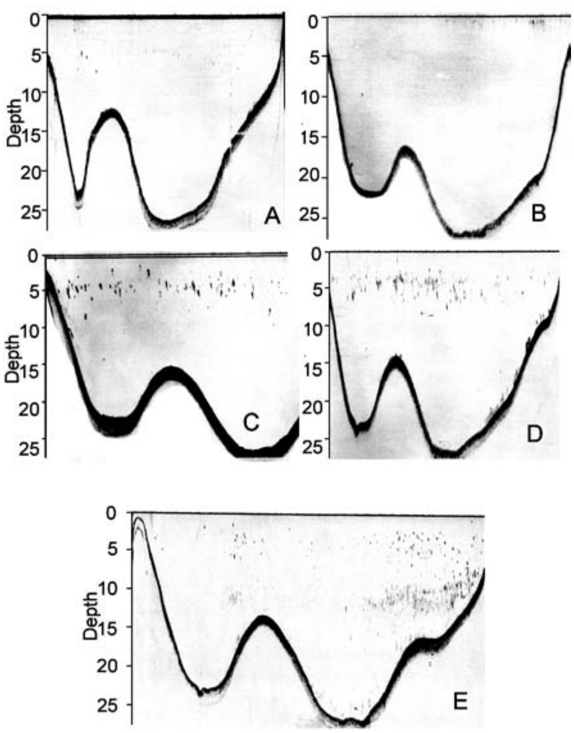

Fig. 2. Changes in the relative abundance of schooling fish (dwarf inanga) in Lake Waikere between 1993 and 1997 as determined by high-frequency echosounding. Echograms are for (*A*) 1993, (*B*) 1994, (*C*) 1995, (*D*) 1996, (*E*) 1997.

The mean CPUE for adult dwarf inanga in the littoral zone of Lake Waikere remained below 1.5 between 1993 and 1998 (Fig. 3A), and it was no different to that in the control lake where trout stocking had continued (ANOVA, $P > 0.05$). The maximum mean CPUE in Lake Waikere of 1.5 is well below the value of 10.0 at which dwarf inanga are common. Therefore, removal of trout did not result in the recovery of dwarf inanga in Lake Waikere, even though it was apparent that the survival of juvenile fish was relatively high in 1995 and 1996. We concluded that trout predation was not the main factor limiting the abundance of dwarf inanga in this lake, and that some other factor must be responsible.

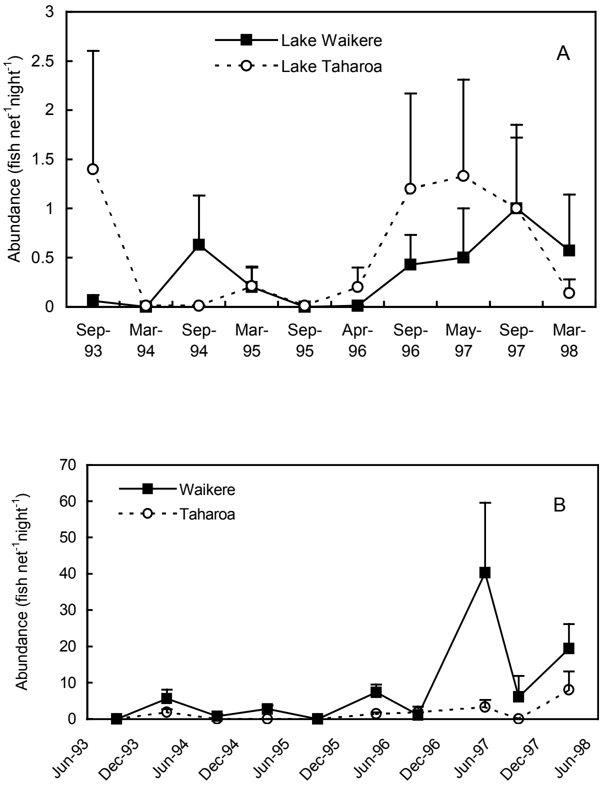

Fig. 3. Relative abundance (mean CPUE ± SE) of (*A*) adult dwarf inanga and (*B*) gambusia in the littoral zone of lakes Waikere and Taharoa between 1993 and 1998.

The only other change to this lake ecosystem was the introduction of gambusia (Rowe and Chisnall 1997b). Gambusia were introduced to the Kai Iwi lakes three years after the introduction of trout. Their introduction had therefore coincided with the decline of dwarf inanga that was attributed to trout predation. Gambusia have been associated with the decline of native fish in other lakes (e.g. Myers 1965; Meffe 1985; Minkley *et al.* 1991; Howe et al 1997), so it seemed reasonable that gambusia could be partly responsible for the decline of dwarf inanga and also responsible for their failure to recover when trout were absent.

THE ROLE OF GAMBUSIA IN THE DECLINE OF DWARF INANGA

Confirmation that gambusia were primarily responsible for the rarity of dwarf inanga in the Kai Iwi lakes was provided by a large mortality of dwarf inanga in Lake Waikere in March 1998. We observed several hundred dead and dying fish around the outer edges of the rush beds in Lake Waikere (but not on the beaches), and collected 160 for examination. The dead dwarf inanga ranged in size from 27 to 69 mm with most being more than 35 mm. Of the 149 that were relatively intact and thus identifiable, 89% exhibited physical damage to either the fins or eyes. The most common injury was complete removal of the tail fin (52% of fish), which would have effectively immobilized the fish, preventing it from feeding. Adult dwarf inanga collected from sandy beaches and then released into the rush beds were quickly attacked by groups of gambusia that soon immobilized them. The gambusia did not feed on the crippled dwarf inanga, and most would have died from either starvation or secondary infections to their fins. The presence of dead and dying dwarf inanga around the outside edge of the rush beds of the littoral zone indicated that they were attacked as they attempted to enter this gambusia-dominated habitat. In other lakes, common bullies occur on the substratum of the littoral zone in and among rush beds. However, none were seen in this habitat in Lake Waikere in March 1988. It was likely that gambusia were excluding common bullies as well as dwarf inanga from this habitat, and that the dependence of adult dwarf inanga on littoral foods resulted in large numbers being killed when gambusia densities were high in such habitat.

The mean CPUE data for gambusia indicated that they were more abundant in Lake Waikere in March 1997 and 1998 than in previous years (Fig. 3B). They were also more abundant in Lake Waikere than in Lake Taharoa in both 1997 and 1998. Their increase in Lake Waikere therefore coincided with the virtual absence of trout in Lake Waikere.

We observed that, in general, gambusia were most abundant in the rush beds of lakes Waikere and Taharoa in summer/autumn and were rarely seen on open sandy beaches in these lakes. However, a notable exception occurred in Lake Kai Iwi where gambusia were highly abundant (> 50–100 fish m^{-2}) over the relatively small sandy beaches as well as in the rush beds. Such differences in habitat use imply that predators such as trout restrict gambusia to rush-bed habitats, which provide cover from predation. In lakes Taharoa and Waikere such habitat is minimal (<25% of shoreline), so gambusia populations are relatively low when trout are present. However, in Lake Kai Iwi, such habitat is extensive (> 80% of shoreline) so gambusia can be expected to be relatively abundant whether trout are present or not. Accordingly, we reasoned that gambusia densities would be high in Lake Kai Iwi and that dwarf inanga could be extinct.

Fyke-net surveys of fish abundance were carried out in Lake Kai Iwi in 1993 and again in 1998. The mean CPUE of gambusia exceeded the highest value recorded in Lake Waikere in both years, and no dwarf inanga were found in either of these, or in another previous survey. Therefore it seems likely that a high and sustained autumnal abundance of gambusia in Lake Kai Iwi will have resulted in the extinction of its dwarf inanga population.

These data collectively indicated that gambusia pose a greater risk to dwarf inanga than trout predation, and that this risk is increased by the extent of littoral vegetation such as rush beds. However, this risk is reduced if trout, which are likely to restrict gambusia to rush-bed habitats, are present.

MANAGEMENT IMPLICATIONS

Because of the risk to dwarf inanga posed by the increase in gambusia, trout stocking was resumed in Lake Waikere at a low density to maintain the balance between gambusia and dwarf inanga. This policy will not increase the population of dwarf inanga, nor lead to its restoration, but it will prevent a further decline.

It is clear that control methods for gambusia that don't affect dwarf inanga are needed to increase dwarf inanga abundance in the Kai Iwi lakes. Until these are developed, management can only focus on other conservation methods such as artificially increasing dwarf inanga recruitment, or establishing new populations in other lakes. Further field trials to find an optimal trout stocking density which maximizes the recruitment of juvenile dwarf inanga while keeping gambusia in check are possible, but would take many years to complete.

McDowall (1984) outlined a number of criteria for selecting New Zealand lakes and rivers as faunistic reserves (or aquatic protected areas) based on the habitat requirements and characteristics of the indigenous fish fauna. Among these were the requirements that such areas should be free of exotic fish and that the indigenous species should not be exploited. The Kai Iwi Lakes Recreation Reserve does not meet these criteria and the lakes within it are subject to a wide range of uses by the public. However, active management is needed to preserve the dwarf inanga populations in these lakes. At present, their low population size means that they are highly vulnerable to extinction (Nyman 1991). Conservation will therefore require the co-operation of the Northland Fish & Game Council over trout-stocking policies for the lakes. Monitoring and management of rush beds to ensure that they do not spread is also required, but it is not clear whether this is the responsibility of DOC or the Kaipara District Council. In any event, such management actions would require a resource consent from the Northland Regional Council. A further management issue is that these lakes are now subject to a claim under the Treaty of Waitangi, with local iwi moving to establish Lake Kai Iwi as an exclusive eel fishery.

The DOC are preparing a management plan for the dwarf inanga populations in the Kai Iwi lakes, as well as for the populations in the remaining lakes where this species occurs. Control of exotic fish, augmentation of wild stocks, habitat manipulation, and translocation have all been considered but, at present, translocation appears to be the only viable option for the Kai Iwi lakes stocks. This was also considered the best and only option for conserving the Irish pollan (*Coregonus autumnalis*) (Harrod *et al.* 2001). However, translocation will not preserve the genetic integrity of the Kai Iwi stocks and it pre-supposes that other suitable lakes for dwarf inanga exist in Northland. In the long term, the development of control methods for gambusia will be required.

Management of lakes to protect a threatened species and to maintain biodiversity requires an holistic, ecosystem-based approach and this cuts across conventional management and planning boundaries for the Kai Iwi Lakes Recreation Reserve. The need to protect and sustain biodiversity in lakes is therefore generating a new management approach to the protection of aquatic ecosystems in freshwaters. The creation of Aquatic Protected Areas as faunistic reserves is a useful approach to maintaining biodiversity in some areas. However, in multiple-use environments such as the Kai Iwi Lakes Recreation Reserve, a more sophisticated system of co-ordinated planning and management is required to balance the maintenance of biodiversity with existing public uses. As the other uses of this reserve will ultimately depend on the maintenance of the integrity of the ecosystem, this management challenge will clearly need to be met.

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CREATING A NATIONAL SYSTEM OF MARINE PROTECTED AREAS – A CONSERVATION PERSPECTIVE

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Abstract

The Australian Marine Conservation Society (AMCS) has been working since 1965 with government agencies and the community to develop Marine Protected Areas (MPAs) around the Australian coastline. Although some progress has been made and Australia is often described as a world leader in marine conservation, it is still a long way from having a national system of marine protected areas that is comprehensive, adequate or representative.

A 'network' of 'no-take' marine sanctuaries is needed within a national system of multiple-use marine protected areas. These 'no-take' areas must be of sufficient size to maintain biological populations and close enough to reflect ecosystem linkages and connectivity to the surrounding system. The AMCS's 'vision' for a national system of MPAs that can benefit all Australians and the nation's outstanding marine environment is outlined.

Keywords: marine sanctuary, no-take, network, Australia

INTRODUCTION

The oceans are considered to be in serious trouble. Widespread coral bleaching, declining water quality, the presence of introduced marine pests, struggling fisheries and species declines are symptoms of a system under stress. Establishing a network of 'no-take' marine sanctuaries as part of a national system of marine protected areas will help reduce this stress and provide a powerful tool for the conservation and management of marine biodiversity.

HISTORY OF MARINE PROTECTED AREAS IN AUSTRALIA

The use of MPAs in Australia dates back to 1938 when the first example of a marine park was declared around Green Island. More than three decades later (1975), this Park was superseded by arguably the world's best-known marine park, the Great Barrier Reef Marine Park (GBRMP).

It was not until 1985, however, when Australia's waters were first divided into 32 marine bioregions that recognition was given to the concept of a representative system of marine protected areas. At the time, this bioregionalisation was seen as a key step in the process leading to the development of a national system of MPAs that would protect a full range of habitats and ecological processes in Australia's marine environment. Five years later during a general assembly of the World Conservation Union (IUCN) in Perth, the then Prime Minister, Bob Hawke, committed the Australian Government to a National Representative System of Marine Protected Areas (NRSMPA).

Another nine years passed before the *Strategic Plan of Action for the NRSMPA* (ANZECC 1999) was finalised and endorsed by federal and State/Territory Governments. The primary goal of the NRSMPA is

> 'to establish and manage a Comprehensive, Adequate and Representative (CAR) 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.'

It was further recommended that each MPA

- Provide a level of protection higher than that achieved in surrounding waters,
- Incorporate areas ranging from highly protected areas (no-take) to sustainable multiple-use areas that accommodate a wide spectrum of human activities, and
- include some highly protected areas in each bioregion.

Although the Australian Marine Conservation Society (AMCS) does not dispute the value of multiple-use MPAs, it is the 'no-take' component of the NRSMPA where the biodiversity and fishery benefits are maximised.

Benefits of 'no-take' marine sanctuaries

The potential benefits of 'no-take' marine sanctuaries can broadly include benefits to biodiversity, science, commercial and recreational fishing and non-extractive tourism.

Over the past 10 years there has been increasing evidence to indicate that when fishing is excluded from areas there is an increase in the diversity, abundance and productivity of marine organisms. Further, by providing refuges where species can grow to maturity and breed, marine sanctuaries can result in increases in the abundance and size of some species in surrounding areas.

There is of course, more to a NRSMPA than an ad hoc arrangement of sanctuaries that represent samples of Australia's marine ecosystems. It is essential that the NRSMPA establish a network of marine sanctuaries that reflect the ecosystem linkages and connectivity of the surrounding region.

Why a network of marine sanctuaries?

Networks recognise that there are many inter-relationships between marine ecosystems. The larvae and spores of many marine species can travel vast distances. Migratory species travel for thousands of kilometres using a range of habitats for their needs. Food derived from one area can provide food for organisms hundreds or even thousands of kilometres away. To protect this high level of connectivity the marine environment would benefit most from a network of sanctuaries.

Thus, the potential benefits for marine biodiversity are far greater if we establish a network of 'no-take' marine sanctuaries. Roberts and Hawkins (2000) make the following comparisons between single sanctuaries and networks of sanctuaries (referred to as reserves):

- Isolated reserves have many benefits but will only be able to protect a limited fraction of marine biodiversity;
- Large numbers of marine species have open-water dispersal phases and can potentially be transported long distances from where they were spawned;
- Individual reserves may be able to sustain self-recruiting populations of species that disperse short distances, but networks will be necessary to protect many of the species that disperse long distances; and
- Reserves in networks need to be close enough for protected populations to interact through dispersal.

Designing an effective network of marine sanctuaries

The optimal design of a network depends on the attributes of the region to be protected. Palumbi (2001) states that the minimum set of network features to be considered comprises

- the span of the network,
- the size of the individual reserves,
- the number of the individual reserves, and
- the placement of the individual reserves.

The above features combined with the ecosystem attributes of each habitat, including its status and vulnerability to disturbance, should ultimately determine the amount of area protected and the level of connectivity. The effectiveness of the NRSMPA will therefore be determined by its ability to consider these factors. The implications are significant, and designing a network of 'no-take' marine sanctuaries within a system of large multiple-use MPAs that span 16 million square kilometres of oceans and almost 60 000 km of coastline is a significant and daunting prospect for any government.

The Great Barrier Reef Marine Park Authority (GBRMPA) has recognised the need to establish a network of marine sanctuaries throughout the entire marine park to ensure the protection of biodiversity.

Creating a network of marine sanctuaries for the Great Barrier Reef

The GBRMP is a multiple-use marine park on the east coast of Australia. It covers 347,000 km^2 and stretches 2000 km along the coast of Queensland. It was established in 1975 in response to public outcry over proposed oil drilling.

Although the accompanying GBRMP Act prohibited drilling and mining for minerals in all areas of the marine park, other extractive uses such as fishing and collecting have continued over most of the region. Recently, GBRMPA acknowledged that the current level of protection was insufficient to achieve long-term protection of the Park's biodiversity and is now in the process of creating a representative network of marine sanctuaries. This initiative, known as the Representative Area Program (RAP) aims to help:

- to maintain biological diversity,:
- allow species to evolve and function undisturbed;
- provide an ecological safety margin against human-induced errors,

- provide a solid ecological base from which threatened species or habitats can recover or repair themselves, and
- maintain ecological process and systems.

At the core of the RAP is the creation of a network of marine sanctuaries in a single step. This will involve a review of the existing zoning throughout the Marine Park. This is an important component of the Program because it provides the best opportunity for scientists and managers to create a network that reflects the connectivity of the surrounding region.

The development of the network is guided by a set of biophysical operational principles prepared by an independent Scientific Steering Committee. In essence, the principles guide decisions regarding the number, size and location of no-take marine sanctuaries and include, as far as possible

- having candidate areas (CAs) whose minimum size is 20 km along the smallest dimension,
- having larger (rather than smaller) CAs,
- having sufficient replication,
- including only whole reefs within a CA (i.e. no split zones),
- having at least three reefs and at least 20% of reef area and reef perimeter per reef bioregion,
- having at least 20% of area per non-reef bioregion, except coastal bioregions which contain finer-scale patterns of diversity due to bays, adjacent terrestrial habitat, rivers,
- including a minimum amount of each type of community and physical environment in the overall network,
- maximising use of environmental information (e.g. currents and connectivity) to determine best configuration of CAs,
- including biophysically special/unique places including significant habitats,
- considering sea and adjacent land uses in determining CAs, and
- capturing cross-shelf and latitudinal diversity.

The above principles are then entered into a mathematical program which generates a range of zoning options from which a network of sanctuaries can be selected. These options help to both maximise and minimise the impact on the socio-economic environment while still achieving the biodiversity goals. If these principles are implemented in full, the Committee expects that around 25–30% of the GBRMP will be protected in marine sanctuaries (green zones). AMCS believes that by declaring the decision rules openly there is a greater degree of certainty in both process and outcome for all stakeholders. Although the level of prescription in each principle may not be directly transferable to other regions, many of the principles are internationally recognised and can be applied across most marine ecosystems.

FROM GBR TO NRSMPA

In the same way that the GBRMPA has recognised the need to establish a network of 'no-take' marine sanctuaries across the Great Barrier Reef, the NRSMPA must do the same at a national level. Future multiple-use marine parks that form part of the NRSMPA must therefore be large enough to accommodate marine sanctuaries that are of sufficient size to maintain biological populations and close enough to reflect ecosystem linkages and connectivity to the surrounding system.

In designing the NRSMPA it is therefore appropriate that a set of 'operating principles' are developed which are similar to those applied to the GBR RAP. This will provide the necessary rules on which to base the selection and identification of candidate networks of both multiple-use and 'no-take' marine sanctuaries.

The identification and selection of representative MPAs is an expected outcome of the South East Regional Marine Plan (SERMP) under Australia's Oceans Policy. The SERMP process offers the opportunity to apply the RAP approach outside the GBR. AMCS emphasises that the establishment of a network of MPAs within the south-east region should occur at one time and not be rolled out as a series of individual declarations.

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ISSUES, ESPECIALLY MARINE PROTECTED AREAS, THAT AFFECT THE AUSTRALIAN FISHING INDUSTRY

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Keywords: Australian fishing industry, marine protected areas, participation, stakeholder cooperation

INTRODUCTION

The Australian Seafood Industry Council appreciates this opportunity to participate in the World Congress on Aquatic Protected Areas.

I have been invited to offer some thoughts on issues affecting the fishing industry, and especially on "marine protected areas (MPAs) as they affect the fishing industry". I shall talk about marine protected areas in particular but shall also introduce other environmental issues that are equally important as we go about the task of preserving a sound long-term future for all ocean users, including fishers.

Environmental issues generally, and in particular the sustainability of stocks and dedication of certain areas for marine protection, are at the top of this industry's agenda. From the outset, I should stress that industry does not see these issues simply as threats. They may be challenges – but they are also opportunities.

SUSTAINABILITY OF STOCKS

No fishery can be profitable in the long term unless the stocks are harvested in a sustainable manner. There have been times in past years when stocks have been depleted too quickly, largely as a result of ineffective management and planning. Today we have moved closer to world's best practice, with tight controls over total allowable catches in Commonwealth fisheries under the Australian Fisheries Management Authority legislation and under similar State bodies.

Industry does not automatically oppose declaration of some areas in order to establish a comprehensive, adequate and representative system of MPAs. About two years ago, for example, Environment Australia moved to declare reserves over the Cartier Islands, off the north-west Australian coastline. The proposed declaration was subjected to quite comprehensive stakeholder consultation. The fishing effort in that region was minimal, and there was a case for protecting the area's breeding and nursery grounds. At the same time, the declaration provided an extra tool against illegal fishing from foreign vessels, by offering new resources for monitoring and surveillance. As it turned out, the area was perhaps most notable for its number of unexploded devices from World War Two! That might help explain in part why the fishing effort was traditionally light! Declaration of Cartier Islands became a "win-win" situation for all stakeholders. The crucial factor was meaningful consultation and a responsible and transparent approach by the authorities to the legitimate needs of ocean users in that region.

INDUSTRY PARTICIPATION

The seafood industry across Australia continues to actively participate in planning marine protected areas. Industry does not oppose MPAs *per se*. Rather, fishers want any MPA proposals to be dealt with individually, on merit. Marine parks and reserves in the right areas can yield benefits such as breeding grounds for some species of fish stocks, while protecting our heritage.

However, care must be taken to ensure that the principles of conservation and fisheries management are not confused in the MPA process, because if they are sited in the wrong areas MPAs create unnecessary limitations on seafood harvest and impose intolerable burdens on fishing families. Many of these issues have been explored in an important independent Report delivered earlier this year by the University of Canberra (Baelde *et al.* 2001). The Report poses many challenges, not just to the fishing industry but to all of us. In particular, it recommends that a national approach be taken to the planning of marine protected areas, instead of the current lack of coordination of State and Federal effort. I quote in part from the Report:

"The implementation of MPAs is being superimposed on a variety of existing conservation and fisheries management initiatives.

"Not surprisingly, this creates uncertainty and apprehension among users of marine resources and particularly among fishers who are most directly impacted by MPAs.

"In the main, MPA policies are being developed without due input from the fishing industry despite the significant potential impact of MPAs on fishers' access to marine resources.

"For issues as fundamental as access to fishing grounds, extensive and intensive consultation and debate are essential.

"Support from the fishing industry will be dependent on clear and unambiguous answers to the many questions which currently cloud understanding of the efficacy of using MPAs as resource conservation and allocation tools".

Let me offer one example which gives a direct insight into why these concerns have become so apparent to the researchers. Currently the Australian Seafood Industry Council (ASIC) is taking part in consultations on 11 areas identified for assessment by the Federal Government. Regrettably these areas were announced in September last year, not long before the election. With a little luck on our side, industry became aware of the proposed nominations for assessment just on the time they were to be announced. We immediately moved to request a formal consultation process. That process began – but the first stakeholder meeting was on the same day that the Minister announced 11 areas for assessment.

Since then, however, a rigorous process has been established and stakeholders are at least being heard. Key issues include the way IUCN (or "World Conservation Union") categories will be applied to Australian waters. Those IUCN categories include, for example, "nature reserve", "wilderness area", "national park", "habitat species management area" and "managed resource protected area". I am pleased to report that Environment Australia has suggested a constructive way forward, creating some flexibility in the way those IUCN categories are applied to our unique conditions in Australian waters.

LINKS WITH NATIONAL OCEANS POLICY

Parallel with these talks is the development and implementation of Australia's National Oceans Policy. Administered through the National Oceans Office in Hobart, this is looking to establish – for the first time – holistic approaches to the legitimate use of marine resources. Again, ASIC is a stakeholder in the National Oceans Advisory Group.

The seafood industry has worked hard to develop meaningful policy on marine protected areas and will continue to protect the interests of seafood producers and the post-harvest sector. Development of a South East Region marine plan – the first of 11 areas of Australian waters – is now well advanced and will serve as a template for marine plans across other sectors of the 11 million sq km of Australian waters.

INDUSTRY RECORD

The Australian seafood industry has a proud record of being pro-active on environmental and sustainability issues. To take one example, since 1989 industry has provided core funding to establish an independent environmental NGO called Oceanwatch, to represent and act upon environmental issues fundamental to the long-term viability of Australia's marine and coastal resources and ecosystems.

For the past three years, Oceanwatch has overseen the delivery of an environmental extension service for the commercial industry called Seanet. The Seanet program now has five extension officers based in four States and one specific to the Eastern Tuna and Billfish Fishery. Seanet officers consult with wild-catch fishers on board, with the following aims: to see where their environmental needs are and how improved environmental practices can be introduced; to enjoy face-to-face contact working with fishers to increase awareness and improved environmental performance in their commercial fishing activities; to facilitate research and communication about new technology to minimise by-catch or waste; and to help to develop environmental management systems which are practical for everyday fishing operations.

A classic example of a Seanet project is the development by Mr Dennis Ballam, Queensland-based extension officer for Seanet, of a so-called "pinger" device. This fairly simple but highly effective device emits a radio signal that alerts marine mammals to the presence of fishing nets and discourages them from entering that immediate area. Trials have been so successful that demand has outstripped the supply for quite some time.

Seanet had been strongly supported by the Federal Government through allocations under the Natural Heritage Trust. The support from the industry base, our wild-catch fishers across Australia, has been outstanding.

This is another example of industry's addressing environmental goals before regulations are needed to force those outcomes. Oceanwatch and its Seanet program continue to build its influence and deliver outcomes around Australia, and I am sure will continue to do so for many years to come.

Conclusions

There are ways forward on marine protected area planning. To achieve outcomes, what the fishing industry seeks is a responsible, co-operative approach by all stakeholders. We can have "win-win" situations; we've shown that in the past. But equally, where unreasonable criteria or regulations may be applied, we can fully expect the fishing industry to stand up for its rights.

Our common, long-term objective must be to continue a viable, productive and sustainable seafood industry that remains cognisant of the need to ensure our high-quality ocean environment remains a resource for us and for future generations.

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Theme 2

KEYNOTE PRESENTATION

OPTIMAL DESIGN OF INDIVIDUAL MARINE PROTECTED AREAS AND MPA SYSTEMS

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Abstract

Aquatic and marine protected areas are being deployed in ever-increasing numbers to meet a wide variety of conservation and resource management objectives. These objectives include what might be considered the traditional goals of conservation: protecting habitat, recovering endangered species, managing fisheries, and creating controlled areas for research on ecology and on efficacy of management interventions. In addition, marine protected areas are sometimes used to provide a basis for sustainable use of resources, to resolve user conflicts, to safeguard traditional livelihoods, to empower local people and give them a larger role in decision-making and management, and to promote local economic development. Some protected areas have narrow goals and are simple in design; others serve multiple uses and can employ quite complicated spatial and temporal regulations concerning use. The physical design of any successful protected area and its governance arrangement must reflect the specific objectives that it targets – and since conditions, needs, and objectives vary so widely around the world, no single model exists for effective marine protected areas. There are, however, consistently applicable approaches for planning both the siting and design of reserves, beginning with the articulation of specific, measurable objectives and benchmarks. This goal-setting is best accomplished through a participatory process, and should not be the purview of scientists, conservationists, or resource managers working in isolation. The process by which marine protected area networks can be designed and implemented can also be standardized, to ensure that both the individual protected areas and the wider system remain flexible as needs and conditions change, and the network ultimately serves its purpose.

Keywords: marine protected area (MPA), planning, MPA networks, objective-setting, multiple use

INTRODUCTION

Interest in marine protected areas (MPAs) has grown at an astonishing rate in the past two decades, leading to a proliferation of both protected areas and experts specializing in them.

The broadening of the discipline and the proliferation of MPAs is attributable to at least two general trends: 1) ubiquitous declines in ocean health and productivity, especially in the nearshore coastal zone, have greatly increased the need for new approaches to resource management and habitat protection; and 2) experience from MPAs that have been in place for some time has allowed us to reap important lessons (Agardy *et al.* 2003). But perhaps the most important factor accounting for the steady increase in MPA use worldwide has been the flexibility that the tool provides. The spectrum of MPAs now runs from small-scale fisheries reserves and tightly controlled scientific research sites to large-scale multiple-use areas and coastal biosphere reserves – spanning an almost infinite number of possibilities in between (Salm and Clark 2000; Ward *et al.* 2002). These diverse approaches to MPA design reflect different problems that managers seek to address with spatial management initiatives – thus some MPAs target the maintenance or recovery of a fishery, others restrict destructive fishing techniques, others promote integrated coastal management, and still others serve sociopolitical ends such as safeguarding livelihoods of local communities or giving such communities a greater role in resource management and decision making.

Although this is in itself a good thing, the variety of approaches, philosophies and ultimate purposes for which MPAs become established has resulted in confusion (and in some cases, antagonism) in the conservation and resource management community (Agardy 2002). As with many popular trends, the frenzy to proclaim myriad, sometimes untenable policy prescriptions and the tendency to put forward extreme views in the hope of attracting attention threatens to make a fad out of serious conservation efforts, and may seriously derail our progress towards effective marine conservation as a result (Agardy *et al.* 2003). Paradoxically, it is the broad applicability of the tool that has set the stage for professional disagreements in the marine conservation community. If these rifts are left unchecked, the end result may well be confusion among decision-makers, causing them to reject the tool and perhaps even dismiss other legitimate conservation approaches, ultimately leading to a derailment of marine conservation efforts altogether.

Marine protected areas can be used to their full potential to combat the widespread degradation of the seas – indeed, they are perhaps the strongest weapon in the arsenal available to us today. However, there are several rules of thumb concerning the selection of sites and the design of MPAs that should be followed in order for protected areas, and thus marine conservation, to be maximally effective. These rules of thumb include the following: 1) be flexible and open-minded; 2) avoid simplistic formulas; 3) set objectives first; 4) follow standardized planning procedures that use science appropriately; and 5) think big, using networks of MPAs and regional coastal planning.

RULE 1: BE FLEXIBLE AND OPEN-MINDED

Marine protected areas are variously defined as purely in-water designations, as coastal management units that include terrestrial and marine areas, as strictly protected reserves, or as any kind of marine managed area. The most commonly used definition of MPA internationally is that provided by IUCN, "any area of intertidal or subtidal terrain, together with its overlying water and associated flora, fauna, historical, or cultural features, which has been reserved by law or other effective means to protect part or all of the enclosed environment" (Kelleher and Kenchington 1992). This generic description has metamorphosed somewhat in subsequent discussions and treaty negotiations, as for example background documents for the Convention on Biological Diversity that state "MPAs are coastal or oceanic management areas designed to conserve ecosystems together with their functions and resources" (deFontaubert *et al.* 1996). To counter the increasingly vague general definitions that have been put forward for MPAs, a profusion of specific terms to describe various sorts of MPAs have been adopted, including marine park, marine reserve, fisheries reserve, closed area, marine sanctuary, nature reserve, ecological reserve, replenishment reserve, marine management area, area of conservation concern, sensitive sea area, biosphere reserve, coastal park, national marine park, and marine wilderness area, among others (Salm and Clark 2000; Jones 2002).

Given the potential for misunderstanding and confusion, it is no surprise that MPA advocates have clamored for a single, broadly accepted definition of what constitutes a marine protected area. However, by contending that a single definition can be broadly applied to MPAs everywhere, we must also contend that MPAs are being used to accomplish the same objectives everywhere they are implemented. In fact, the array of goals, and their order of priority, varies enormously from place to place – so much so that one could almost say that every MPA is unique, having been tailored to meet the specific circumstances of the place wherein it is established. It thus may be more useful to adopt the single umbrella term – marine protected area – which can apply to a wide range of different habitat protections, and work to create a broadly accepted typology of terms that would clearly define each sort of MPA according to the objectives it is setting out to achieve.

The fact that MPAs can accomplish a broad range of objectives does not mean that the tool is inherently ambiguous and not scientifically rigorous, nor that the process of designing and implementing MPAs cannot be standardized. What it does imply, however, is that MPA planners and advocates (including those lobbying for policy reform to ensure that MPA designation will be possible in places where legislation is currently inadequate) must be careful to clearly define targeted objectives for MPA systems as well as for each individual MPA.

There is already some misunderstanding regarding multiple-use MPAs and the extent to which their conservation benefits can be adequately assessed. Although multiple-use situations are inherently more complex than fisheries reserves or scientific research sites that have a narrower set of goals, even very large and complex multiple-use areas can be rigorously monitored to see whether their management objectives are being met over time. When multiple-use areas are zoned for specific uses – allowing the segregation of potentially conflicting activities –the performance of each zone can be readily assessed (Day *et al*. 2003). The prerequisite, of course, is that specific objectives be articulated for the areas (or for each zone) and that measurable indicators and benchmarks be defined against which progress can be objectively measured. When only broad goals are mentioned in planning, such as "conservation of biodiversity" or "integrated coastal management", measuring progress becomes almost impossible. MPAs then suffer the risk of becoming meaningless paper parks. Luckily, thanks in part to demands made by donor community, conservation NGOs and government agencies are increasingly obliged to identify performance indicators and measure management effectiveness throughout the life of the projects. Such monitoring not only serves donors that demand information on their investments, it also facilitates the kind of adaptive management that is necessary in light of our limited understanding of the marine environment.

Denying uncertainty and pretending that a single model MPA exists that can be applied in all situations creates a risk we cannot afford to take. When advocates of MPAs make sweeping statements about the benefits of MPAs, we raise expectations in user groups and put MPA cynics on their guard (Agardy *et al.* 2003). Meeting these often unrealistically high expectations then puts unnecessary pressure on MPA managers, threatens the continued existence of these MPAs, and even endangers future MPA designations. The consequences are not just disappointments and bruised egos – in many cases sunset clauses are written into MPA legislation, requiring that certain targets (usually increases in fishery biomass) be reached within a certain timeframe lest the MPA be revoked, or at least deprived of its funding. Although it is imperative that performance be strictly monitored in all MPAs, we should be wary of traps that unrealistic targets pose for conservation interests.

Some scientists contend that only strictly protected, no-take fisheries reserves are legitimate as a conservation tool. In doing so, they ignore the host of reasons for which marine protected areas can be established. Even when fisheries management is the objective, fisheries uses can and should be accommodated when they present no threat to fish stocks or marine biodiversity. An example of this is the Mafia Island Marine Park in Tanzania, where local communities successfully established a park to stop the dynamiting of reefs by mainland fishers (Agardy 1997). Although this was not a no-take reserve (and hence was viewed by some as a non-legitimate MPA), the regulations prohibited destructive fishing while allowing other non-intensive and sustainable extractive activities to occur. Yet theoretical scientists who are not versed in real-life problem solving might see the park, and others like it, as a non-rigorous approach that led to failure. Practitioners might better recognize it as an adequate solution.

The problem that a rift between researchers and practitioners creates is twofold: 1) rather than clarifying the scientific validity of MPA benefits, it creates confusion for those who are searching to find the appropriate conservation tool to fit their needs, and 2) it dismisses the very valid other sorts of benefits that MPAs can and should be used to achieve in many parts of the world. Such benefits include resolving user conflicts through multiple-use zoning, empowering local communities in decision-making concerning management of local areas, and, perhaps most importantly, providing small-scale examples of integrated and equitable coastal management in regions of the world where how to achieve coastal management is not well understood (Agardy 2000).

A third, and perhaps the major, problem with all this has to do with perception and misconception; by implying that only MPAs that fence off the ocean and keep people out are worthwhile, scientists unwittingly draw battle lines between themselves (and the decision-makers they have been able to convince) and user groups. Experience shows that this dangerously undermines the ability of managers to successfully implement marine reserves and protected areas (Agardy *et al.* 2003). In fact, the best examples of MPAs around the world are those that have drawn fishers and other users into the planning process, creating strong advocates for the MPAs among the very groups most highly affected by the restrictions MPAs put in place (Agardy 1997). To draw such battle lines and confront fishermen and other legitimate users of the oceans is to enter into unnecessary, and probably costly, battles that we cannot afford.

RULE 2: AVOID SIMPLISTIC FORMULAS

The push to create scientific consensus statements and publish theoretical papers on MPA design criteria is easily understood as a necessary response to the proliferation of many meaningless MPA designations and other toothless management measures being adopted around the world. Taking stock of what we know from scientific study is an important prerequisite to being able to move away from paper parks, and move toward more effective marine and coastal management. However, the rush to categorize, quantify and simplify the complex issues around MPA design may be counterproductive. This is especially true regarding efforts to identify and use a single target to describe the minimum amount of area of any protected marine habitat that should be set aside as no-take.

Much recent scientific discussion has centered on trying to identify such minimum targets to ensure that MPAs meet their objectives in a rigorously quantifiable way (Bohnsack 1996). These discussions have been spurred by an increasing frustration among decision makers and managers regarding the lack of objective design criteria and quantifiable benchmarks for performance evaluation. And, for reasons of human nature and the ontogeny of MPA popularity among scientists, these scientific deliberations have created rigid dogma concerning what size of area should be set aside within reserves as no-take (areas with no resource extraction whatsoever). The origin of the 20% is unclear (or, in any case, widely debated), but it is certain that it was extrapolated from very specific localized studies of particular fisheries within particular habitats – not from representative community ecology data from a wide range of habitat types. The initial science concerning minimum no-take determinations included home-range studies and population dynamics data that were used to predict the minimum area needed to reach a particular fisheries management goal. And for a small subset of fisheries in a particular biome, the figure may indeed be valid (NRC 2001).

Now, however, in light of the paucity of other data supporting MPA design criteria, this 20% figure has been elevated to a standard for the minimum size a reserve must set aside as no take in order to be effective. And it is now hallowed ground. Scientists are pushing for 20% no-take targets in all coral reef systems (e.g. the US Coral Reef Task Force decision of 2000 stipulating a national target of 20% no-take in coral reefs under US jurisdiction), and the figure has even come up as a potential target for all marine habitats within US jurisdiction (e.g. discussions leading up to the MPA Executive Order of 2000). Other countries are following the USA lead in clinging to the magical 20% figure, without understanding the shortcomings of doing so.

What could possibly be wrong with establishing hard targets for MPAs, in the context of greater demands for rigorous approaches to marine management? One set of potential problems centers on whether the figure is right, and whether adherence to such a hard target will produce the expected results under widely varying circumstances and in different habitats (Lauck *et al.* 1998). Though it is alluring to think that a single areal target will truly describe the minimum level of protection needed to maintain productivity and biodiversity (as in species assemblages) of any given community of organisms, it is probably disingenuous to make the claim. In fact, studies of highly productive and dynamic temperate water systems suggest that up to 80% of the area would need to be set aside as no-take in order to derive the kinds of fisheries management and biodiversity benefits that scientists advocating 20% no-take claim their formula will accrue (Parrish 1999). Hence, one very real danger of pushing the 20% minimum no-take target is that even such rigorously designed MPAs may not meet expectations – causing the public and the decision-makers who represent them to abandon support for MPAs altogether. For MPA legislation that includes sunset clauses if goals are not met within a certain period of time (e.g. the regulations establishing the Florida Keys National Marine Sanctuary), the raising of unrealistically high expectations can yank the very teeth out of MPA legislation that the advocates for areal targets are pushing to have in place.

Another potentially enormous problem with simplistic policy targets such as 20% no-take areas is that they provide no guidance on what areas should be protected, from what. Take a typical coral reef system in the USA, where managers are now struggling with the question of what portion of the reef to set aside as the no-take zone. Even in a situation where there are no local inhabitants to rally in opposition of new restrictions (such as actually exists in remote parts of the Hawaiian Island chain), the formulaic command to zone 20% of the area as no-take comes with no guidance as to which parts of the reef should be strictly protected, in one big patch or many small ones. Unlike this relatively simple situation, in most areas where MPAs are being designed to address conservation needs, human populations and marine tenure do factor in, and the decision on where to site the no-take areas to make up the target becomes even more complex. In the end, the tendency will always be to establish the no-take areas in the remotest, least used areas, where strict restrictions can be imposed with minimal resistance. These, unfortunately, are the areas where MPAs are least needed – often being relatively unproductive areas with little biodiversity value, and places where conservation problems arising out of resource-use conflicts are absent altogether. Such stances also threaten to create the false sense of security that marine issues are being dealt with adequately.

RULE 3: SET OBJECTIVES FIRST - ALL ELSE WILL FOLLOW

As mentioned previously, and perhaps by now *ad nauseum*, marine protected areas can serve a wide variety of goals and objectives (Jones 1994). In recognizing this broad spectrum of possibility, we must take as a basic assumption that since goals for MPAs vary, so do the ways that they are designed, implemented, and evaluated. Yet there are standardized procedures that one can follow to ensure that MPAs have a solid grounding in science and will thus be optimal in reaching those goals (such a procedure is discussed in the next section). Arguably the most important step in the process is to identify and clearly define the reason the MPA is being established – a process otherwise known as goal-setting (Agardy 2000). This crucial first step is not the purview of scientists (except where scientists constitute a single, legitimate stakeholder group). Instead, the process should be participatory, involving as many views and stakeholder interests as possible.

Individual MPAs can be a powerful tool for conservation, and are urgently needed to stem the tide of marine biodiversity loss. Such marine protected areas run the gamut from strictly protected areas closed to all extractive activities (e.g. fisheries reserves) to large-scale multiple-use protected areas (marine sanctuaries and biosphere reserves). These protected areas or reserves could protect key habitats by restricting activities harmful to the environment and to biodiversity. In this, marine reserves are a crucial conservation measure to safeguard vulnerable populations, species and habitats from the negative impacts caused by exploitation of marine resources (Pauly *et al.* 2002; Ward *et al.* 2002). However, they serve other important functions as well. In addition to providing conservation *in situ* by limiting destructive activities in a particular place, marine protected areas also provide benefits *ex situ* by boosting production in areas outside as well as inside the reserve (Halpern and Warner 2002), and by providing much-needed demonstration models of how to integrate management of coastal and marine resources across sectors (Agardy 1997). Marine protected areas similarly provide vital testing grounds for management measures and much-needed control sites to gain better scientific understanding of how marine systems function as well as how we can better protect them. Finally, they serve to highlight the inherent value of coastal areas, bringing new attention to these important habitats and creating the political will to shift from reactive to proactive mode (Agardy 1999).

Whatever the specific goal of an MPA, the crucial requirement is that identification of that goal occurs through a participatory process using mediators or experts in conflict resolution. Objectives should never be decided unilaterally by scientists who want to force their agendas on society as a whole. There are appropriate roles for both science and scientists, but dictating MPA planning is not one of them!

RULE 4: FOLLOW STANDARD PLANNING PROCEDURES THAT USE SCIENCE APPROPRIATELY

Science can be used, and abused, in MPA planning. To ensure that science is used appropriately, planners need to think through when science should be harnessed, and how – including reviewing the choice of the scientific disciplines to be enlisted in the planning process. Economic, political, and social science will be every bit as useful as the natural sciences, especially since what underlies MPAs is changing human behavior rather than changing nature.

A scientifically rigorous, objective standardized procedure for selecting MPA sites and designing individual MPAs would have the following basic elements:

- Identify and involve all user groups or stakeholders;
- Set realistic objectives through a participatory process;
- Study the area (using all applicable sciences, as well as local knowledge) to determine environmental threats and impediments to realizing objectives;
- Develop outer bounds of the MPA to reflect objectives (societal and ecological);
- Develop a preliminary zoning plan to accommodate different uses (if multiple use is a goal);
- Amend zoning to reflect user-group expectation and needs;
- Formulate a management plan to address threats and accomplish objectives;
- Develop necessary regulations and voluntary compliance to carry out management;
- Monitor to see if objectives are being met over time; and
- Amend management as necessary.

We should do everything in our power to develop full support for the use of MPAs, in all their various forms, to meet a wide range of objectives in a diverse set of circumstances. But rather than looking for simplistic solutions to complex problems, and rather than letting scientists drive what should be an inclusive, participatory process, we should use science appropriately. It is possible to harness science to tell us what we know and do not know about marine systems, and what can be done to maintain them. Part of this scientific role is to determine the relative ecological value of certain areas – information that will help guide planners and managers in designing both multiple-use MPAs and no-take reserves. Another crucial scientific role is to objectively evaluate threats to marine biodiversity and ecology, so that we can tailor solutions to threats for maximum effectiveness. And yet another critical role of science is to help objectively monitor our progress, and in so doing develop the information necessary to amend and improve management over time. Science can also be used to predict the outcomes of different management policies, so that decision-makers can have a clear understanding of how to weigh choices (e.g. Villa *et al.* 2002). In all of this, we should strongly advocate the most rigorous possible approaches, and defend the use of science, yet we must be careful that science does not drive policy unilaterally. Honesty is important – whether or not such honesty highlights how little we know, or what we can indeed never know scientifically.

RULE 5: THINK BIG, UTILIZING NETWORKS AND REGIONAL COASTAL PLANNING

Sadly, individual MPAs will never get us to where we want to be in conserving ocean and coastal ecosystems, no matter how well planned and executed. Recent reviews of MPAs (e.g. Halpern and Warner 2002; Ward *et al.* 2002), though highlighting the benefits of MPAs, can only serve to underscore this point. We must look up from the largely localized set of conservation problems existing in individual MPAs and look at the big picture. Aiming high will undoubtedly result in some frustrations and occasional failure, but not aiming high will result in our going nowhere.

Recognizing that marine ecosystems, species and coastal communities are inexorably linked, and that piecemeal efforts to protect the marine environment have been largely unsuccessful, a need exists for a strategically developed system of marine protected area networks spanning the critically important coastal waters. The linkages in these networks have a dual nature: they connect physical sites deemed ecologically critical, and they link people and institutions in order to make effective conservation possible (Agardy and Wolfe 2002). Because marine protected areas and networks of areas can target a wide range of objectives and vary greatly in scope, a system of networks that is essentially a hierarchy can be planned to optimally conserve entire regions (Zacharias and Roff 2000). At each level within this hierarchy, both humans and marine ecosystems are drawn into networks, making coordinated, effective and efficient management possible.

Such a hierarchical approach would allow the neighboring states and nations to address various geographic scales and scopes of marine conservation problems simultaneously. The hierarchy is not an artificial construct. Ecosystems and socio-political systems are hierarchical. Marine conservation issues thus vary in scale, hence the goals of marine protected area must likewise vary. Thus, the hierarchical approach is a natural response to a complex set of problems, and is likely to be the most efficient way to allocate scarce time and resources to combating the issues.

At the very grandest scale –a large geographic region – a system of marine protected areas and linked organizations that effectively protect representative samples of marine biodiversity at the habitat level could be instituted. This representative system would eventually comprise at least one example of every marine and coastal habitat type. Developing such a representative system would require development of a clear, consistent and mutually acceptable system of classification of marine eco-regions and habitat types within them. Of course, one would also have to know what elements were already being protected in various sorts of protected areas (including traditional MPAs, coastal MPAs that might be wetland protected areas and even riverine protected areas, and private land-holdings or tenurial arrangements). However, elementary, accurate and comprehensive inventories of MPAs do not exist for most places, including North America. Nonetheless, achieving a representative system of MPAs is not an insurmountable task – but it does require the big-picture view. In Australia, the Great Barrier Reef Marine Park Authority has gone a long way towards this ambitious goal, and many other countries will soon be able to look to the Australian experience for guidance.

At the next level down in the nested hierarchy, networks of MPAs could be used to conserve key species, whether they be those commercially valuable to a wider region or those threatened across a wide geography. Use of species as focal points for conservation is controversial (see Zacharias and Roff 2001), yet it is highly appropriate in a nested hierarchy of MPA networks where different jurisdictional entities can agree on shared conservation priorities. In North America, for instance, the Commission of Environmental Cooperation (known as the CEC – the environmental sidearm of NAFTA) has catalysed a process by which the three countries of North America (Canada, USA and Mexico) have selected Marine Species of Common Conservation Concern (MSCCC). The CEC convened a multilateral Task Force to identify marine species of common conservation, and is working to develop a set of networks of MPAs to protect the most critical habitats of these species. A system of MPA networks at this scale of organization would seek to protect the focal species through linked protected areas designed to address the specific threats affecting these species in each specific locale. The collective protected areas constitute the sum total of critical areas, identified either by knowing ecological requirements of key species or by having identified the most critical habitats needing conservation in order to preserve overall ecosystem functioning. Umbrella species like those on the MSCCC list can be used to capture what is important from a target-species point of view and from the overall ecosystem perspective. These critical elements (such as upwelling areas and other feeding zones, shallow-water banks, and migration bottlenecks) must be maximally protected and can then be linked by the virtual corridor. Thus, targeted policy reform must ensure that the connectivity is preserved and that these most vital parts are not degraded by direct and indirect impacts of human activity. At this level of organization, it is species and their movements that provide the linkages within the system.

At the next level of organization, the sites selected to be in a representative system or to protect critical habitat for key species would themselves contain networks of MPA sites that act to draw attention and management resources to those areas most ecologically critical. This is a functional approach, akin to identifying and protecting the most vital organs of an ailing organism in order to hasten its recovery. Here, the point of the network would be to focus conservation and management attention on the ecologically most critical areas, address the key threats in those areas, and work to preserve the overall ecological integrity of the site. At this level of the hierarchy, the connectivity or linkages are provided by the dynamics of the ecosystem, and its ecosystem processes.

Finally, there are networks within these networks as well. For each site, effective protection of the vital organs requires coordination of efforts at the national, state or provincial, and local level, as well as a coming together of many different disciplines. Optimally, the network would tie together statutory protections with more widespread (and socially acceptable) voluntary ones (Gubbay 1993). The connectivity in these networks is the linkages between people and institutions – and it points us back in the direction of a parallel system of human networks that are needed to make MPA systems a meaningful reality.

As is the case with individual marine protected areas, networks can be developed to achieve a wide range of specific objectives. Such objectives might include fisheries management/ fish stock recovery, equitable fisheries resource allocations, endangered species monitoring and recovery, scientific research to better understand the marine environment, conflict resolution through zoning, integrated management/adaptive management, and creation of buffer systems for coping with global change (Agardy and Wolfe 2002).

Is 'thinking big' really feasible? Can we jump ahead to hierarchical networks, even as we falter on establishing truly effective single MPAs in so many parts of the world? The answer, I believe, is yes: we must aim high because, if we do not, even our few highly effective marine protected areas are at risk. Unless we protect the context in which these islands of protection sit, and work towards region-wide coastal and marine management, all our hard work and investment will be for nought.

There is another side to our long history of ineffective approaches to marine conservation. The fact that we have had difficulty establishing truly effective marine protected area networks may make this the right time for surging ahead with a new system of protection. There is growing public awareness of our ineptitude in dealing with marine environmental issues, and solid data have been amassed that suggest that marine protected areas are truly effective in meeting many important conservation goals. Furthermore, sectors of society that might have been disinclined to support protected areas in the past are now seemingly ready to do so - as is the case with fishers in the US Northeast – Atlantic Canada who historically resisted additional regulations on fishing but now demand better conservation as demersal fish stocks, and the industry built upon them, have collapsed. The loss of marine biodiversity and the decline in ocean health are of concern to everyone now - and momentum is growing to take what we have learned from conservation elsewhere and apply it to safeguard our own seas and shores. Moreover, with no rigid frameworks already in place, countries have room to be ambitious and creative in their approach.

In essence, the innovativeness of hierarchical networks and regional approaches (rule 5) gives way to flexibility and open-mindedness (rule 1). In the way that marine areas are linked, and socio-political systems cycle, our rules of thumb are also inter-related and synergistic. There is reason for optimism, for where we are today gives us room to get to where we want to be tomorrow.

CONCLUSION

All of us working in marine conservation, whether through advocacy, as purveyors of scientific information, or as practitioners (or trainers of practitioners), welcome the newfound and widespread interest in MPAs that has emerged in the past couple of years. In the USA, a National Research Council panel was assembled to look at MPAs and reserves in the context of USA marine conservation needs (NRC 2001). In Australia, the recent review of Ward and co-authors signifies how seriously the issue is being taken, and how rigorous the discussions around it have become (Ward *et al.* 2002). The publications that emerge out of such deliberations have proven to be invaluable reference documents as well as useful elements in the conservation community's arsenal for turning public opinion around in favor of MPAs. At the same time, the quest for rigor has brought inflexibility, and threatens the progress made to date. Narrow interpretations of what constitute MPAs, objective-setting that is done by a single special interest group (scientists or academics) as opposed to the broadest possible array of stakeholders, adherence to scientifically questionable targets that raise expectations and create easy ways out for decision-makers, and the disingenuous labeling of opinion as scientific truth are all extremely dangerous tactics that will not serve defenders of MPAs or marine conservation interests well in the end. Instead, we must be open-minded, avoiding one-size-fits-all formulae, and we must work collectively to define objectives, using science wisely. Finally, we can and should be ambitious – daring to think big and scale up wherever and whenever we can.

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BROAD-SCALE BIODIVERSITY ASSESSMENTS FOR MARINE PROTECTED AREAS IN NEW SOUTH WALES, AUSTRALIA

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Abstract

The NSW Marine Parks Authority aims to establish and manage a comprehensive, adequate and representative system of marine protected areas (MPAs) to help conserve marine biodiversity and maintain marine ecosystem processes. Recent projects have attempted to systematically assess broadscale patterns of biodiversity within each of five NSW marine bioregions and identify where additional MPAs may be required. The assessments have relied primarily on coarse surrogates for biodiversity, supplemented with finer-scale community and species data where available. A hierarchical classification of ecosystems, habitats, communities and species has been mapped in a geographical information system along with basic information on condition, vulnerability, existing MPAs and other conservation measures. Using this information, alternative locations for MPAs are being assessed with the aid of explicit models of MPA objectives, criteria and performance indicators and a range of support tools for reserve selection and multiple-criteria decision making. This approach is now providing input into selection processes for marine protected areas and a basic framework for ongoing planning, management and research.

***Keywords*:** marine protected areas, biodiversity, conservation

INTRODUCTION

Marine biodiversity assessments in New South Wales (NSW) aim to identify potential locations for marine protected areas (MPAs) in each of the State's marine bioregions (Fig. 1). Scientists and conservation managers have identified 65 Australian marine bioregions and provinces (Interim Marine and Coastal Regionalisation of Australia 1998) to help plan a national system of MPAs. Inclusion of the characteristic biodiversity of each bioregion in a system of MPAs aims to manage a representative cross-section of all marine ecosystems for conservation and sustainable use.

National guidelines and criteria have been developed to identify and select MPAs (ANZECC TFMPA 1998a, 1998b, 1999). These have been adapted to assess existing MPAs in NSW and identify sites where additional MPAs may be required. In NSW, MPAs are of three types: generally large, multiple-use marine parks managed by the Marine Parks Authority; aquatic reserves managed by NSW Fisheries; and those areas of national parks and nature reserves below mean high water, managed by the National Parks and Wildlife Service.

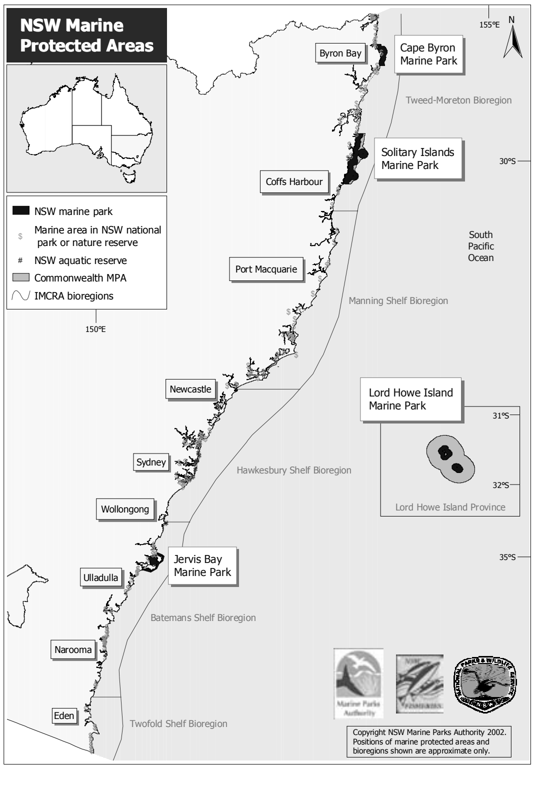

Fig. 1. Interim marine and coastal bioregions for NSW

This paper discusses the broad-scale methods and information being used to identify options for new MPAs on the basis of ecological criteria alone (Figs 2–5). A separate selection process is then required for more detailed site assessment and consultation to consider social, economic and cultural criteria (Fig. 6).

GOALS AND CRITERIA FOR IDENTIFICATION AND SELECTION OF MARINE PROTECTED AREAS

For practical application, national MPA guidelines were interpreted through hierarchical, 'tree-like' models of goals, criteria and performance indicators that interpret broad objectives in terms of more specific criteria and data. The models clearly display and organise goals and criteria and allow MPA options to be systematically assessed from a range of specific information sources using multiple-criteria analyses.

Using this approach, MPA identification and selection criteria can be split into primary ecological goals to conserve biodiversity and ecosystem viability, and secondary goals to provide for human use. Ecological criteria can be grouped under three main branches: comprehensiveness, representativeness and adequacy (Fig. 2).

Comprehensiveness

Comprehensiveness is defined as including within MPAs "the full range of marine ecosystems and habitats" (ANZECC TFMPA 1998a, 1998b, 1999). As marine ecosystems are diverse, continually changing and difficult to define accurately, "surrogate" measures are used to map generally recognised, broad-scale patterns in biodiversity. For the assessments, ecosystem and habitat measures are based on broad-scale differences in geomorphology, depth, substratum and tidal exposure (Fig. 3). These largely physical differences in environments are assumed to reflect a corresponding diversity of habitats, species, and ecological processes.

Representativeness

Representativeness means that MPAs should "reasonably reflect the biotic diversity of the marine ecosystems from which they derive" (ANZECC TFMPA 1998a, 1998b, 1999). MPAs should include a reasonably unbiased, and sufficiently large, representative proportion of the biological variation found among and within habitats and assemblages of species. The aim is to protect typical species, processes and areas as well as the known, charismatic, rare, threatened, scenic, commerical or convenient elements of biodiversity (Fig. 4).

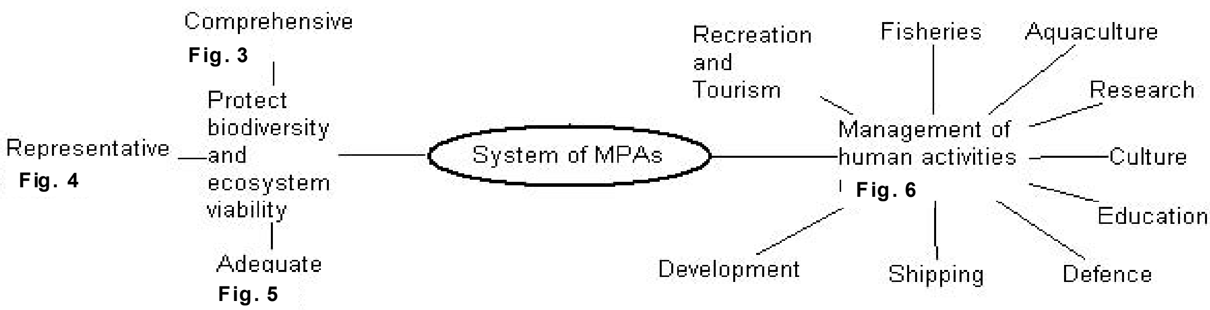

Fig. 2. Primary and secondary goals for a representative system of MPAs.

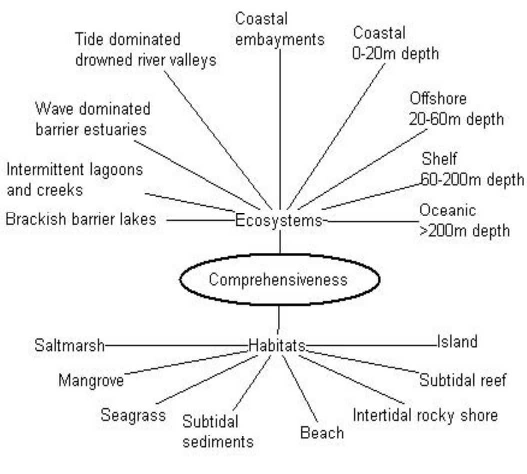

Fig. 3. Criteria for a comprehensive range of ecosystems and habitats in MPAs.

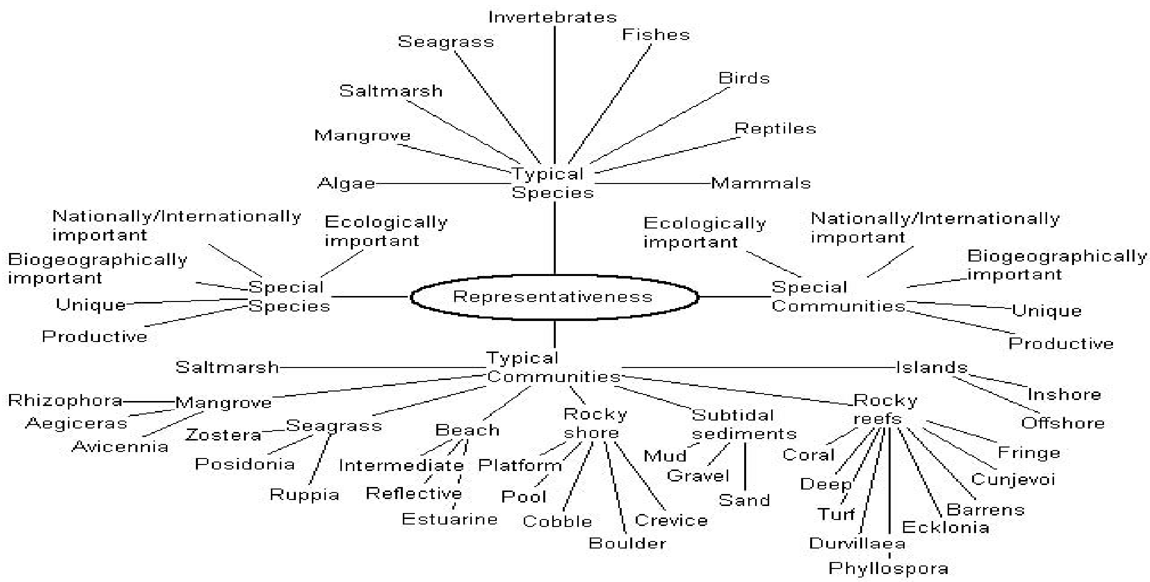

Fig. 4. Identification criteria for representing biodiversity in MPAs.

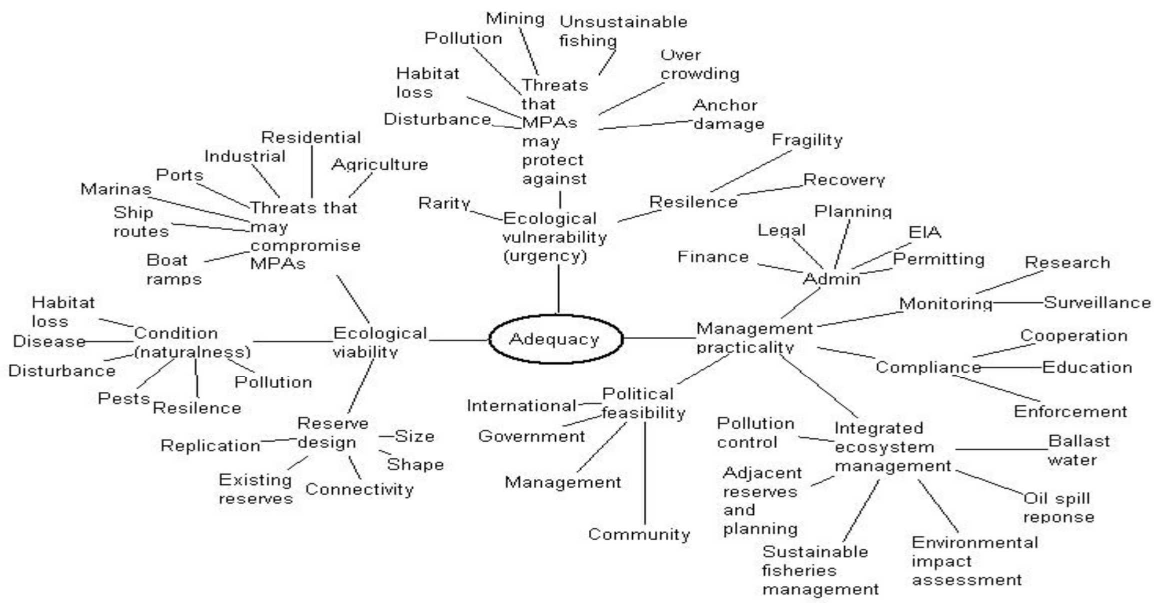

Fig. 5. Identification criteria for adequacy in maintaining biodiversity in MPAs.

Adequacy

Adequacy is defined as the "level of reservation to ensure the ecological viability and integrity of populations, species and communities"

(ANZECC TFMPA 1998a, 1998b, 1999). Adequacy includes criteria that affect the ability of MPAs to sustain biodiversity and involves consideration of condition, vulnerability, reserve design, and practical MPA management (Fig. 5).

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

Vulnerability may be interpreted in two ways. Where there is a range of options available for protection of a feature, it may be preferable to include areas that are least threatened, especially where the threats are beyond the direct control of MPA management. An example would be the selection of marine protected areas with catchments protected by terrestrial reserves. However, where only a few areas for a habitat or species exist, there may be urgent reasons for protecting the areas most threatened, particularly if the threats can be prevented by MPA management.

Design of reserves should reflect the intended purpose of individual MPAs as well as their role in a functioning system of reserves. Reserve design for biodiversity, fisheries management, sedentary and migratory organisms, whole ecosystems or individual species may differ markedly (Agardy 2000; Planes *et al.* 2000; Roberts and Hawkins 2000; Salm *et al.* 2000). Other ecological design considerations include: the use of highly protected 'no take' core zones and surrounding buffer zones; the inclusion of whole ecosystems and habitats within natural boundaries; appropriate sizes, numbers and shapes of reserves; maximisation of habitat complexity; connectivity among habitats; continuity of ecosystem processes; and risk management through replication and application of the uncertainty principle.

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

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

MPAs have a crucial role as reference sites in understanding changing marine environments, the impact of human activities and the effectiveness of management. Without reference sites, there are no reliable baselines for distinguishing natural from human disturbance or for identifying causes of impacts. There should therefore, be careful consideration of experimental design when selecting MPAs. Particularly if subsequent monitoring programs are to provide an objective assessment of whether MPAs are achieving their aims.

Human use

Criteria for human activities are scheduled by national guidelines into a separate site 'selection' process. The potential for conflict among conservation values and competing human interests is evident in even the simplified view shown in Fig. 6. Careful consideration of human activities is therefore required if MPAs are to be successfully implemented and provide for sustainable use.

Where consistent with ecological goals, the selection process aims to accommodate human activities, and even help enhance cultural, social and economic values. In many cases the ecological options for MPAs may be flexible enough to allow for a variety of sustainable use.

In addition, stakeholders can often contribute valuable information on patterns of use, as well as information on species distributions, habitats, vulnerability, condition and threats. When used cautiously, such information may lead to more realistic MPA designs and strategies that adapt to local conditions, habitats and communities (Johannes *et al.* 2000). Subject to intellectual property rights, indigenous and other cultural knowledge should be included in MPA assessment, research, management and education.

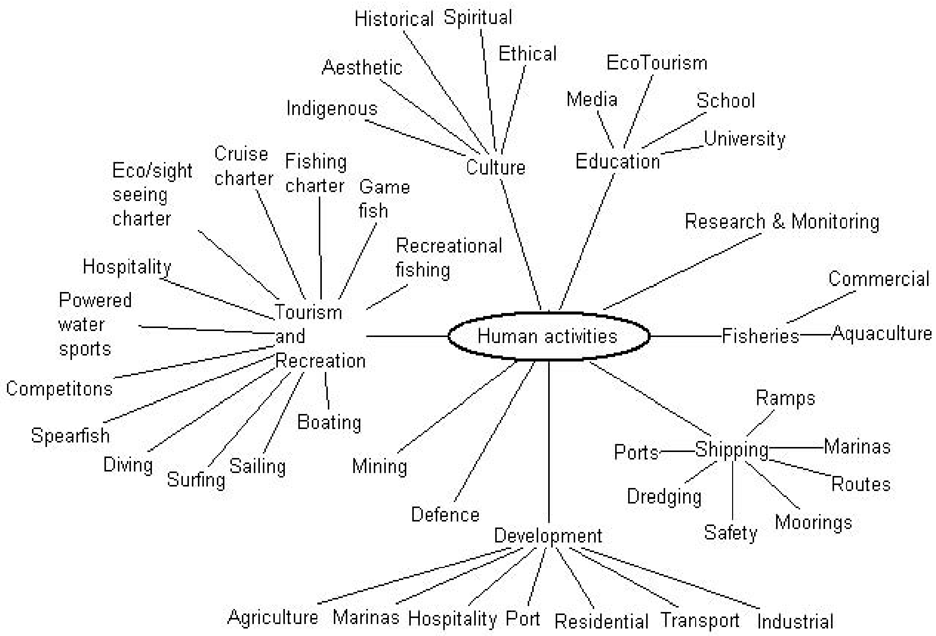

Fig. 6. Selection criteria for managing and allowing for human use.

MEASURES OF COMPREHENSIVENESS AND REPRESENTATIVENESS

The assessments rely primarily on an environmental classification to approximate broad-scale, conspicuous patterns in biodiversity for large areas. Although the mainly physical predictors used are an indirect means of measuring biodiversity, this approach may avoid biases resulting from sampling of selected species and research locations. Physical predictors are also likely to be more stable through time, and inclusion of a range of physical environments within MPAs may assist in maintaining the processes on which biodiversity and its continued evolution depends.

On the other hand, the approach assumes that the mapped physical features correlate with actual variation in the distribution, abundance and diversity of organisms. For obvious patterns in biodiversity, such relationships may be well documented. However, these assumptions should not remain untested, and conservation programs should aim to understand the biological basis for biodiversity and manage accordingly.

A hierarchical classification developed in conjunction with the NSW Marine Parks Authority Research Committee is being used in the assessments to represent progressively finer scales of ecological variation. Levels in the hierarchy are:

- IMCRA bioregions
- 'ecosystem' classes based on estuary type and depth zone
- 'habitat' classes based on substratum, exposure and vegetation
- 'community' classes based on more detailed physical surrogates, dominant biota or species assemblages
- estimated distributions and abundances of species and populations.

Methods used to map the marine ecosystem and habitat classes are discussed below and their use justified.

Estuary ecosystem classes

Coastal waterbodies from the NSW Waterways Geographical Information System (GIS) coast coverage were classified on the basis of coastal morphology, entrance type and tidal exchange according to Roy *et al.* (2001) who associate these differences with characteristic ecosystem processes and related assemblages of organisms. The classes include:

I. Ocean embayments. These semi-enclosed bays are transition zones between estuaries and ocean with communities of both, generally low turbidity, ocean tidal ranges and salinities, and sandy areas with seagrass beds in protected locations, e.g. Jervis Bay.

II. Tide-dominated drowned river valleys. These are tidal, generally deep, narrow estuaries with rocky sides, sometimes with large, submerged, sand deltas extending up the estuary, e.g. Port Stephens.

III. Wave-dominated barrier estuaries. Young barrier estuaries in the early stages of infilling have large shallow lagoons with dense seagrass beds away from the main tidal channels, e.g. Wallis Lake. Mature estuaries in the late stages of infilling form a riverine estuary with extensive flood plains and coastal wetlands. They often have narrow, elongated entrance channels and broad barrier sand flats, e.g. the Clarence River.

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

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

Ocean ecosystem classes

Depth contours digitised by NSW Waterways from Australian Hydrographic Office nautical charts were used to divide the shelf into four depth zones: 0–20 m, 20–60 m, 60–200 m and waters deeper than 200 m. These zones aim to account for biotic and abiotic variation across the shelf in algae (Womersley 1981), sponges (Roberts and Davis 1996), benthic fauna (Coleman *et al.* 1997, Gray 1997), fish assemblages (Graham *et al.* 1996), sediments, currents, temperature, salinity and water chemistry (Short 1993; Chapman *et al.* 1982; Skene and Roy 1986; Colwell *et al.* 1981; Rochford 1975; Godfrey *et al.* 1980).

Seagrass, mangrove and saltmarsh habitats

The distributions of seagrass, mangrove and saltmarsh habitats were estimated from a GIS coverage digitised by the National Parks and Wildlife Service from West *et al.* (1985). Mangrove and saltmarsh communities contribute significantly to the productivity of estuaries through nutrient cycling and trapping of sediments and detritus. They provide habitat for characteristic and highly diverse assemblages of fish, birds and invertebrates (Hutchings and Recher 1982; Saenger 1999). Seagrass beds are widely recognised for their role in providing habitat for diverse assemblages of flora and fauna (Bell and Pollard 1989; Hannan and Williams 1998; Howard and Edgar 1999).

Intertidal rocky shore habitats

A linear GIS coverage of intertidal rocky shore was classified using the AMBIS (Australian Land Information Group's Australian Marine Baseline Information System) high water coastline and 1:25,000 topographic maps from the NSW Land and Property Information Centre (LPIC).

Otway (1999) observed that the number of species on a given shore was positively correlated with the number of different substrata present. Sections of rocky shore were therefore scored for the presence of five "community" level substrata (platform, boulder, cobble, pool, crevice) identified by Otway and Morrison (unpublished) during field trips to accessible sites.

For some locations, areas of intertidal rocky shore were also mapped as the difference between high and low water AMBIS coastlines. Aerial photographs (~1:10,000 scale, from the NSW Department of Land and Water Conservation – DLWC) were used to class shores as either bedrock or boulders and gravel.

Intertidal beach habitats

Beaches along the AMBIS high water coastline were classified according to 1: 25,000 topographic maps (LPIC) and Short (1993). Areas of intertidal beaches were mapped as the difference between the AMBIS high and low water coastlines and classified using ~1:10,000 scale aerial photographs (DLWC).

Justification for the classification of beaches in NSW is provided in part by Hacking (1998a, 1998b), and is based on relationships described in Brown and McLachlan (1990). In general, the species richness and abundance of invertebrate macrofauna increases from low (reflective) to high (dissipative) energy beaches (Hacking 1998a; Brown and McLachlan 1990).

The waters over beaches and intertidal flats also support characteristic phytoplankton and fish (Robertson 1999). In Western Australia, surf zones of exposed sandy beaches are important nursery grounds for fish previously considered to be estuary dependant (Lenanton and Caputi 1989; Robertson 1999).

Detached macrophytic algae, commonly found drifting in the surf-zone following heavy seas, supports characteristic communities of organisms different to those found on plants of nearby reefs (Robertson 1999).

Dunes and sand spits above the littoral zone provide important nesting and feeding sites for a range of wader and seabirds including the threatened little tern (*Sterna albifrons)* and beach stone-curlew (*Esacus neglectus*), and the vulnerable broad-billed sandpiper (*Limicola falcinellus)* and black-tailed godwit (*Limosa limosa*).

Island habitats

An arbitrary 100 m buffer was extended from islands and emergent rocks in the AMBIS low water coastline to represent their influence on surrounding waters. Islands and their surrounding waters provide unique and important habitats for seabirds, marine mammals, fish, invertebrates and other species. They have been shown to interact with southward and northward flowing currents to generate fronts, wakes and other oceanographic features that extend well beyond the rock or island (Cresswell *et al.* 1983). Such features are important for the feeding ecology of many fish and invertebrate species and the transport and retention of larvae (Kingsford and Choat 1986; Kingsford 1990; Kingsford and Suthers 1994, 1996; Wolanski 2000).

Subtidal reef habitats

Shallow near-shore reef systems and intervening sediment patches were mapped from 1:10,000–1:25,000 scale aerial photographs (DLWC) to a depth of 10–20 m depending on conditions at the time the photographs were taken. This coverage of mostly inshore reefs was supplemented offshore with reefs and shoals evident in Australian Hydrographic Service 1:150,000 scale nautical charts. The latter coverage provided a crude indication of the position and extent of more prominent reefs but no information on deeper, low relief reef systems known to exist on the NSW inner continental shelf. For some areas, high resolution bathymetry was plotted from Royal Australian Navy hydrographic survey manuscripts.

Subtidal rocky reef areas in NSW provide habitat for distinctive assemblages of invertebrates, algae and fishes. Habitats within shallow rocky reefs have been described by Underwood *et al.* (1991) as a mosaic "seemingly related to depth, wave exposure and a number of biological processes, particularly herbivory". Although recurring assemblages of organisms have been described by Andrew (1999) and Underwood *et al.* (1991*)* and mapped for small areas, quantifying these patterns over large areas was not possible for the broad-scale assessments. These important marine habitats are among the least studied and should be a priority for further research.

Subtidal sediment habitats

Nearshore subtidal sediments were mapped by aerial photo interpretation as described above. However, no attempt was made to classify sediment types or to delineate soft sediments beyond the nearshore zone or in estuaries. Benthic fauna is known to vary significantly with depth and grain size (Coleman *et al*. 1997) but there is little easily accessible broad-scale information on the distribution of sediments. Simplified models of shelf variation in sediment distribution were at least partially represented by ocean 'ecosystem' depth zones and for some areas there was evidence of localised patterns due to currents and sediment inputs from the larger coastal rivers (e.g. Godfrey *et al*. 1980). Further research and collation of existing data is required in this area.

DATA FOR INDIVIDUAL SPECIES AND OTHER CONSERVATION VALUES

Direct observations on the location, abundance and diversity of marine organisms range from incidental sightings, museum collections and commercial harvest data to systematic surveys designed to provide statistical estimates of abundance and variation. Where available, the latter data may provide reliable measures of biodiversity for the organisms sampled, and approximate indicators for other associated biota.

Information was available for some communities and species including surveys of estuarine vegetation (West *et al.* 1985 and recent surveys underway, R.J. Williams, NSW Fisheries, *pers. comm.*), juvenile fish biodiversity in estuaries (R.J. Williams *pers. comm*.), intertidal rocky shores (Otway 1999; Otway and Morrison, unpublished) and threatened grey nurse shark (Otway and Parker 2000). Other, less systematic data sources for species include analyses of commercial fish catch data (Pease 1999), and sightings databases kept by NSW Fisheries and the NSW National Parks and Wildlife Service.

MEASURES OF ADEQUACY – CONDITION, VULNERABILITY, RESERVE DESIGN AND MANAGEMENT

There was little direct information available on condition, threat or vulnerability for marine habitats across whole bioregions. However, data sets indicative of condition, potential threats and vulnerability were available for adjoining terrestrial areas. These included GIS maps of national parks and nature reserves, state forest, wetlands, wilderness, littoral rainforest, land capability, built-up areas, acid sulfate soils, and the Australian River and Catchment Condition Database (Stein *et al.* 2000). Indices of the percentage area of these attributes within catchments and immediate shoreline areas were calculated for estuaries and sections of coast.

The results of previous conservation assessments for wetlands (National Parks and Wildlife Service

2000), estuaries (Bell and Edwards 1980; Digby *et al.* 1998; Frances 2000; Healthy Rivers Commission 2002) and rock platforms (Griffiths 1982; Short 1995; Otway 1999) were also summarised and related to MPA identification and selection criteria along with descriptive information from coastal management plans.

Preliminary guidelines for reserve design were compiled from recent studies (Planes *et al.* 2000; Agardy 2000; Salm *et al.* 2000; Roberts and Hawkins 2000) and used in conjunction with quantitative techniques.

METHODS TO ASSESS MPA OPTIONS

A systematic approach is being used to help document alternatives, and to interpret the many criteria and sources of information used to assess options for MPAs. Methods used include summary statistics, GIS maps and spatial analyses, irreplaceability analyses, multiple-criteria analyses and reviews of literature and existing conservation plans.

Two types of spatial planning units are being used to help summarise information: fine-scale (1–4 km^2) hexagonal plan units (Fig. 7) and relatively large, broad-scale units representing whole estuaries and sections of coast and shelf (Fig. 8). The small planning units are useful for summarising local patterns, and for identifying small-scale planning options. The large planning units have been more useful for summarising broad-scale regional patterns, analysing patchy data and identifying MPA options at wider scales.

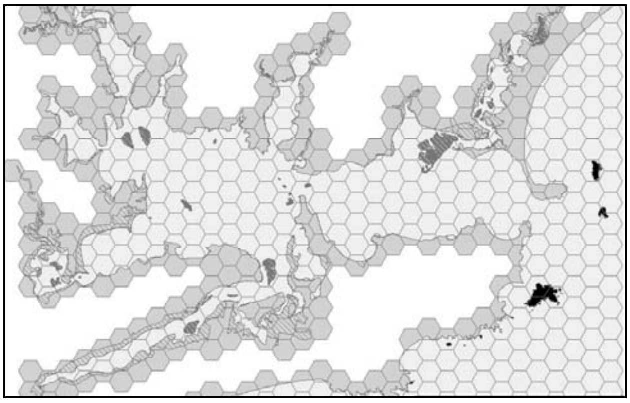

Fig. 7. Small-scale GIS planning units linked to C-Plan reserve selection software.

The NSW National Parks and Wildlife Service reserve selection software "C-Plan" (NPWS 2001) is being used to compute: 'irreplaceability' for ecosystem and habitat classes (estuary types, ocean depth zones, seagrass, mangrove, saltmarsh, rocky intertidal shore, beach, reef and islands); juvenile fish and invertebrate data (R.J. Williams, *pers. comm.*); commerical fish catch data (NSW Fisheries); bird sightings data (NSW National Parks and Wildlife); threatened species data (NSW National Parks and Wildlife and NSW Fisheries); and surveys of rock platform fauna and flora (Griffiths 1982).

The software calculates statistical estimators of 'irreplaceability' (Pressey *et al.* 1994; Ferrier *et al.* 2000) to evaluate the likelihood that a planning unit is required for representation of a range of conservation values in a reserve network (Pressey *et al* 1994). Links between C-Plan and ArcView GIS (Fig. 7) allow operators to quickly map the results of analyses and include or exclude potential sites from MPA networks while assessing the consequences of alternative decisions. The rapid display and analysis capabilities of C-Plan make it a useful tool in workshops and in exploring scenarios during decision making.

Multiple-criteria analyses (Criterium Decision Plus, InfoHarvest 2000) are then used to assess general goals as a function of the combined scores for many criteria. These methods have been applied widely in management, environmental impact assessment, fisheries (Mardle and Pascoe 1999) and in the selection and management of reserve networks (Fernandes 1996; Rothley 1997).

In marine biodiversity assessments in NSW, multiple-criteria analyses are being used to help assess comprehensiveness, adequacy and representativeness for potential MPA locations. Fig. 8 shows an example (without data, weights or resulting overall scores), of estuarine plan units in the Manning Shelf Bioregion assessed against indicators of ecosystem, habitat and species irreplaceability, habitat and ecosystem area, area for threatened and internationally important species, and the potential cost in area to competing uses.

The techniques allow for weighting of criteria, calculation of trade-offs, representation of uncertainty, sensitivity analyses of the relative influence of criteria, and the ability to combine and assess alternative models, data and sources of opinion. These techniques may be particularly valuable in achieving consensus where differences of opinion occur among experts, stakeholders and management.

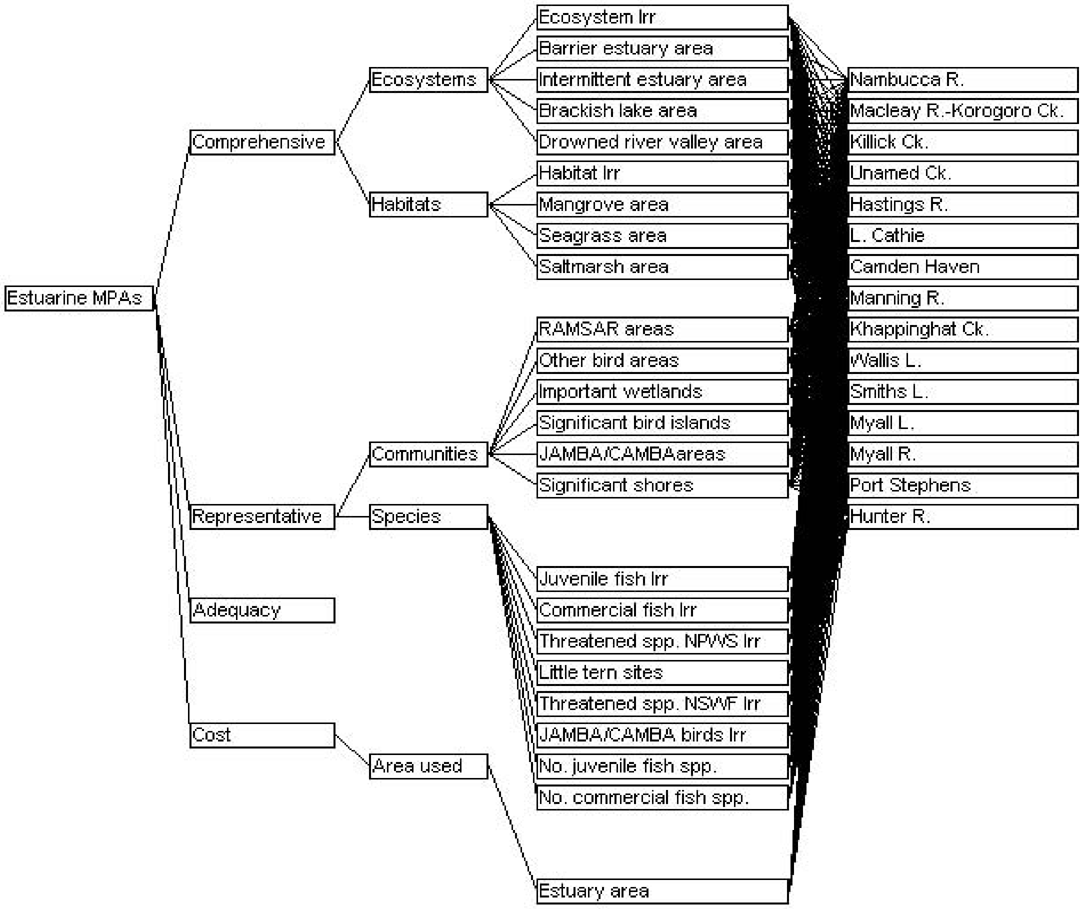

Fig. 8. Multiple-criteria analysis tree used to assess comprehensiveness and representativeness for estuaries in the Manning Shelf Bioregion (Irr=irreplaceability).

CONCLUSION

The assessments provide the basic broadscale information and methods to help plan a system of marine protected areas in NSW. Mapping for the assessments was rapid, low cost, and based largely on the modification of existing data into an appropriate GIS format. Its principal constraint was the scarcity of biological data for community and species level variation across whole bioregions, and the absence of detailed maps of subtidal substrata beyond the nearshore zone. In general, these methods reflect the overall urgency for basic data and a significant byproduct of this work is in identifying gaps in our knowledge of marine biodiversity in NSW.

Once options for MPAs are identified at the bioregional level, finer-scale data will be required for planning, management and monitoring within MPAs. The biodiversity assessments provide a framework for these studies and will assist in applying MPA objectives to ongoing planning and management.

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A FRAMEWORK FOR SYSTEMATIC MARINE RESERVE DESIGN IN SOUTH AUSTRALIA: A CASE STUDY

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Abstract

Ad hoc reserve design has been shown to produce inefficient reserve systems in terrestrial environments, limiting opportunities to achieve conservation goals. This paper presents a framework for systematic marine reserve design using South Australia as a case study. The framework consists of the reservation goals, a database of conservation features, a method for identifying conservation priorities, and measures to evaluate the performance of alternative marine reserve systems. MARXAN, a reserve selection algorithm, was used to identify marine reserve systems for three different planning scenarios. In the first scenario, MARXAN was free to either ignore or incorporate South Australia's existing marine reserves. The second scenario was required to identify marine reserve systems that built on South Australia's existing marine reserves. The third scenario gave preference to sites adjacent to South Australia's existing coastal and island reserves. Expanding on the existing marine reserves led to significantly larger marine reserve systems, reflecting the inefficiency of *ad hoc* decisions in the design of marine reserves in South Australia. Performance measures, such as efficiency, were then used to explore the consequences of spatial design requirements in alternative marine reserve systems.

Keywords systematic marine reserve design, marine reserves, conservation planning, optimization, efficiency, South Australia

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INTRODUCTION

Efficiency is an important consideration for *in situ* biodiversity conservation because of the limited availability of resources to achieve reservation goals (Pressey and Nicholls 1989; Bedward *et al.* 1992; Pressey *et al.* 1993, 1999; Pressey 1994; Freitag *et al.* 1996; Araujo 1999; Rodrigues *et al*. 1999; McDonnell *et al*. 2002). It describes the ability of a conservation planning process to represent regional biodiversity in the least number of available sites (Pressey and Nicholls 1989; Underhill 1994; Camm *et al.* 1996; Freitag *et al.* 1996; Pressey and Cowling 2001). This expression of the reserve design problem was first described by Kirkpatrick (1983) as the minimum representation problem. It is derived from the idea that whilst biodiversity conservation objectives may wish to maximize the area within the reserve system, they must compete against social, economic and management constraints (Possingham *et al.* 2000). Being efficient ensures flexibility in the future and provides the opportunity to negotiate acceptable outcomes.

Since Kirkpatrick's publication of the first reserve selection algorithm, the use of iterative algorithms to identify the minimum (or near-minimum) representation solution has been successfully applied to reserve design in terrestrial systems (Kirkpatrick 1983; Margules and Nicholls 1988; Bedward *et al.* 1992; Williams *et al.* 1996; McDonnell *et al.* 2002). Despite considerable interest, reserve selection algorithms have only recently been applied to marine reserve planning (Ward *et al.* 1999; Ardron *et al*. 2001; Ball and Possingham 2001; Beck and Odaya 2001; Leslie *et al*. 2003; Stewart *et al*. 2003). Our aim is to show how mathematical methods can be applied to identify efficient marine reserve systems. In particular, we investigate the utility of mathematical algorithms as a flexible decision-support tool to investigate options for marine reserve planning using different design constraints.

As efficiency of sampling, and performance of marine reserve systems in general, is determined largely by how the reserve design problem is framed, we begin with the development of a systematic marine reserve design framework, using South Australia as our case study. The framework consists of reservation goals, a database of conservation features, a method for identifying conservation priorities and measures to evaluate the performance of alternative marine reserve systems. We show how alternative marine reserve systems can be generated under different circumstances by applying the reserve design framework to three scenarios. The first scenario ignores the status of South Australia's existing marine reserves, the second scenario incorporates South Australia's existing marine reserves for the design of all marine reserve systems, and the third scenario preferentially selects sites adjacent to existing coastal national parks or offshore island reserves.

We consider how the efficiency of the marine reserve systems is influenced by the constraints of each scenario and, particularly, how efficiency is affected when the existing marine reserves are retained. We then evaluate the performance of marine reserve systems against secondary goals such as size and shape. To provide a context for the application of a marine reserve design framework in South Australia, we commence with a brief overview of the status of marine reserve planning in South Australia.

Status of South Australia's marine reserve system

South Australia is located on the temperate coast of Australia in a region that has been geographically and climatically isolated for around 65 million years. It features some of the highest levels of marine endemism in Australia and the world, with 90–95% of known species endemic or of restricted range (IMCRA Technical Group 1997; Edyvane 1999). The general consensus is that marine reserve planning in South Australia has been *ad hoc* and is inadequate to meet present conservation objectives (Government of South Australia 1998). A recent report listed 15 marine protected areas amounting to almost 60,000 hectares (Edyvane 1999). This represents 0.9% of the State waters and a contribution of less than 0.2% to the national total. The establishment of the Great Australian Bight Marine Park has since increased this value to around 4.5% of State waters, however, a considerably smaller amount is dedicated to no-take areas.

In recognizing the inadequacies of the existing system, the South Australian government has announced its intention to establish a system of multiple-use marine protected areas. Their proposed goal is "to maintain the long term ecological viability and processes of marine and estuarine systems, and contribute to ecologically sustainable development" (Government of South Australia 2001). In addition, they carry a responsibility to contribute to the primary goal of the National Representative System of Marine Protected Areas, "...to establish and manage a comprehensive, adequate and representative system of MPAs…" (marine protected areas) (ANZECC 1999).

METHODS FOR ESTABLISHING A MARINE RESERVE DESIGN FRAMEWORK FOR SOUTH AUSTRALIA

System goals and objectives

Mathematical approaches require a clear statement of the conservation objective in order to inform how reservation will proceed. For example, a marine reserve system that emphasizes the protection of charismatic, rare and threatened species will differ from a reserve system that aims to maximize representation of marine biodiversity. In this paper, our goal is to identify marine reserve systems that are as representative of biodiversity as possible (Ballantine 1991; Agardy 1994; Kelleher 1997; ANZECC 1999; Pressey and Cowling 2001).

As biodiversity is still a vague concept for which there is no simple, comprehensive and fully operational definition (Noss 1990), our first task is to establish operational definitions for biodiversity. For both marine and terrestrial systems this is hampered by limited information and poor understanding of ecological processes (Pressey and McNeill 1996). As many aspects of reserve design are subject to uncertainty, existing scientific information has been considered, in order to identify some general principles of reserve design (Salm 1984; Ballantine 1991; Pressey et al. 1993). Reserve design theory supports a system-based approach with replication and larger rather than smaller reserves. Ideally, reserves would be situated at 'source' locations in an arrangement that supports positive ecosystem linkages (Pulliam and Danielson 1991). Reserve size would take into consideration the long-term viability of populations and communities, with provision for catastrophes (Allison et al. 2003).

The objective and goal for our analyses is to design marine reserve systems that are representative of chosen surrogates for biodiversity, but use the least number of sites. Representation targets for all biodiversity surrogates were set at 10% of the total amount of each conservation feature. In this study, explicit rules for replication and minimum patch size were ignored.

Database

The lesson for marine reserve planning is not to be constrained by single indices for which data are available, but where possible to recognize and provide for the inherent complexities of marine systems and their functions by representing as many features as possible. This is feasible when using reserve selection algorithms, which guarantee that representation targets will be met. The critical issue then is whether the biodiversity surrogates used to identify representative marine reserve systems can ensure that systems are comprehensive as well. This calls for the identification of surrogates at an appropriate level of organization (Noss 1990) and, quite possibly, at multiple levels of organization.

To capture patterns of biodiversity, we proceeded with a hierarchical approach that focused on providing a consistent framework for conservation planning (Noss 1990, Pressey *et al.* 2000). In this method, the basic unit of biodiversity (species) is generalized into more heterogeneous classes such as ecological communities or species assemblages (Margules and Usher 1981). However, the more generalized classes become, the less confident we can be that reserve systems are comprehensive, for the variability within a class is ignored. We can attempt to counter this effect by increasing the level of representation for each conservation feature (i.e. total amount) and replication (i.e. multiple occurrences).

We outline the conservation features used as biodiversity surrogates for the design of representative marine reserve systems for South Australian state waters. Conservation features were identified from 6 data layers obtained from state government agencies: the Department for Environment and Heritage, and the Department for Primary Industries, South Australia. An additional feature class was delineated to represent the status of South Australia's existing reserves.

Biogeographic regions (meso-scale 100–1000s of kilometres)

Biogeographic regions have been identified at the national level in a marine ecosystem-based classification scheme, known as the Interim Marine and Coastal Regionalisation of Australia (IMCRA). The classification provides a scientific basis for reporting on the adequacy of marine ecosystem representation in the national system of marine reserves. It was derived from a combination of expert field ecological knowledge and interpretation of existing regionalizations (IMCRA Technical Group 1997). These include sea-floor topography, sea-floor sediments, physical characterics of the water column , coastal geomorphology, and pelagic and demersal fish regionalizations. Each bioregion comprises a cluster of interacting ecosystems that are repeated in similar form throughout. There are 60 bioregions delineated for the Australian coastal and offshore waters, and 8 of these occur wholly or partly within South Australian state waters.

Biounits (micro-scale 10–100s of kilometres)

At this micro-scale level, distinct regional and local variations in habitat and biodiversity occur. These variations are classified on the basis of local-scale ecological units (e.g. rocky shores, shoals or reef systems) and information on the spatial extent of these units. South Australia's micro-scale biounits were defined primarily on the basis of coastal physiography, topography and major marine physical habitat or seascape features, as well as habitat distribution (Edyvane 1999b). In total, 35 biounits have been identified and comprise 30 coastal biounits and 5 offshore units (Edyvane 1999b). For the coastal biounits, seaward boundaries were bound by the 30 m depth contour. For the offshore biounits, boundaries were bound by the 50 m depth contour. Because biounits are not always nested within the meso-scale bioregions, bioregions and biounits were each treated as a unique conservation feature classes.

Marine benthic habitat maps

The idea of habitat representation is based on the notion that by conserving all habitats, the maximum number of species will be represented, including species not used to define the habitats (Margules and Nicholls 1989). Ward *et al.* (1999) concluded that habitat-level surrogates can be used effectively to delineate reserves for conserving marine biodiversity. Benthic habitat coverage has been mapped for coastal and gulf waters of the state. Habitats were identified by tracing discernible underwater features on satellite images. Aerial photographs were used for 'truthing'. The resulting dataset uses biological data for classification of seagrass densities and geomorphological descriptions for reefs. As a benthic classification scheme has not yet been developed for South Australia, habitat type is classified at a broad level (i.e. seagrass, platform reef, sand) and does not incorporate information on the dominant species assemblages. We identified 6 unique conservation features in this feature class.

Coastal saltmarsh and mangrove habitats

Intertidal vegetation of the coastal regions of South Australia has been identified from digitized 1:40000, 1:15000 and 1:10000 non-rectified aerial photography. Habitats were classified and coded using landform, life form and condition categories. Classification was based on aerial photo interpretation, survey data, ground truthing and expert knowledge. In total, 65 habitat classes were identified, with 11 of these relating to intertidal or tidal areas. To approximate a similar scale of resolution to that of the marine benthic habitat categories identified above, these classes were collapsed into the more generalized categories of intertidal/tidal marine algae; intertidal and tidal bare sand; intertidal/tidal seagrass, mangroves and saltmarsh. This classification provided a further 5 unique conservation features.

Species occurrence

Species occurrence data were incorporated where coverage extended across the study area and occurrence records were of sufficient quality. On these grounds, we identified 3 conservation features from the data layers available for South Australian distributions of seabirds, Australian sea lions and New Zealand fur seals.

Seabirds of South Australia – This data set describes population and breeding seasons, including mainland and offshore island sites. The dataset is suitable for the identification of significant seabird habitat communities within South Australia.

Australian sea lions and New Zealand fur seals within South Australian waters – sea lion *(Neophoca cinerea)* and fur seal (*Artocephalus forsteri*) locations are identified, with population, breeding season, and breeding and haul-out sites for the mainland and for island locations. The dataset is suitable to identify significant breeding and haul-out sites for habitat conservation purposes.

Bathymetry

This dataset is a compilation of water depths for the offshore waters of South Australia, with depth values representing depths to the seabed. Data were collected as part of the National Mapping Bathymetric Program, which was designed to provide generalized detail of the seabed of the Continental Shelf. Depth values were delineated into the following categories, <10 m; 10<20 m; 20< 30 m; 30<40 m; 40<50 m and 50+ m. By seeking proportional representation of each depth category, we aim to achieve representation of offshore environments, where biological data are often limited. We therefore devised a constraint requiring representation of each depth category within each of the 8 bioregions. This provided an additional 40 conservation features.

Protected areas

This data layer provides an accurate location for the legal boundaries of both terrestrial and marine reserves and is used to formulate marine reserve planning scenarios. In South Australia, different legislation can be used to designate marine protected areas (marine reserves), and the level of protection afforded to a reserve can vary widely (i.e. netting closures, marine sanctuaries and multiple-use marine parks). We generated a user-defined marine reserve theme as a subset of protected areas derived from the Australian Collaborative Protected Area Database. It includes areas recognized as marine reserves, as well as coastal/offshore island reserves with significant marine components. Rocklobster sanctuaries and netting closures were not included. Consequently, in this paper South Australia's existing marine reserves comprised 21 marine reserves with a total area of approximately 2880 km^2.

Conservation features

The conservation features are what our reserve system will attempt to cover. They are recorded as either distributions (i.e. coverage) or abundance (i.e. species abundance). Our database encompassed 102 conservation features identified from the 6 biophysical data layers described above.

The database was formatted using a series of geographical information system (GIS) processing steps to define the planning units (i.e. the basic selection units) and create suitable data files (Stewart *et al*. 2003). For the analyses, we created a grid extending west to east from the Western Australian border to the Victorian border and north to south encompassing all South Australian state waters. This process delineated 3119 planning units, with each planning unit a 5 km by 5 km cell. Owing to the irregular shape of the study area, a number of planning units were truncated at the coastline and offshore islands, providing some size variation across the study area. Information on the amount of each conservation feature within individual planning units is then compiled. The amount of each conservation feature j in each planning unit i formed the basic data matrix a_{ij}. With 102 conservation features and 3119 planning units, 19597 occurrences were recorded in all.

Mathematical methods for systematic marine reserve design

Reserve selection algorithms differ from traditional marine reserve selection methods in the way they define the reserve selection problem. At the core of the problem is the goal of minimizing the area encompassed in the reserve network, described by Kirkpatrick (1983) as the minimum representation problem and formulated by Possingham *et al.* (2000) as a non-linear integer programming problem:

minimize the objective function:

$$\sum_{i=1}^{M} c_i x_i + BLM\left(\sum_{i=1}^{M} x_i l_i - 0.5\sum_{i=1}^{M} x_i \sum_{k=1}^{M} x_k b_{ik}\right) \quad (1)$$

subject to the constraints:

$$\sum_{i=1}^{M} a_{ij} x_i \geq t_j \sum_{i=1}^{M} a_{ij} \text{ for all } j=1..N,$$

$$x_i \in \{0,1\} \text{ for all } i = 1..M, \quad (2)$$

where x_i are the control variables such that if x_i is 1 then site *i* is selected in the reserve system and if x_i is 0 then site *i* is not in the reserve system; c_i is the "cost" of site *i*. In this paper, all planning units have an equal cost of 1, such that the overall reserve system cost is expressed as the total number of planning units in the marine reserve system. The parameter l_i is the perimeter or boundary length of site *i*, and b_{ik} is the common boundary length of sites *i* and *k*. *BLM* is the boundary length modifier variable that controls the weight given to the boundary length. It allows spatial design requirements to be incorporated by determining the relative importance placed on minimizing the boundary length relative to minimizing area. When BLM is very small then the algorithm will concentrate on minimizing area, whereas if BLM is relatively large then there is greater emphasis on minimizing the boundary length and more spatially compact reserve systems are configured.

Expression 1 is our objective: minimize a linear combination of the reserve system cost and it's boundary length (the length of the border between selected and unselected planning units). Expression 2 is a set of constraints that ensures that the target for each conservation feature is met, where a_{ij} is the abundance of the feature type *j* in site *i*, and t_j sets the target fraction for each feature (in this paper we assume t_j to be 10% for all *j*). There are *N* different conservation features spread across *M* different sites. A feasible solution is one that selects a set of sites (using the control variables x_i), whilst ensuring that the specified level of representation for each conservation feature is met.

MARXAN reserve selection algorithm

MARXAN, a tool for marine reserve design (Ball 2000; Ball and Possingham 2000) is based on terrestrial reserve design software. The software is freely available and can be downloaded from www.ecology.uq.edu.au.

Because our reserve design problem is large (2^{3119} possible marine reserve systems), it is virtually impossible to find an optimal solution in reasonable time. MARXAN provides an alternative method, using optimization algorithms to identify reasonably good solutions, which are assigned an objective function score. In this paper, we use the simulated annealing algorithm with iterative improvement to select planning units that satisfy a set target of ecological, spatial, social and economic criteria. A planning unit is randomly added to the reserve system and the change to the system is evaluated. The planning unit is then added or removed, depending on the evaluation. This process continues for a set number of iterations and has the advantage of allowing the reserve system to move temporarily through sub-optimal solutions space, increasing the number of routes by which the global minimum might be reached (Possingham *et al.* 2000). This method generates marine reserve systems that can have identical or very similar objective function scores but with different configurations. By repeating the selection process (simulation), MARXAN can identify a range of reasonably good solutions to the same problem.

Generating alternative solutions for marine reserve design

Our analyses consider the effect of two variable factors: the reserve design constraints (*No Reserves*; *Reserves Fixed; Reserves Free*) and the boundary length modifier (set at values of 0, 0.1, 0.5 and 1.0). In total, this provided 12 design problems. For each problem, 10 replicate simulations were performed, with each simulation comprising 1000 runs. This generated a total of 10,000 alternative marine reserve systems.

Reserve design scenarios

Three planning scenarios were devised for the design of marine reserve systems in South Australia, with each being formulated in mathematical terms using expression (1) and (2) described above.

The "No Reserves" scenario follows the problem defined by expressions (1) and (2). The number of conservation features *N* is 102, the number of sites *M* is 3119. As we ignore the status of South Australia's existing marine reserves, our control variable x_i can assume a value of either 0 or 1 for all 3119 planning units. If $x_i = 1$ then that planning unit forms part of the reserve system and if $x_i = 0$, planning unit *i* is excluded from the reserve system. We set the cost variable c_i to 1, which means that every site has equal cost. The parameter l_i is the boundary length (km) of planning unit i, and b_{ik} is the common boundary length (km) of planning unit i and k. This delivers a benefit to planning units that share a common boundary. Conservation feature targets are set to a target fraction of their regional coverage, t_j, in this case at 0.1 (10%). Lastly, the abundance of the conservation feature type j in planning unit i is depicted by the variable, a_{ij}.

The "Reserves Fixed" scenario alters the 'status' of individual planning units by locking in the sample of planning units that represent South Australia's existing reserves. We revisit expressions (1) and (2) to formulate the problem, where we maximize the objective function and constraints as before but with $x_i = 1$ for all sites that are in the existing reserves. This amounts to 288 planning units that represent South Australia's existing reserves. We adopt the matrix defined for the *No Reserves* scenario with minor amendment, because there are now fewer sites available for selection (the existing reserves are all locked-in). Accordingly, our control variable x_i now assumes a value of either 0 or 1 for only 2831 planning units. The problem is to add to the existing reserves until the conservation targets are met.

The "Reserves Free" scenario uses the same matrix as defined for the *No Reserves* scenario, except that information on adjacent land types is incorporated to modify boundary length cost. So where planning unit i is adjacent to an existing coastal or marine reserve, the boundary length parameter l_i (km) of planning unit i is set to 0. This scenario assumes that the adjacent land type makes some contribution towards the reserve system goals such as reduced management costs, which may be the case when marine reserves abut coastal reserves.

Performance measures

For each marine reserve system generated, MARXAN generates summary data, which includes the objective function score, the number of planning units and the total boundary length. The best (near-minimum) marine reserve system configuration is identified as the one with the lowest objective function score (from a total of 10,000 alternative solutions). These are then mapped in a geographical information system (GIS) platform and the number of individual reserves and total area of the best marine reserve system calculated. Because solutions with the lowest objective function score may not always be the preferred marine reserve system when other constraints are considered, we employ alternative measures to evaluate differences among marine reserve systems generated for our reserve design problems.

To assess how efficiently representation targets are being met, we use Pressey's (Pressey and Nicholls 1989) measure of *efficiency*. It varies from 0 to 1, with 1 being the most efficient solution.

$$E = 1 - X/T \tag{3}$$

where E is efficiency, X is the number of planning units needed to meet the constraints, and T is the total number of planning units.

As we are also interested in the spatial configuration of the reserve system, we provide *compactness* as a measure of the ratio of the reserve system boundary length to the circumference of a circle with the same area (the theoretical minimum). As values approach 1 the solution becomes more compact, and as values increase solutions become more fragmented.

$$\text{Ratio} = \frac{\text{Boundary Length}}{2\sqrt{\pi \times \text{Area}}} \tag{4}$$

The effect of spatial clustering on the reserve system perimeter, area, compactness and marine reserve system combination size (number of planning units in a reserve system) was examined by using different values for the BLM. Analyses were performed on the best marine reserve system for each different reserve design problem.

As there are many possible marine reserve systems, it is also useful to know something about the relative importance of individual planning units for conservation planning. We use selection frequency counts, otherwise referred to as 'summed irreplaceability', to provide a measure of the contribution of any one planning unit to the reservation goals (Pressey *et al.* 1994; Ball and Possingham 2001; Leslie *et al.* 2003; Stewart *et al. 2003*). It assumes that the relative value of a planning unit increases the more times it is selected. In this paper, we use an approach defined by Stewart *et al.* (2003) that uses the mean combination size (Ferrier *et al.* 2000) to determine the probability, p, that any planning unit is chosen by random. This allows us to determine whether a planning unit is selected by MARXAN more than randomly, by comparing it's selection frequency with that expected for a binomial distribution with probability, p, of success over 10,000 trials. By identifying Tukey's 95% confidence interval around the binomial distribution, we can assert that a planning unit selected more often than the upper 95% confidence limit, was included in the reserve system more than could be expected from chance alone and thus could be considered to be irreplaceable.

RESULTS AND DISCUSSION

No spatial clustering

Setting the BLM to 0 is equivalent to having no spatial clustering effect. Marine reserve systems configured under these conditions delivered the lowest objective function score and produced the most efficient solutions within their respective design scenarios (Table 1). In contrast, the compactness of these solutions was very poor, as marine reserve systems with no spatial clustering requirements were highly fragmented, and consisted of a large number of small individual reserves (Table 1). The configuration of marine reserve systems with and without spatial clustering is shown in Fig. 1. From a management perspective, the feasibility of reserve systems that ignore spatial clustering may be questionable, and this suggests that marine reserve systems should not be selected on the grounds of efficiency of sampling alone.

The number of planning units which form the reserve system varied across the three planning scenarios, when spatial clustering was ignored (BLM = 0). Results indicate that nearly twice the number of planning units were required in the *Reserves Fixed* scenario to achieve the same representation targets as for the *No Reserves* and *Reserves Free* solutions. We discuss reasons for the inefficiency of the *Reserves Fixed* scenario when we consider the effect of the design scenarios below.

Spatial clustering

More compact, yet less efficient reserve systems are identified, when spatial design requirements are incorporated in the reserve planning problem. This is because of the requirement to minimize the perimeter (i.e. boundary length) of the reserve system such that planning units become clustered to configure more compact solutions. The trade-off if that a greater number of planning units are required to achieve the reservation goals (Table 1). This highlights the trade-off between the area and perimeter of a marine reserve system, and between efficiency and compactness, when representation targets remain the same. For example, in the Reserves Free scenarios, increasing the boundary length modifier from 0 to 1 lead to a larger, more compact marine reserve system with fewer individual reserves (Fig. 1). Arguably, the small gain in area (approximately 7%) is an acceptable trade-off for fewer, larger and possibly more viable marine reserves.

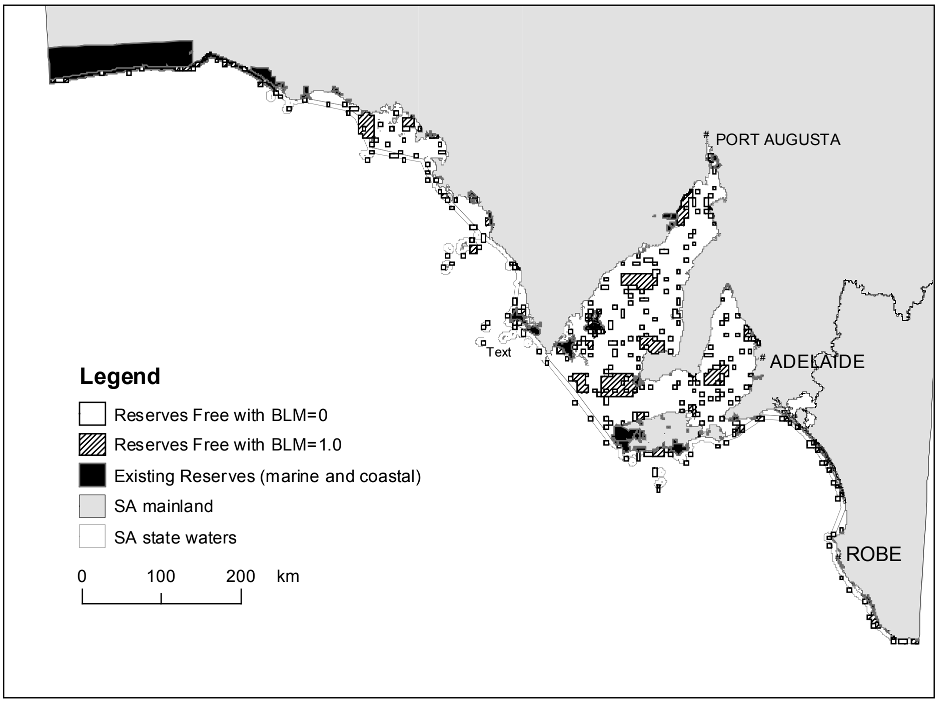

Fig. 1. Effect of the boundary length modifier (BLM) is shown for two alternative marine reserve systems with the same reservation goals. The BLM enables the planner to control the level of fragmentation in a reserve system, by including a requirement for spatial clustering among planning units. No spatial clustering (BLM=0) leads to a highly fragmented reserve system, comprising many individual reserves, compared to the reserve system which includes spatial clustering as part of the design constraint (BLM=1). The marine reserve systems are generated for the *Reserves Free* scenario, which provides an additional benefit to reserves adjacent to South Australia's existing coastal and marine reserves (cross hatched).

Design scenarios

Inspection of the summary data (Table 1), and the reserve system combination size in particular, suggests that the performance of the best marine reserve systems are influenced by factors other than spatial clustering (i.e. BLM values). Analyses indicate that this variability is due to the unique constraints of our three design scenarios. Single-factor analysis of variance reported a significant difference among the mean combination size ($p<0.05$) for the *No Reserves*, *Reserves Fixed* and *Free Reserves* scenarios. Multiple-comparison procedures concluded that for all BLM values, the observed difference was between the mean combination size of the *Reserves Fixed* and the *No Reserves/Free Reserves* scenarios. Clearly, this provides evidence that constraints of the design scenarios have a variable effect on the number of planning units selected and therefore on the efficiency of the marine reserve system solutions. This in turn influences the size, shape and connectivity of the marine reserve systems generated.

For example, marine reserve systems identified for the *No Reserves* and *Reserves Free* scenarios were always more efficient than the *Reserves Fixed* system at the corresponding BLM value. Indeed, the *No Reserve* systems achieved the same representation targets in approximately 66% of the area required by *Reserve Fixed* systems. For the *Free Reserves* scenario, this value varies between 55% and 60% of the overall area of the *Reserves Fixed* systems. So if instead, our representation goal were to maximize the representation target within a given amount of area, then the *No Reserves* and *Reserves Free* scenarios would be the more efficient systems. As it is, they achieve the same representation targets at a much-reduced cost. We conclude that the existing marine reserve system does not efficiently contribute to the reservation goals identified here. This is perhaps, not so surprising, given that South Australia's existing marine reserve system has been driven by objectives other than representativeness and comprehensiveness. Locking-in these areas exerts a significant constraint on expansion of the system, since areas are added in a way that best complements the existing values.

Table 1. Summary data for the best marine reserve systems generated for No Reserves, Reserves Fixed, and Reserves Free scenarios, with different values for the boundary length modifer (BLM). Performance measures include: the objective function score; the number of planning units in the reserve system; the total perimeter of the reserve system; the reserve system area and the number of individual reserves that are contained within the reserve system. Compactness measures the ratio of the perimeter of the reserve system to the perimeter of a circle with the same area, therefore a value of 1 is the ideal (most compact) configuration. Efficiency is measured as 1 – (No. sites selected / total number of sites) (Pressey, 1989), therefore values are bound between 0 and 1, with 1 being the most efficient system. The combination size is the no. of planning units in the best reserve system (calculated from 10 simulations of 1000 runs each). SD = standard deviation

Scenario	BLM	Score	No. Planning Units	Perimeter (km)	Area (km^2)	No. Individual Reserves	Compactness	Efficiency	Combination Size mean +/- sd
No Reserves	0	276.1	273	5120	6767	233	17.6	0.912	273.9 +/- 0.32
	0.1	519.1	284	2263	6922	48	7.7	0.909	283.3 +/- 1.16
	0.5	1246.7	307	1823	7302	30	6.0	0.902	306.7 +/- 5.12
	1.0	2128.8	322	1800	7518	32	5.9	0.897	317.6 +/- 6.60
Reserves Fixed	0	497.0	497	6192	10568	192	17.0	0.841	497.8 +/- 0.63
	0.1	970.2	504	4545	10689	165	12.4	0.838	504.9 +/- 3.11
	0.5	2443.0	524	3725	11089	44	10.0	0.832	524.4 +/- 4.40
	1.0	4274.0	536	3628	11357	39	9.6	0.828	541.0 +/- 6.94
Reserves Free	0	275.8	274	5127	6784	235	17.6	0.912	273.9 +/- 0.88
	0.1	451.2	286	2742	6867	73	9.3	0.908	286.9 +/- 2.13
	0.5	934.5	317	2721	7251	67	9.0	0.898	310.8 +/- 5.31
	1.0	1529.0	314	2596	7314	65	8.6	0.899	322.8 +/- 7.16

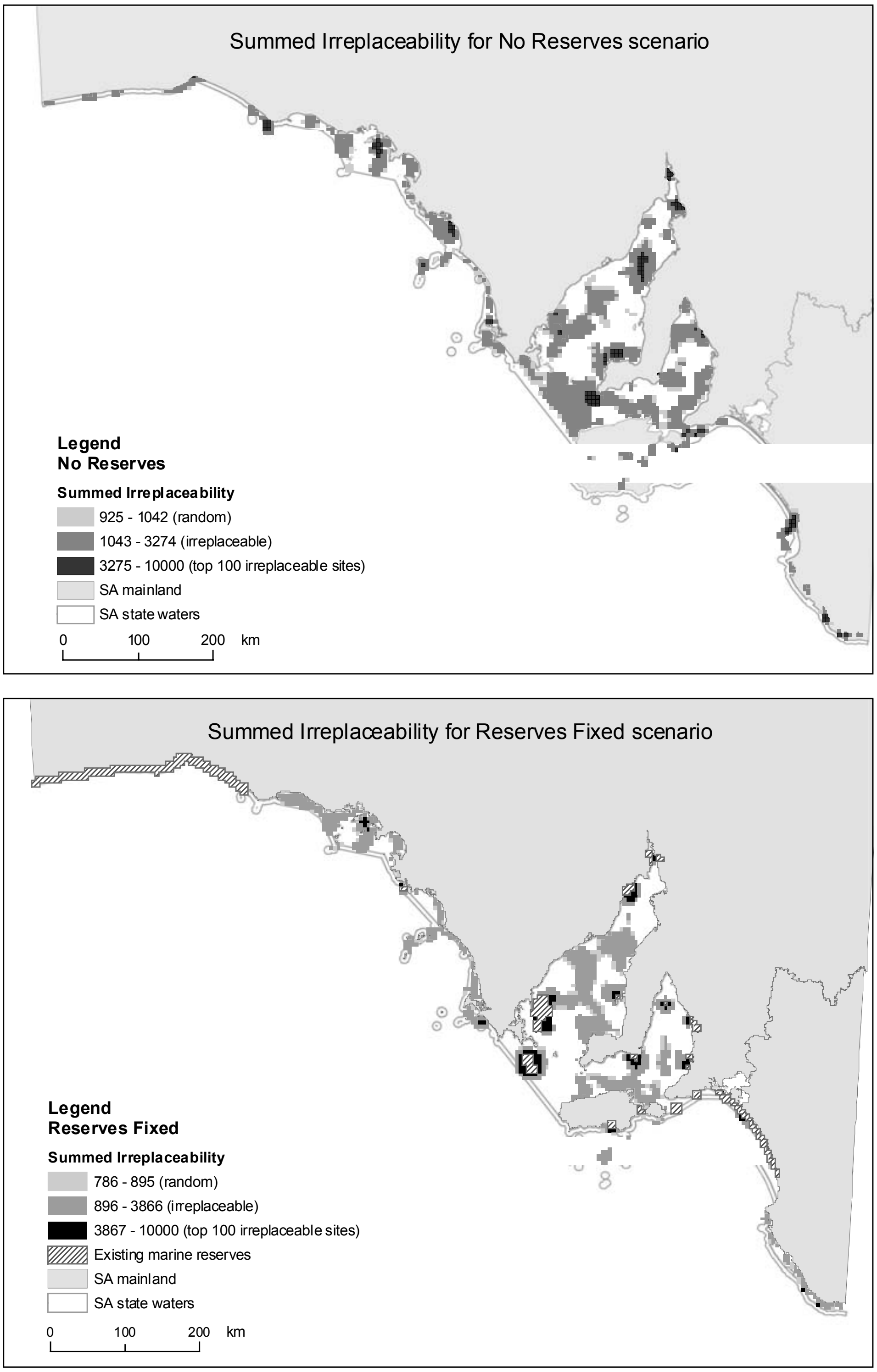

Fig. 2. Summed irreplaceability for marine reserve systems under *No Reserves* and *Reserves Fixed* scenarios with the boundary length modifier set to 0.5. A planning unit's contribution to the reserve system goals are measured as a summed irreplaceability value, derived from selection frequency counts; it assumes that the more times a site is selected, the more valuable it is for reserve planning. Shading denotes planning units that are selected more than could be expected from random sampling. A planning unit's irreplaceability value varies according to how the reserve design problem is framed. Locking-in South Australia's existing marine reserves (cross-hatched) as a design constraint (*Reserves Fixed),* leads to the reserve system being established around this initial seed. So, planning units in close proximity to the existing reserves have a high summed irreplaceability value, compared to the *No Reserves* scenario, when the existing reserves are ignored.

Next, we examined the *Reserves Free* scenario with emphasis on spatial clustering (BLM is set to a value greater than 0). This design scenario generated marine reserve systems that had a greater number of individual reserves, were more fragmented, and with a larger overall perimeter than in the *No Reserves* scenario (Table 1). However, the score and number of planning units of marine reserve systems were similar for both scenarios. This suggests that individual reserves contained in the *Reserves Free* systems were often adjacent to the existing coastal/marine reserves. Since planning units adjacent to these areas received a benefit in the form of a free boundary length, a portion of the overall perimeter does not incur boundary length costs. So although the *Reserves Free* solutions are not as compact as marine reserve systems generated in the *No Reserves* scenario, they do provide feasible alternatives at minimum cost. We conclude that compactness may not always be a good measure of performance of the marine reserve system in this planning scenario because it does not account for benefits that arise from the spatial arrangement of marine reserves adjacent to coastal reserves. With approximately one-third of the South Australian coast under some form of protected area management, the *Reserves Free* solutions suggest that there is sufficient flexibility to create efficient marine reserve systems that are adjacent to SA's existing coastal reserves.

Summed irreplaceability – identifying conservation priorities

We have shown how mathematical methods can be used as a flexible tool to explore the consequences of alternative reserve design problems. Here, we focus on their role as a tool for identifying conservation priorities. The non-unique occurrence of many indices of biodiversity means there is often more than one way to achieve our goals (Possingham *et al.* 2000). However, it follows that some planning units are likely to make a more valuable contribution than others and so the options for replacing a planning unit with an alternative site may be much reduced (Pressey 1993).

Selection frequency counts were reported for planning units across the planning region and compared with the 95% confidence interval of the predicted probability distributions to determine their relative conservation value. Fig. 2 presents results for the *No Reserves* and *Reserves Fixed* scenarios, with a BLM value of 0.5. Areas of high conservation priority are mapped according to their summed irreplaceability values. The different values identify planning units selected less than could be expected from chance; those selected as often as could be expected by chance; and those that were selected more than could be expected by chance. Using this technique, we evaluate the importance of individual planning units according to how often they are selected in a marine reserve system. This is clearly a useful measure to assist identification of core areas that attain some critical value for marine reserve system design and for regional planning at a broader scale.

CONCLUSION

We have shown how a properly posed design problem and mathematical methods can be used to investigate the implications of alternative planning scenarios for a marine reserve system. In particular, we show how design tools can be used to evaluate alternative options in a way that can demonstrate the trade-offs that result from different constraints. Using South Australia's existing marine reserve system, we have illustrated how selection of reserves in an *ad hoc* manner can have significant implications for the design of representative marine reserve systems; it was less efficient to establish reserve systems around these pre-existing areas than to ignore them. Although we don't expect reserve planners to abandon existing marine reserves on the basis of results shown here, we emphasize that past and present decisions have great effect on the design of marine reserve systems. Overall, mathematical algorithms provide a means to consider different variations to the reserve design problem and have an important role in supporting informed decisions.

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A FIRST STEP TOWARD BROAD-SCALE IDENTIFICATION OF FRESHWATER PROTECTED AREAS FOR PACIFIC SALMON AND TROUT IN OREGON, USA

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Abstract

Decision makers, concerned with Pacific salmon and trout, must often select freshwater areas to protect or restore based on only site-scale information. In response, the Coastal Landscape Analysis and Modeling Study (CLAMS) has developed broad-scale models based on topographic features of watersheds to assess potential use by steelhead (*Oncorhynchus mykiss*) or by coho salmon (*O. kisutch*). The modeled attribute, termed intrinsic potential, was expressed for each species as the geometric mean of classified channel gradient, valley constraint, and mean annual discharge. These components were derived from 10-m Digital Elevation Models (DEMs) for all streams in two large basins in the Coastal Province of Oregon, USA. Because the types of topographic features associated with steelhead and coho salmon differ, stream reaches with high intrinsic potential (values ≥ 0.8) for these two species generally did not overlap. Streamside areas adjacent to reaches with high intrinsic potential were characterized relative to land ownership and use. High-intrinsic-potential reaches typically occurred on publicly owned forestlands for steelhead and on privately owned lands with various uses for coho salmon.

Results are relevant in describing the likelihood of finding unimpaired habitat in high-intrinsic-potential reaches for these species and in assessing the feasibility of conservation options, thus in identifying freshwater protected areas. Findings for steelhead and coho salmon in the study basins suggest how the approach and developed models might be applied to other aquatic species for which links to topographic features are known or scaled-up to aid in regional prioritization of reaches or watersheds as protected areas. Tailoring actions to the intrinsic potential of an area should enhance the efficacy and efficiency of broad-scale freshwater conservation strategies so may improve their societal support.

Keywords: costal landscape analysis and modeling study, freshwater protected areas, coho salmon, steelhead, channel gradient

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INTRODUCTION

Salmon and trout (*Oncorhynchus* spp.) are integral components of ecosystems in the Pacific northwestern United States of America (Willson and Halupka 1995). Adults of the anadromous forms link the marine and terrestrial environments by returning essential resources to the relatively nutrient-poor streams in which they spawn. Juveniles complete the freshwater phase of their life history in rivers, rearing in all parts of the network from headwaters to estuaries. Salmon and trout in this region are also commercially, recreationally, and culturally important.

Many populations of Pacific salmonids are considered at risk (Nehlsen *et al.* 1991), and some of these have been listed under the United States *Endangered Species Act 1973*. A variety of factors may contribute to declining fish abundances (National Resource Council 1996). Regularly included among these are loss and degradation of freshwater habitats from human activities. Consequently, habitat protection and restoration are common objectives of salmonid conservation strategies (e.g. USDA and USDI 1994; State of Oregon 1997). Measures to protect and restore freshwater habitats are often perceived to conflict with the goal of maximizing profits from land-use. Thus, tools that can focus salmonid recovery efforts by identifying locations with the greatest potential to yield conservation benefits should hold value for policy makers, regulators, and land managers.

Stream reaches with significance to salmonid conservation can be distinguished, in part, by their topographic characteristics. Specific landforms may affect the capacity of reaches to develop high-quality habitat (Frissell 1992;

Montgomery and Buffington 1997) and, ultimately, to support salmonids. Different species of salmonids have been associated with particular landform types. For example, in coastal Oregon, USA, juvenile steelhead (*O. mykiss*) dominated high-gradient stream reaches constrained by adjacent hill slopes (Burnett 2001). However, juvenile chinook salmon (*O. tschawytscha*) and coho salmon (*O. kisutch*) in these same streams were observed primarily in unconstrained reaches (Burnett 2001). In western Washington, USA, channels with lower gradients contained greater numbers of coho salmon adults returning to spawn (Pess *et al.* 2002) and of coho salmon smolts migrating to the ocean (Sharma and Hilborn 2001). Consequently, a regional conservation strategy aimed at protecting and restoring the most topographically favorable stream reaches for a particular salmon or trout population can logically prioritize limited funds and improve the likelihood of success.

This research is intended to develop and demonstrate tools for identifying topographic characteristics associated with use by aquatic species. Specifically, the potential of all stream reaches to support steelhead or coho salmon was characterized and mapped in two major coastal Oregon drainages, the Tillamook Bay and Nestucca River basins. Additionally, the distribution of stream reaches with the highest potentials for each species was examined relative to land ownership and use. As is true throughout the region, many coastal Oregon salmonid populations have decreased in number, with freshwater habitat loss and degradation being suggested as important causes (Nehlsen *et al.* 1991; Nickelson *et al.* 1992). Much historical evidence indicates that large salmon runs (e.g. coho salmon, Lichatowich 1989) were maintained in coastal Oregon streams. However, neither these streams, generally, nor those in the Tillamook Bay and Nestucca River basins, specifically, have been comprehensively evaluated for the potential to support salmonids.

STUDY AREA

The Tillamook Bay and Nestucca River basins drain westward and comprise approximately 2300 km² of the Coastal Province of Oregon (Fig. 1).

These basins support five of the seven species of anadromous trout and salmon occurring in the Pacific Northwest (steelhead, cutthroat trout (*O. clarki*), chinook salmon, coho salmon, and chum salmon (*O. keta*). The climate is temperate maritime with restricted diurnal and seasonal temperature fluctuations; mean annual temperatures range from approximately 1°C in January to 15°C in August (Daly *et al.* 1994). Most of the 300 cm of annual precipitation arrives between September and May, principally as rainfall. Peak stream flows are flashy following winter rainstorms rather than associated with spring snow melt, and base flows occur between July and October.

Fig. 1. Location of the Tillamook Bay and Nestucca River basins in the Coastal Landscape Analysis and Modeling Study (CLAMS) area of western Oregon, USA.

The study area is underlain primarily by sandstone and basalt formations, and except for a few interior river valleys and a prominent coastal plain, is dominated by mountains (Orr *et al.* 1992). Elevations range from sea level to approximately 1100 m. Uplands are highly dissected with drainage densities up to 5.0 km/km² (FEMAT 1993). Montane areas are predominately in conifer and broadleaf forests that include tree species of Douglas fir (*Pseudotsuga menziesii*), western hemlock (*Tsuga heterophylla*), and red alder (*Alnus rubra*). Western red cedar (*Thuja plicata*) and big leaf maple (*Acer macrophyllum*) are typical additions in riparian areas. Forests span early successional to old-growth seral stages as a result of a disturbance regime driven by timber harvest and by infrequent, intense wild fires and windstorms (Franklin and Dyrness 1988). Using data from Ohmann and Gregory (2002), we determined that approximately 7% of the original coastal temperate rainforest remains in these basins.

METHODS

Unless otherwise noted, digital data layers were developed for the Coastal Landscape Analysis and Modeling Study (CLAMS) (Spies *et al.* 2002). Each data layer was clipped to the drainage boundaries of the Tillamook Bay and Nestucca River basins.

Streams

From 10 m digital topographic data (i.e. drainage-enforced digital elevation models (DEMs) (Underwood and Crystal 2002)), a high-resolution stream network was developed for the study area. Algorithms used to model streams allow flow dispersion over topographically divergent areas until a channel is initiated, regulate the degree of topographic convergence permitted at channel heads, and vary the approach by which channels are initiated based on underlying processes (Miller 2002). To minimize extensions of the derived network into planar areas, channels were initiated for the Tillamook Bay and Nestucca River basins from a slope/drainage-area relationship where fluvial processes dominate at gradients less than 25%, and from a single 0.75 ha drainage-area threshold where mass-wasting processes dominate at gradients greater than or equal to 25%. The channel network was divided into reaches by aggregating contiguous pixels with uniform DEM-derived geomorphic and hydrologic characteristics. Endpoints of reaches were placed at tributary junctions except where distances between tributary junctions exceeded the allowable length for a particular drainage area (i.e. 50 to 200 m for drainage areas between 0.04 and 50 km^2, 50m for drainage areas less than 0.04 km^2, and 200 m for drainage areas greater than 50 km^2).

Intrinsic potential

Intrinsic potential to support steelhead or coho salmon was expressed as the geometric mean (Van Horne and Wiens 1991) of classified channel gradient, valley constraint, and mean annual discharge (Table 1), an approach similar to that taken by Gregory *et al*. (2001).

Classes of the attributes reflecting strength of association with steelhead or coho salmon were based on available literature and field observations. The approach assumed that the three attributes were partially compensatory but weights the calculated intrinsic potential by the classified attribute with the smallest value. Calculated intrinsic potential values ranged from zero to one. Intrinsic potential was determined for each reach in perennially flowing streams (drainage area exceeding 0.04 km^2) (Clarke *et al.* 2002) below known barriers to migrating adult salmon (Brodeur and Bowers 2000; Gresswell *et al.* 2000).

Channel gradient

Channel gradient was obtained from the 10 m DEM by fitting a second-order polynomial to stream pixel elevations in a variable-length moving window (Miller 2002). The length of the window was 300 m for channel gradients less than 0.1%, was 30 m for channel gradients greater than 20%, but varied linearly for channel gradients between 0.1% and 20%.

Table 1. Attributes and classified values used to calculate intrinsic potential of streams to support steelhead and coho salmon.

Channel gradient (%)	Classified Value	Valley constraint	Classified Value	Mean annual flow (m^3/s)	Classified value
		Steelhead			
0.00 - 2.00	0.80	Low	0.5	$\leq$ 0.06	0.75
2.01 - 3.00	1.00	Medium	1.00	0.07 - 2.10	1.00
3.01 - 5.00	0.75	High	1.00	2.11 - 21.23	0.75
5.01 - 6.00	0.75			> 21.23	0.25
6.01 - 8.00	0.50				
8.01 - 10.00	0.25				
10.01 -15.00	0.10				
> 15	0.00				
		Coho salmon			
0.00 - 2.00	1.00	Low	1.00	$\leq$ 0.06	0.75
2.01 - 3.00	0.50	Medium	0.50	0.07 - 21.23	1.00
3.01 - 5.00	0.25	High	0.25	0.07 - 21.23	1.00
5.01 - 10.00	0.10			> 21.23	0.25
> 10.00	0.00				

Juvenile steelhead in Oregon coastal streams are commonly found in gradients up to about 6% (Dambacher 1991; Roper *et al.* 1994; Burnett 2001) but have been observed to use low-gradient areas in steeper reaches. Roper *et al.* (1994) determined that densities (number/100m^2) of one-year-old steelhead were positively related to reach gradient for gradients between 0.7% and 2.9% and that one of the lowest observed densities was in the single examined reach where gradient exceeded 6%. Steelhead in this same age class were found predominantly in streams with gradients between 2% and 3% (Hicks 1989). Consequently for steelhead, we assigned the highest value to channel gradients between 2% and 3% and assumed no use upstream of reaches with gradients exceeding 15% (Table 1).

Coho salmon in the Coastal Province of Oregon rear typically in low-gradient stream reaches and decrease in density as gradients increase to about 10% (Nickelson 1998). For example, Schwartz (1990) found a negative relationship between the density (number/100m) of juvenile coho salmon and channel gradient for gradients between 0.5% to 7% and the greatest densities of coho salmon in gradients below 2–3%. Similarly, Hicks (1989) observed juvenile coho salmon predominately in streams with gradients less than 2%. Thus for coho salmon, we assigned the highest value to channel gradients less than or equal to 2% and lower values to gradients exceeding this (Table 1). We assumed that coho salmon did not use areas upstream of reaches with gradients greater than 10%.

Valley constraint

Valley constraint was determined for each stream reach through a generalized linear model between DEM-derived valley width index (VWI) and four classes of field-assigned channel form (Clarke *et al.* 2002; Moore *et al.* 1997; Firman and Jacobs 2001). Valley width index is the ratio of valley-floor width to active-channel width. The valley-floor width for each stream reach was estimated from the 10-m DEMs (Miller 2002). The active-channel width for each stream reach was predicted from DEM-derived watershed area using a regression model developed with field-measured active-channel widths for 264 reaches of stream (Moore *et al.* 1997; Firman and Jacobs 2001; Clarke *et al.* 2002). Values of the DEM-derived valley-width index corresponding to field-assigned channel-form classes were aggregated into three classes of valley constraint (i.e. low: VWI > 8.0; medium: 5.0 < VWI ≤ 8.0; high: 0.0 < VWI ≤ 5.0).

Densities of juvenile coho salmon tend to be greater in unconstrained than constrained reaches (Hicks 1989). Juvenile coho salmon selected unconstrained reaches over other reach types in multiple years, but one-year-old steelhead often avoided unconstrained reaches (Burnett 2001). Reaches with low valley constraint were assigned the highest value for coho salmon but the lowest value for steelhead trout (Table 1).

MEAN ANNUAL DISCHARGE

Mean annual discharge for each stream reach was predicted as a function of drainage area and mean annual precipitation (Lorensen *et al.* 1994). Drainage area to each pixel was calculated from the 10-m DEMs (Tarboton 1997). Each reach was assigned the drainage area of the furthest downstream pixel in that reach (Miller 2002) and the weighted average over that drainage area (Miller 2002) of mean annual precipitation (from PRISM data; Daly *et al.* 1994).

Steelhead occur in a range of stream sizes from upper mainstem rivers to small tributaries (Meehan and Bjornn 1991; Benke 1992). Coho salmon are thought to occur primarily in mid-sized mainstem rivers to small tributaries (Sandercock 1991; Rosenfeld *et al.* 2000). One-year-old steelhead in a coastal Oregon basin selected tributaries over the mainstem in some years but used both stream-system types with equal probability in other years (Burnett 2001). Juvenile coho salmon in this same basin selected for the mainstem in some years but for mid-sized tributaries in others (Burnett 2001). Thus, we assigned a similar range of values to mean annual discharge for each species, but streams with a mean annual discharge between 2.10 and 21.23 m^3/s were assigned a slightly lower value for steelhead than coho salmon (Table 1).

Land ownership and land use

Land-ownership data (Fig. 2 *left*) were derived from the Western Oregon Industrial Forest Land Ownership digital coverage. Data were aggregated into six classes: United States Forest Service, United States Bureau of Land Management, State of Oregon, miscellaneous public, private industrial, and private non-industrial. The miscellaneous-public class included various developed and less-developed lands such as cities, road right-of-ways, and county parks. The private-industrial class included owners with at least 20 km^2 of timberland and/or a log processing facility (Gedney *et al.* 1986). The private-non-industrial class included owners of timberlands not meeting these criteria or of lands managed for purposes other than timber production.

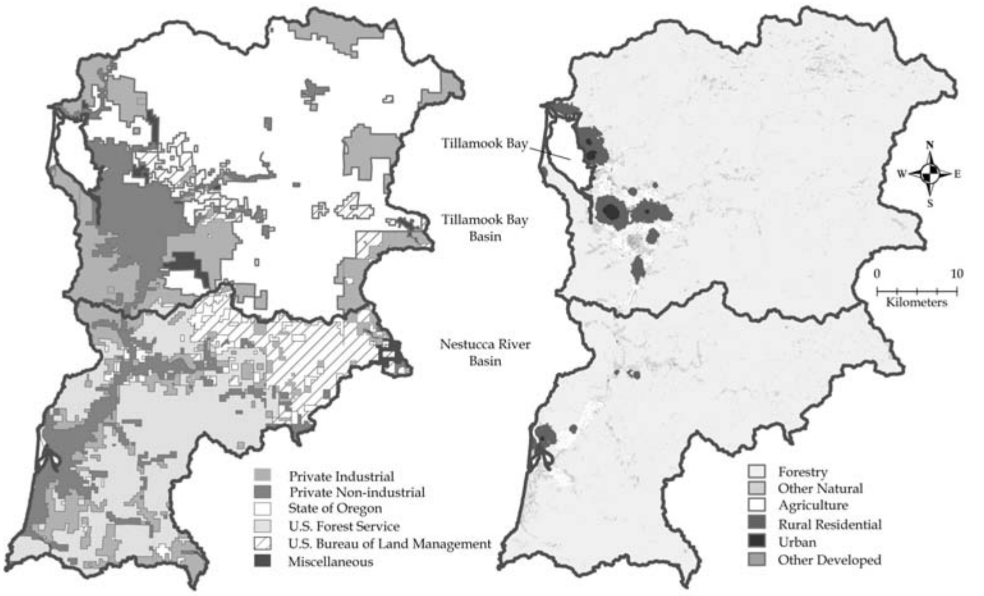

Fig. 2. Landownership (*left*) and land use (*right*) in the Tillamook Bay and Nestucca River basins, Oregon, USA.

Land-use data (Fig. 2 *right*) were obtained by combining raster layers of human development, forest cover, and the 1992 National Land Cover Data (Vogelman *et al.* 2000). Six land-use classes were identified: rural, urban, forestry, agriculture, other natural areas, and other developed areas. The human-development layer was derived by interpolating structure densities (number of structures in a 32 ha circle around a photo point) among a grid of regularly spaced photo points from 1995 (Kline *et al. in press*). Lands classed as rural had 0.25–2.5 structures/ha, and those classed as urban had more than 2.5 structures/ha. Where structure densities were less than 0.25/ha, forest-cover data at 25-m resolution were modeled by integrating vegetation measurements from field plots, mapped environmental data, and Landsat Thematic Mapper imagery from 1996 (Ohmann and Gregory 2002). The forestry class contained open areas resulting from timber harvest, semi-closed canopy forest on private industrial timberlands, semi-closed canopy forest that resulted from timber harvest on other land ownerships, and closed-canopy forest. Forest cover data consisting of open areas not due to timber harvest, water, and woodlands/other vegetation were considered non-forested (K.N. Johnson *pers. comm.*). Land uses for these non-forested areas were determined from the 30-m resolution National Land Cover Data (NLCD). Lands classed as: (*1*) 'agriculture' included orchards, vineyards, pasture/hay/grains, row crops, and fallow areas on the NLCD; (*2*) 'other natural areas' included water, bare rock/sand/clay, perennial ice/snow, and all other natural vegetation (e.g. grasslands, wetlands, and shrub lands) on the NLCD; and (*3*) 'other developed areas' included transportation corridors, quarries/strip mines/gravel pits, urban recreational grasses, and any other developed land use on the NLCD.

Characterizing reaches with high intrinsic potential

Reaches were classified as having a high species-specific intrinsic potential when the calculated value was at least 0.8. Such reaches were assumed to be the most capable of supporting the species. A buffer was generated that extended 60 m on either side of these stream reaches with high intrinsic potential. The buffer width was intended to encompass the zone most likely to directly influence these reaches, so approximated the expected height of old-growth conifer trees in the study area. Buffers surrounding high-intrinsic-potential reaches were characterized relative to the percent area in each land ownership and land-use class.

RESULTS

On the basis of the 10-m DEMs, 10,421 km of streams were delineated for the Tillamook Bay and Nestucca River basins. Approximately 2160 stream kilometers were believed to be accessible by steelhead because none of these were upstream of known barriers or of reaches with a gradient exceeding 15% (Fig. 3 *left*). Of this accessible stream length, 545 km were classed as high intrinsic potential to support steelhead (Fig. 3 *left*). Coho salmon were assumed to have access to 1479 km of the modeled stream network that were not upstream of mapped barriers or of reaches with gradients exceeding 10% (Fig. 3 *right*). Reaches with high intrinsic potential to support coho salmon constituted 268 km of this accessible length (Fig. 3 *right*). Reaches with high intrinsic potential for steelhead and coho salmon occupied 5.2% and 2.5%, respectively, of the total modeled stream length.

Land in the Tillamook Bay and Nestucca River basins is distributed among six ownership classes, but the State of Oregon owns the largest percentage of the area (Fig. 4 *upper*). For steelhead, land ownership in the buffers adjacent to reaches with high intrinsic potential reflected overall land ownership in the two basins with a few minor exceptions (Figs 4 *upper* and 4 *lower*). As an example, the State of Oregon owns 38% of the basin area but 44% of the buffered area adjacent to high-intrinsic-potential reaches for steelhead. For coho salmon, the distribution of land ownership in the buffer adjacent to the high-intrinsic-potential reaches differed from overall land ownership (Figs 4 *upper* and 4 *lower*). Approximately 95% of the buffered area adjacent to the reaches with high intrinsic potential for coho salmon was privately owned, with the majority of this being held by non-industrial owners.

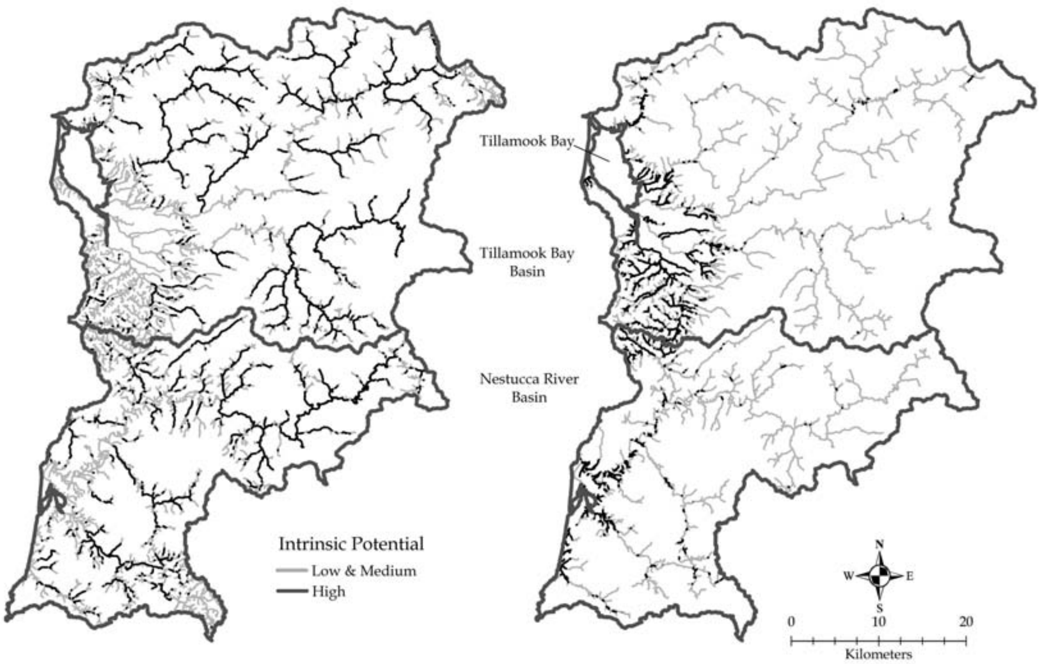

Fig. 3. Intrinsic potential of stream reaches assumed accessible in the Tillamook Bay and Nestucca River basins, Oregon, USA, by: (*left*) steelhead and (*right*) coho salmon. Areas upstream of reaches with gradients exceeding 15% were assumed inaccessible by steelhead and exceeding 10% were assumed inaccessible by coho salmon. Values of intrinsic potential ≤ 0.8 were classified as high.

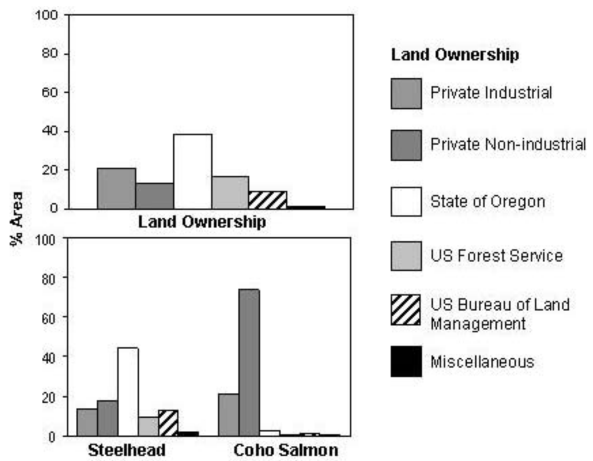

Fig. 4. Percent area by landownership classes (*upper*) in the Tillamook Bay and Nestucca River basins, Oregon, USA, and adjacent to reaches in these basins with high intrinsic potential (≤0.8) for (*lower*) steelhead and coho salmon.

Although forestry is the dominant land use in the Tillamook Bay and Nestucca River basins, more intensive uses such as agriculture, rural residential, and urban are also present (Fig. 5 *upper*). The percentages of area are almost identically distributed among land-use classes in these basins and in the buffers adjacent to reaches with high intrinsic potential for steelhead (Figs 5 *upper* and 5 *lower*). This is not true for the buffers adjacent to reaches with high intrinsic potential for coho salmon (Figs 5 *upper* and **5c**). Forestry constitutes approximately 46% of this buffered area as contrasted with 91% of the basin area. Analogously, more intensive land uses occupy 34% of buffered area adjacent to the reaches with high intrinsic potential for coho salmon but only 5% of the basin area.

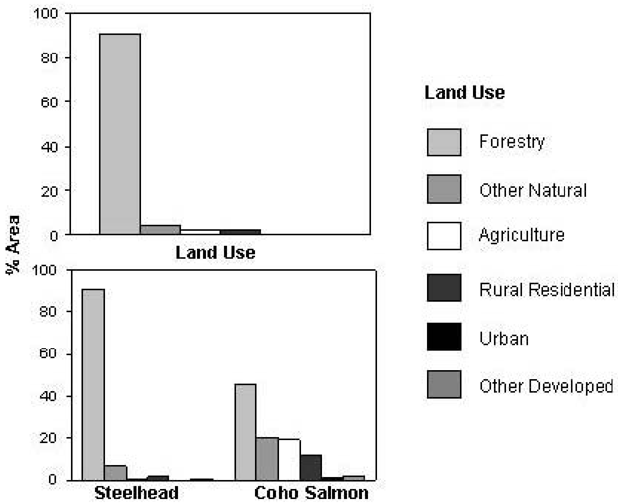

Fig. 5. Percent area by land-use classes (*upper*) in the Tillamook Bay and Nestucca River basins, Oregon, USA, and adjacent to reaches in these basins with high intrinsic potential (≥0.8) for (*lower*) steelhead and coho salmon.

DISCUSSION

The intrinsic potential of streams to support steelhead and coho salmon was modeled from digital topographic data. Because types of landforms associated with steelhead and coho salmon differ, stream reaches identified with high intrinsic potential for these two species generally did not overlap. Reaches with high intrinsic potential typically occurred on publicly owned forestlands for steelhead but on privately owned lands with various uses for coho salmon. These results are relevant in describing the likelihood of finding unimpaired habitat in reaches with high intrinsic potential for these species and in assessing the feasibility of conservation options, thus in identifying freshwater protected areas. Additionally, findings for the Tillamook Bay and Nestucca River basins demonstrate that the models and approach may be readily used over broad spatial extents. Although this study focused on steelhead and coho salmon, the developed tools may be adapted and applied to help identify protected areas for other freshwater species with distributions that are influenced by topographic features.

Stream reaches with high intrinsic potential on public lands may contain less impaired habitat than those on private lands because private lands have been intensively and consistently managed for much longer than public lands. Settlement in the Coastal Province of Oregon began in the mid 1800s and gradually progressed upslope and upstream from the easiest locations in low gradient, unconstrained valley bottoms around river mouths (Sedell and Luchessa 1982). By the 1880s, forests were cleared along main tributaries of most major rivers in western Oregon, and activities associated with development greatly reduced habitat quantity and quality (Sedell and Luchessa 1982). Logging on most private industrial timberlands has continued under a relatively short rotation interval (40–60 years). The majority of the land in the study area owned by the State of Oregon had been logged or recently burned when the State began acquiring it the 1920s and 1930s; thus, timber harvest on these lands has been limited while forests have been allowed to grow. Timber was rarely harvested from federal lands until private lands were unable to meet demands generated by World War II and post-war economic expansion (Wilkinson 1992). Logging accelerated on federal lands in the study area until the late 1980s then declined precipitously. Although histories of State of Oregon and USA government lands are different, net results are similar – fish habitats have been exposed to less management activity on public than on private lands during the past 150 years, probably leaving more unimpaired habitats on public lands.

High-intrinsic-potential reaches occurring on lands governed by relatively permissive land-use policies are more likely to contain impaired aquatic habitats. With laws passed in the 1970s, forestry became and remains the most regulated land use in Oregon regarding non-point-source water pollution. However, policies differ among forest ownership classes; regulations are more restrictive on federal lands (USDA and USDI 1994), intermediate for State lands (Oregon Department of Forestry 2001), and least restrictive on private lands (see Young 2000 for summary). Aquatic-related measures regulating urban and rural land uses in the study area, though mandatory, are less stringent than those for forestry, and aquatic-related measures for agriculture are largely voluntary. Because more of the area surrounding reaches with high intrinsic potential for steelhead was on public lands, subjected to less-intensive uses and managed under more protective policies, less impairment is expected in key habitats for steelhead than for coho salmon.

Probable locations of future freshwater protected areas may be influenced by land ownership and use because these can affect degree of habitat impairment and dictate applicable land-use policies. Given differences in land-use history and governing policies, reaches with high intrinsic potential may have fewer impaired habitats on public lands than on private lands, especially those managed for intensive uses. Thus, high-intrinsic-potential reaches on public forestlands are anticipated to supply some of the best-quality habitats remaining in the study area for both steelhead and coho salmon. Once this is corroborated directly from in-channel conditions (e.g. pool density or large wood volume) or indirectly from management indicators (e.g. forest stand age or road density), these reaches and encompassing watersheds [catchments] may contribute substantially to conservation if protected. When ancillary data suggest impairment, habitat restoration is likely to yield positive biological results in reaches with higher than lower intrinsic potentials. High-intrinsic-potential reaches on public lands can be easily incorporated into watershed protection frameworks. Watersheds have been identified as logical conservation units for aquatic systems because habitat conditions may be largely determined by upslope and upstream influences (Reeves *et al.* 1995; Moyle and Randall 1998). Specific watersheds, in which stream protection and restoration are emphasized, were identified on federal forestlands (Key Watersheds) (USDA and USDI 1994) and are proposed for State forestlands (Salmonid Emphasis Watersheds) (Oregon Department of Forestry 1999) in the study basins. Areas where reaches with high intrinsic potential are concentrated for steelhead, for coho salmon, or for both species, can help decision makers select watersheds to include in these protection frameworks.

Although directing conservation activities toward reaches with high intrinsic potential on public lands may be necessary and garner less societal resistance, this may be insufficient for conserving all species of salmonids. Private forested, agricultural, rural, and urban lands in the study basins represent a substantial percentage of areas adjacent to reaches with high intrinsic potential for coho salmon. Thus, widespread recovery for coho salmon is doubtful unless private land owners can be encouraged to protect and restore habitat in high-intrinsic-potential reaches through education, incentives, or stricter regulations. Additionally, State and federal forests, despite encompassing a large percentage of the Tillamook Bay and Nestucca River basins, do not occupy as much area for every basin in the Coastal Province of Oregon. Consequently, a narrower range of conservation options will be available for steelhead in basins with less public land unless current policies governing other ownerships are expanded and strengthened.

By identifying the most topographically favorable stream reaches for salmonids, this research provides tools that help focus broad-scale programs on areas most likely to deliver conservation benefits. Targeting activities may increase opportunities for success and is more efficient so may decrease economic costs. These outcomes should improve societal support of efforts to protect and restore habitats for Pacific salmon and trout. We think these results are important first steps in providing a basin-scale context for numerous pending site-scale habitat protection and restoration decisions in the Tillamook Bay and Nestucca River basins. Our next steps include incorporating models for cutthroat trout and chinook salmon and evaluating variation among land ownership and use classes in habitat impairment for reaches with high intrinsic potential. Although work remains, this study demonstrates how the approach and developed models might be applied to other species associated with topographic features or scaled-up to aid in regional prioritization of reaches or watersheds as protected areas.

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BIOREGIONAL FRAMEWORKS FOR ASSESSMENTS OF FRESHWATER BIODIVERSITY IN AUSTRALIA

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Abstract

Criteria such as species richness, endemism, rarity, comprehensiveness, adequacy, representativeness, and refugia are often used to assess biodiversity values and priorities for protected area networks. Bioregional frameworks are essential for the application of these criteria. Although hierarchical biogeographic units from regional ecosystems (Sattler and Williams 1999) to bioregions have been defined for terrestrial (Thackway and Cresswell 1995) and marine and coastal (Thackway and Cresswell 1998) biodiversity in Australia, lack of agreed bioregional frameworks currently hinders assessment of freshwater biodiversity values. This particularly applies to representativeness criteria used for protected area planning (Nevill 2001, 2002).

Different components of freshwater biodiversity form bioregional relationships at different scales in response to different biogeographic features, the distribution abilities of biota, and river basin/geological histories. Consequently no single bioregional framework may have application across all components of freshwater biodiversity (Wells and Newall 1997). Substantial data collection and research are needed to progress toward the possible definition of universally applicable Australian freshwater 'bioregions'. Meanwhile, prudent and pragmatic approaches involving the use of existing regionalisations and data are required to serve present freshwater biodiversity assessment and conservation planning needs.

In this paper we consider the potential for applying spatial frameworks provided by terrestrial bioregions, river basins, riverine ecological process zones (Whittington *et al.* 2001), and geographic patterns of aquatic biota including findings from phylogenetic studies, to freshwater biodiversity conservation evaluation and protected area planning. The role and potential of assessments of aquatic ecosystem condition (e.g, the Assessment of River Condition (NLWRA 2002)) to defining the areal status of defined biogeographic units is also discussed.

Keywords: freshwater biodiversity, protected area planning, conservation assessment, biogeographic regionalisation

INTRODUCTION

Biogeographic regions, also known as bioregions or ecoregions, are defined as units of land with relatively homogeneous ecological systems or relationships between organisms and their environment (Omernik 1987). In Australia, bioregions have been developed at a continental scale for terrestrial ecosystems (*Interim Biogeographic Regions of Australia* (IBRA) (Thackway and Cresswell 1995, 1998)), and marine ecosystems (*Interim Marine and Coastal Regionalisation of Australia* (IMCRA)), but not for freshwater ecosystems.

The definition of bioregions is considered an essential step for nature conservation planning, particularly for the design of an ecologically or biogeographically representative system of protected areas (Thackway and Cresswell 1998). Bioregions and subregions are used for two main planning applications regarding biodiversity conservation: as a framework to assess biological resource condition (Fig.1); and to define progress toward representative protected area networks (Fig. 2) (NLWRA 2001).

Most jurisdictions in Australia have made commitments to the development of representative protected area networks for freshwater biodiversity, particularly riverine ecosystems (Nevill 2002). The need for such commitments has been realised within the context of established protected area networks, which have been primarily based on terrestrial ecosystems and biota, and where inclusion of riverine ecosystems has generally been by default rather than design.

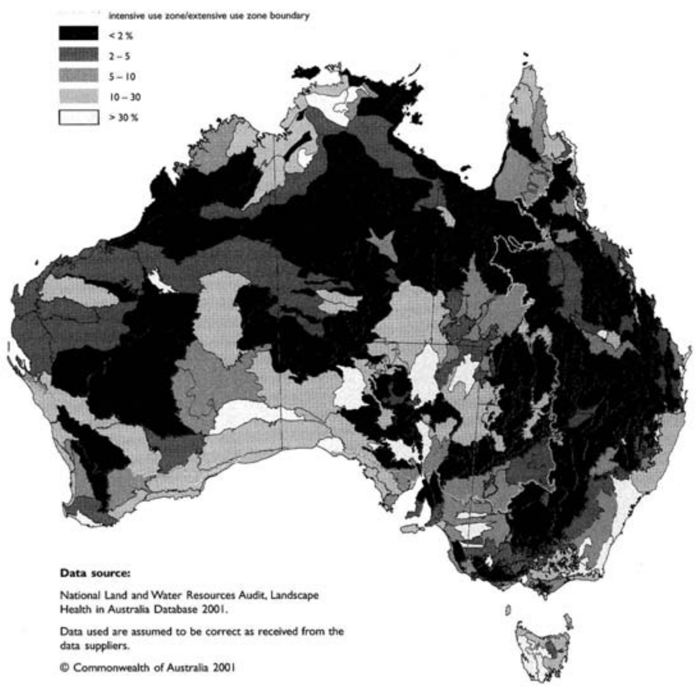

Fig. 1. Percentage of *Interim Biogeographic Regions of Australia* subregions in conservation reserves (NLWRA 2001).

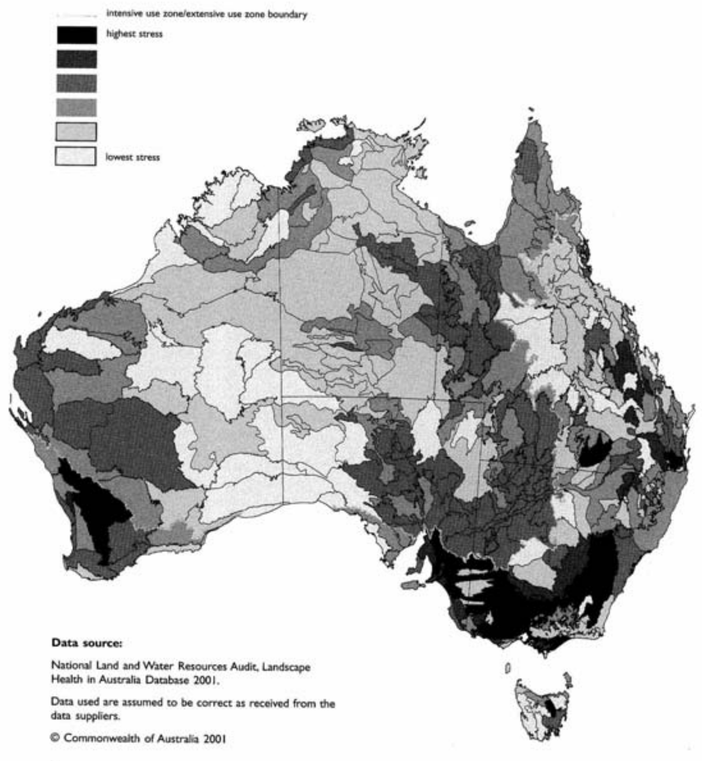

Fig. 2. Continental landscape stress of *Interim Biogeographic Regions of Australia* subregions (NLWRA 2001).

Previous assessments have found terrestrially defined bioregions wanting in terms of application for freshwater biota (Wells and Newall 1997; Turak *et al*. 1999), and the need to develop a biogeographic regionalisation of Australian inland waters is well recognised as a national priority for the protection and management of freshwater biodiversity (Georges and Cottingham 2002).

In the absence of bioregions that can be used to assess freshwater biodiversity, proponents of riverine protected areas (e.g. Cullen 2002) have proposed that river basins in better ecological condition be primarily considered for 'Heritage River' protection. Although ecological condition is a legitimate criterion for selection of protected areas (Dunn 2000), without a bioregional assessment framework and the application of associated criteria such as representativeness, there is a risk that only river basins less affected by development pressures will be protected. This contrasts with bioregional-based assessments of freshwater biodiversity (e.g. Whiting *et al.* 2000), which identify high values in terms of diversity, endemicity, critical species, representativeness and complimentarity in regions with substantially modified catchments and high land-use pressures. Although it is more challenging to implement and manage protected areas in these catchments, they are potentially more crucial for the protection of biodiversity values.

One of the primary constraints limiting the development of a freshwater bioregional framework in Australia, with the exception of a few well studied groups (Choy and Marshall 2000; Georges and Cottingham 2002; Wells *et al*. 2002), is our ignorance of aquatic species and their distribution patterns. However, in the past decade there have been developments that contribute toward the working definition of such a framework, including the following: national-scale sampling of macroinvertebrates for the National River Health Program (Davies 2000); biogeographic reviews of key taxa including fish (Unmack 2001), molluscs (Ponder and Walker 2001) and turtles (Georges and Thomson 2002); the development of molecular tools for mapping phylogeographic regions (Hughes *et al.* 1996; Avise 2001; Hurwood *et al.* 2001; Georges *et al*. 2001; Ponder and Walker 2001); further refinement of existing terrestrial bioregions (e.g. IBRA version 5.1 (Environment Australia 2001)); and new biophysical classification frameworks for rivers and wetlands (Blackman *et al.* 1992; Semeniuk and Semeniuk 1995; Calvert *et al.* 2001; Thoms *et al.* 2001; Thoms and Parsons in press).

PREVIOUS WORK-*A PRIORI* REGIONALISATION

The various approaches to the definition or application of bioregions for inland waters in Australia have been driven by their intended application. This has ranged from predicting water-quality characteristics (Tiller and Newall 1995), assessing ecological condition (Turak *et al*. 1999; Choy and Marshall 2000; Choy *et al.* 2002) and planning for biodiversity conservation (Whiting *et al*. 2000). Several studies described below have used regions defined *a priori* on geomorphic and climatic data for freshwater applications; however, this paper recommends that in planning biodiversity conservation the distribution of aquatic biota should have precedence in the definition of bioregions and that the primary regional framework should be provided by drainage units and within drainage position.

It is only recently that the development of bioregions based specifically on aquatic ecosystems has been progressed in Australia (Wells and Newall 1997). This work followed the example of North American workers (Omernik 1987) in developing *a priori* regionalisations using largely terrestrial attributes (e.g. climatic surfaces, physiography (altitude and landform) and pre-European vegetation). Defined regions were then 'tested' against observed water-quality characteristics, macroinvertebrate assemblages and other biophysical regionalisations. A key limitation of this work was that natural boundaries provided by watersheds were not considered in the definition of regions, despite the recognition that drainage network and positioning were likely to explain much of the observed subregional variation (Wells and Newall 1997). Also, although intrinsic regionalisations evident in the biota (macroinvertebrate) data were acknowledged as an appropriate means of defining the scale of regions, they were not proposed as a primary protocol for the definition of aquatic ecoregions.

Whiting *et al.* (2000) provide another recent example of the application and limitations of *a priori* defined terrestrial bioregions for defining aquatic conservation priorities. They quantify biodiversity values for freshwater crayfish taxa in terms of diversity, endemicity, critical species and complementarity within IBRA regions (Interim Biogeographic Regions of Australia) (Thackway and Cresswell 1995). As the concordance of individual crayfish species and community distributions with the applied IBRA regions was not assessed, the resolution of defined regional conservation values is limited (Fig. 3). For example, resultant conservation planning would still need to make reference to individual species' distribution data to select between regions and identify priority catchments or sites to protect representative examples of the crayfish community. One important finding of that study was that regional biodiversity-conservation values defined in terms of species richness were distinct for different taxa, with northern Australian tropical regions being most important for amphibians (Tyler *et al.* 1981), and subtropical and temperate regions most significant for crayfish.

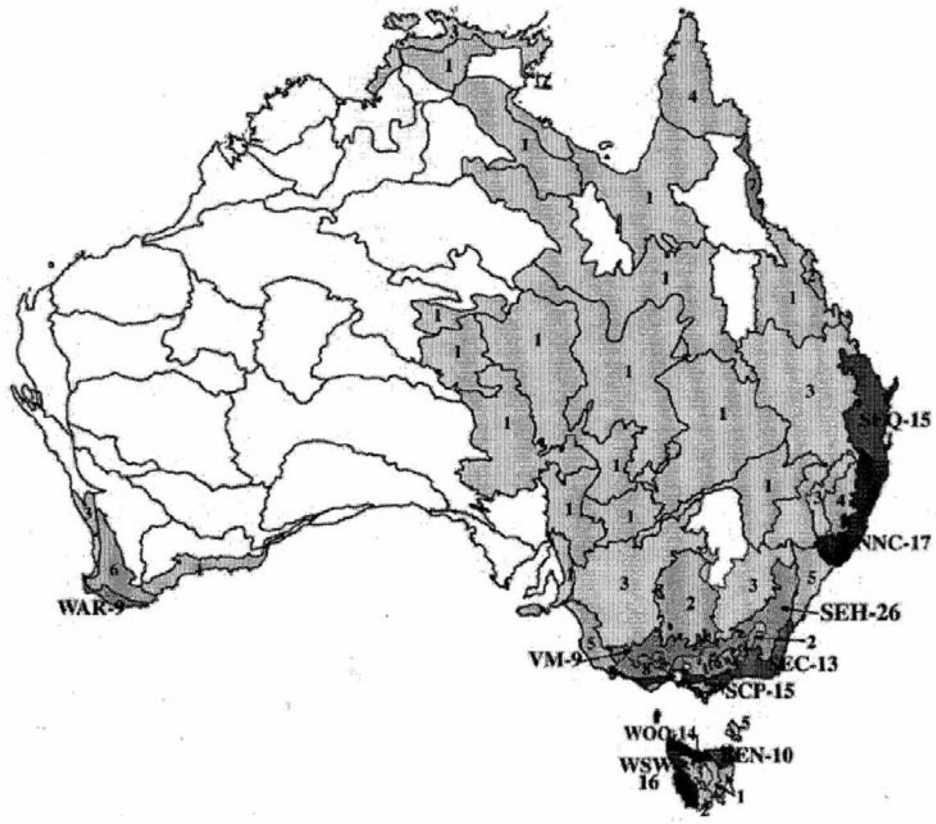

Fig. 3. Species richness of Australian freshwater crayfish taxa within IBRA bioregions (Whiting et al. 2000). Abbreviations are bioregion names (Thackway and Cresswell 1995); shading density reflects species richness classes.

INTRINSIC BIOTA REGIONALISATION

Defining intrinsic regional patterns in aquatic biota, particularly macroinvertebrates, have been progressed by many workers involved with the Australia-wide sampling underpinning the National River Health Program (Davies 2000). This program has developed a RIVPACS-type predictive modelling capacity for regional- and reach-scale macroinvertebrate assemblages. The primary use of this data has been assessment of riverine ecological condition through the comparison of observed and expected values (Turak *et al.* 1999; Huong *et al.* 2000). However, this national data set does have substantial potential for regionally based biodiversity assessment and protected area planning (Wells *et al*. 2002). One limitation of much of the data is that macroinvertebrates have been described only to family level. Although a predictive capacity for macroinvertebrate family assemblages has served riverine condition assessment (NLWRA 2002), defined regions are broad and often do not recognise distinct biogeographic boundaries such as drainage divides (Wells and Newall 1997; Turak *et al*. 1999). In some jurisdictions where macroinvertebrate data have been defined to species level, their potential for defining bioregions has been recognised (Doeg 2001; Wells *et al.* 2002).

ORGANISM VAGILITY

Among Australian States, Victoria has made the greatest progress toward the definition of representative riverine regions using both invertebrate and vertebrate biota (Doeg 2001). That work has shown that an important consideration in the use of biota for the definition of aquatic biogeographic regions is the vagility of different taxa, particularly of totally aquatic organisms in comparison with those with terrestrial life stages or distributional abilities.

In contrast to vagile terrestrial organisms, organisms that are restricted to fresh water (e.g. freshwater fishes) suffer unique biogeographic constraints (Unmack 2001). Their ability to distribute to suitable habitats or move in response to climate change or geological events is limited to the pattern of connectivity of freshwater bodies, which is usually catchment constrained but can include rare events such as drainage rearrangements, changes in width and depth of the continental shelf, and major pulses of fresh water into oceans (Unmack 2001).

THE CASE FOR DRAINAGE BASINS AS A PRIMARY FRAMEWORK

Drainage basins have been considered the most meaningful regionalisations for inland waters because surface waters 'are arranged spatially as a network throughout the landscape effectively controlled by topography' (Georges and Cottingham 2002). Recognition of drainage-network boundaries and their present and historical connectivity is perhaps one of the most important considerations for the definition of freshwater bioregions.

Unmack's (2001) work on the biogeography of Australia's freshwater fishes provides one of the most substantial developments toward the definition of freshwater bioregions in Australia. It restricts its analysis to fish with life histories restricted to fresh water and uses drainage units as the starting point for the definition of regions (Fig. 4) based on discontinuities of fish community distributions using a range of methods.

Where the vagility of individual aquatic taxa is not considered, the resolution of defined regions is poorer. For example, even at species level, macroinvertebrate associations used to define Victoria's river regions (Doeg 2001) are broad and cross major drainages (Fig. 5). With the inclusion of purely aquatic taxa (i.e. freshwater fish) the bioregions more closely define drainages (Fig. 6) recognising major catchment divides (Doeg 2001). Interestingly, associations of both invertebrate and vertebrate biota typical of steep-gradient upper-catchment areas defined for Victorian regions appear less affected by catchment divides occurring in low-order streams on both sides of the dividing range and define a riverine region that straddles both coastal and inland drainage systems (Doeg 2001).

We propose that, after drainage boundaries, the second most important consideration in the definition of freshwater bioregions is within-drainage position. The recognition of within-drainage regional associations reflecting upper, mid and lower catchment areas is a significant finding of the Victorian work (Doeg 2001) and earlier fish-based assessments (Pusey *et al.* 1993, 1995; Gehrke 1997).

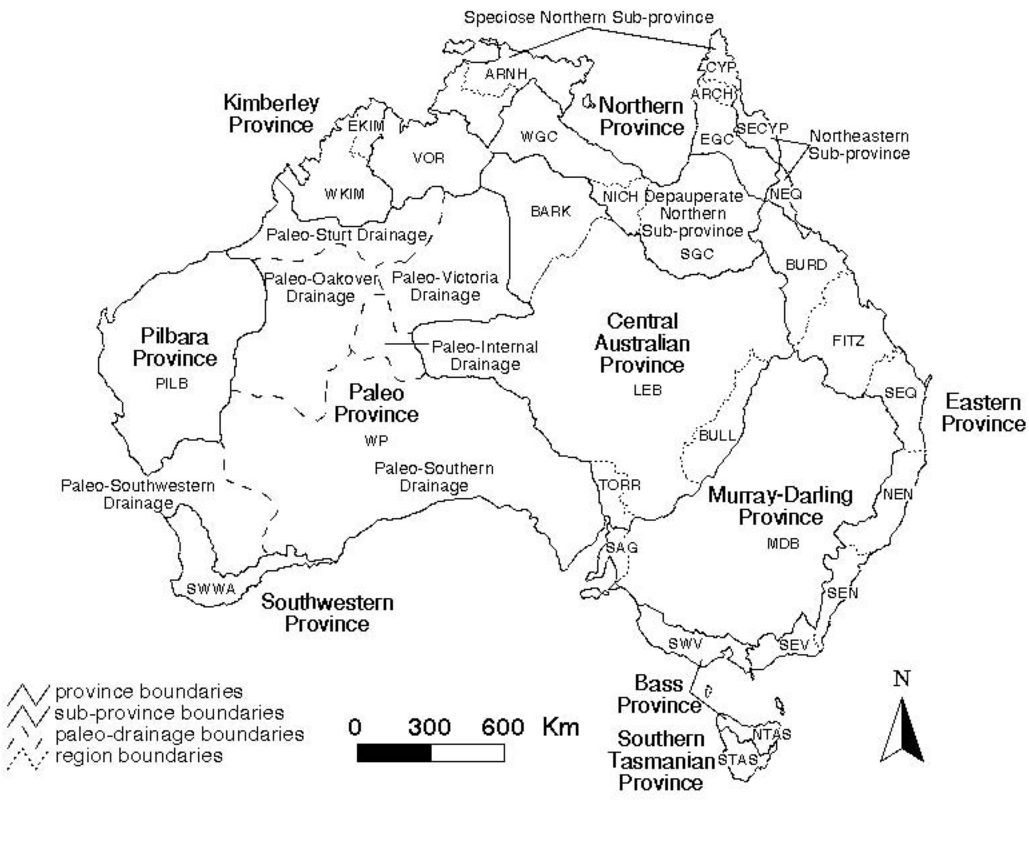

Fig. 4. Freshwater-fish biogeographic provinces proposed for Australia (Unmack 2001)

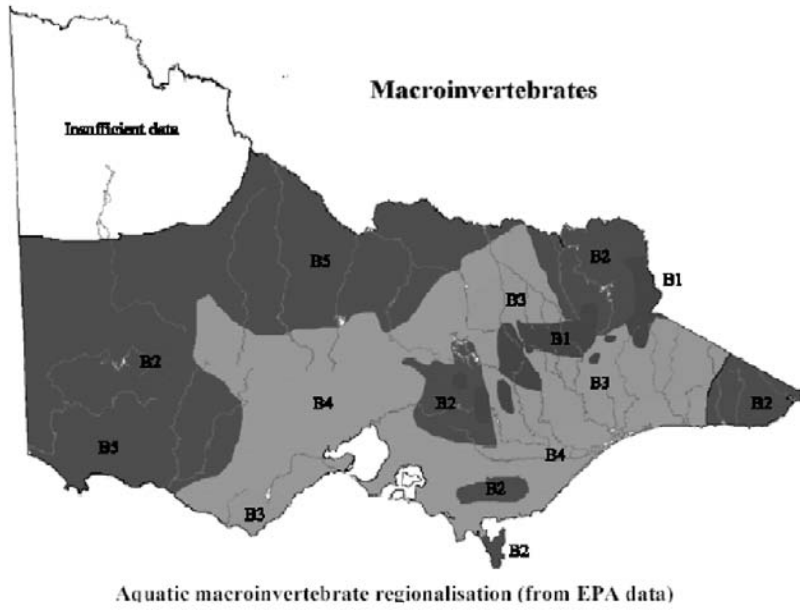

Fig. 5. Macroinvertebrate regions defined for Victorian rivers (Doeg 2001).

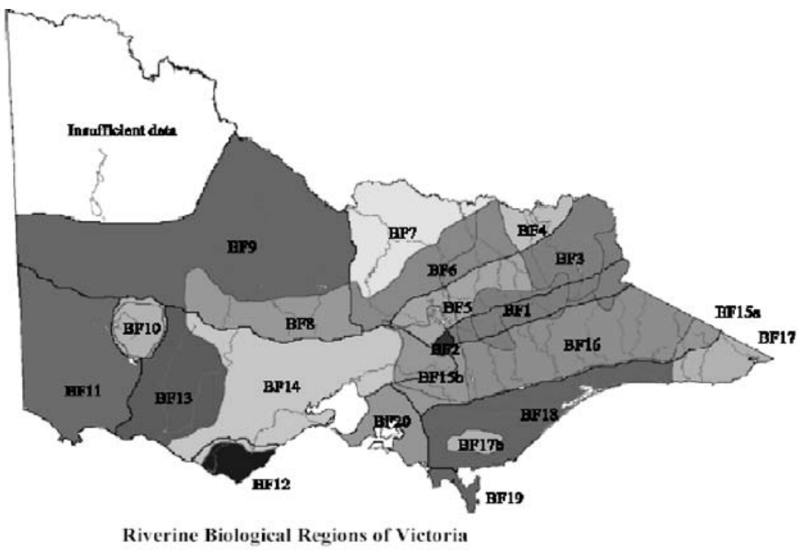

Fig. 6. Riverine biological regions defined for Victorian rivers on the basis of macroinvertebrate and fish biota data (Doeg 2001).

PHYSICAL RIVER CLASSIFICATION

Freshwater ecologists have long recognised that invertebrate and fish community composition is strongly influenced by drainage position and catchment area. This is related to the range of physical habitat settings and associated biophysical processes that influence riverine systems from upper to lower catchment areas (Calvert *et al.* 2001; Thoms *et al.* 2001; Thoms and Parsons in press). Compared with the distinct breaks in aquatic habitat connectivity, and hence aquatic biota community, that exist between drainages, within-drainage distinctions in biota composition are likely to be less well demarcated except where major biogeographic boundaries and discontinuities exist such as waterfalls and lakes. Analysis of intrinsic patterns observed within Australia aquatic vertebrate and invertebrate biota does suggest that upper-, middle- and lower-catchment species associations and, hence, regionalisation can be defined (Pusey *et al.* 1993, 1995; Wells and Newall 1997; Gehrke 1997; Doeg 2001; Georges and Thomson 2002).

The demonstration of concordance between within-drainage biota associations and physical river classifications (e.g. Choy *et al.* 2002) would present the opportunity to divide aquatic bioregions defined on the basis of drainage units (e.g. Unmack 2001) into upper- and lower-catchment subunits with some confidence where detailed biota data are lacking. The recent definition of riverine process zones (Whittington *et al.* 2001; Thoms *et al.* 2001, Thoms and Parsons in press) that integrate both physical and ecological process attributes may distinguish more ecologically meaningful boundaries reflected by the aquatic biogeography. Biotic interactions such as competition and predation could also be considered as additional attributes but data would often be lacking.

ARE TERRESTRIAL BIOREGIONS USEFUL?

Workers in aquatic biogeography have generally dismissed the suitability of terrestrially defined bioregions for explaining patterns in freshwater biota (Georges and Cottingham 2002). This is partly related to the perception that the original 80 bioregions defined for Australia (Thackway and Cresswell 1995) were too broad for the scale of patterns observed in freshwater biota (Marchant *et al*. 1997; Turak *et al.* 1999). However, recent developments in hierarchical terrestrial bioregional frameworks have resulted in finer-scale units including subregions (Environment Australia 2001) and regional ecosystems (Sattler and Williams 1999). These regionalisations have not been tested for their application to freshwater biodiversity, but, given that they 'capture' some of the key geomorphic drivers affecting aquatic habitats and ecological processes, we hypothesise that they would have a legitimate application, particularly for more vagile or terrestrially associated components of freshwater biodiversity such as riparian vegetation communities and associated fauna. Where detailed information on riparian communities is available, indicative assessments show major stratifications of riparian community types across subregion boundaries (Fig. 7). We also hypothesise that aquatic biota with terrestrial adult stages, particularly those that can fly (e.g. many aquatic insects), are likely to have distributions associated more with terrestrial regional ecosystems and not be constrained by drainage boundaries. This would be particularly true for insects that have a relatively long adult flying stage and are strong flyers (e.g. dragonflies).

Ultimately, the suitability and scale of freshwater or terrestrial regionalisations that may be applied for describing the distribution of aquatic biota will be related to the biota's vagility, particularly its ability to distribute across drainage divides, and the extent to which its life cycle is restricted to aquatic habitats. We suggest a generalised relationship between the vagility of aquatic organisms and the suitability of freshwater *versus* terrestrial regionalisations (Fig. 8). Considerations of vagility can also be used to hypothesise appropriate scales of association and applicable bioregional frameworks for different components of freshwater biodiversity (Table 1).

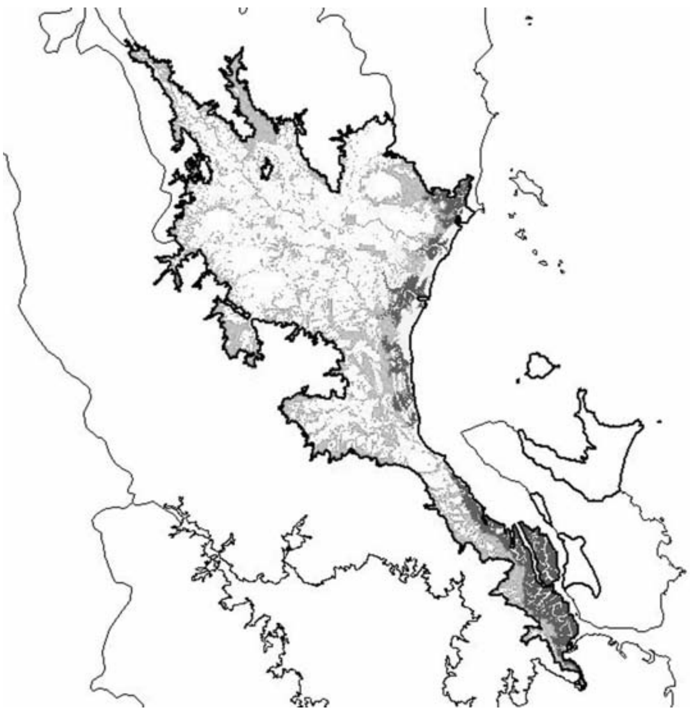

Fig. 7. Remnant vegetation of the Tully Subregion of the Wet Tropics Bioregion. Different shades represent four different land zones. Distinct riparian communities stratify across the different land zones (Qld EPA).

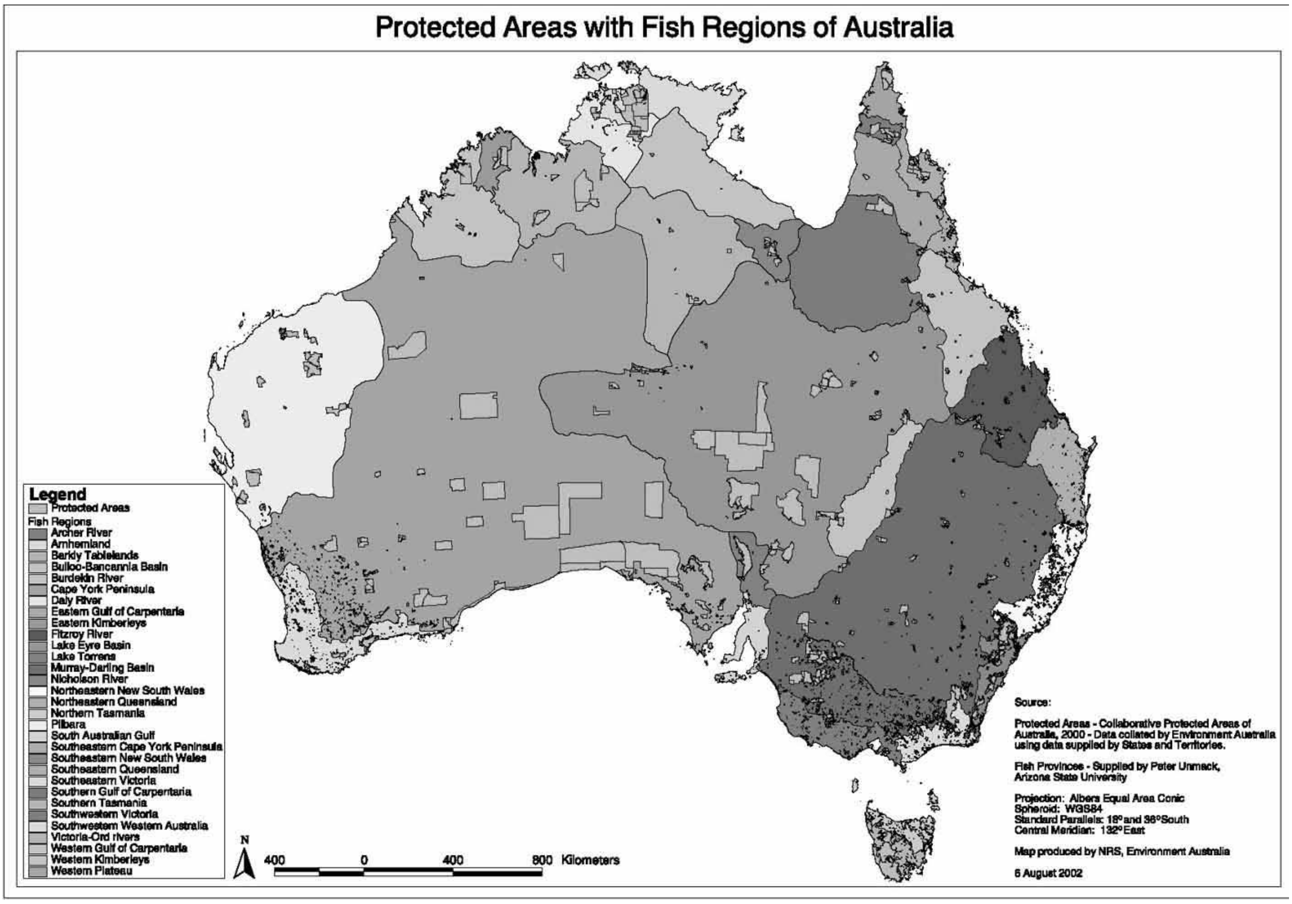

Fig. 8. Generalised relationship between the vagility of aquatic organisms and the suitability of freshwater *v.* terrestrial regionalisations..

Table 1. Hypothesised scales of association and applicable bioregional frameworks for components of freshwater biodiversity with differing vagility

Component of Freshwater Biodiversity	**Distributional patterns, associations & constraints**	**Applicable bioregional framework**
Completely aquatic macroinvertebrates and vertebrates (eg freshwater fish)	Distributions generally constrained by drainage boundaries and prior connection history, within basin distributions controlled by river process zone/wetland type and finer scales of hydrological/habitat stratification.	'Provincial' (Unmack 2001) drainages (e.g. those with shared history of biota exchange) stratified by riverine process zones, or valley scale physical habitats.
Aquatic macroinvertebrates with terrestrial adult phase	Depending on length of adult phase and flying strength, adult distributions associated with suitable terrestrial habitats independent of catchment divides, within basin distributions controlled by river process zone/wetland type and finer scales of habitat.	Riverine process zone/wetland type/regional ecosystems stratifications within IBRA bioregions.
Semi- aquatic vertebrates (ie amphibians, reptiles, birds, mammals)	Distributions associated with suitable terrestrial, riparian and wetland habitats relatively independent of catchment divides (exceptions noted for freshwater turtles; A Georges pers comm.)	Riverine process zone or wetland type/regional ecosystem stratifications within IBRA bioregions or grouped 'Provincial' (Unmack 2001) drainages.
Aquatic plants	Dependent upon distribution abilities of species, pattern of distributions relatively less constrained by catchment boundaries and histories than fauna, within basin distributions controlled by river process zone/wetland type and finer scales of habitat stratification.	Riverine process zone or wetland type/regional ecosystem stratifications within IBRA bioregions or grouped 'Provincial' (Unmack 2001) drainages.
Emergent and terrestrial (riparian) plants	May exhibit some level of catchment bounded distribution but generally associated with suitable terrestrial / riparian / wetland habitats– can cross catchment divides.	Riverine process zone or wetland type/regional ecosystem stratifications within IBRA subregions – bioregions.

WETLANDS OTHER THAN RIVERINE ECOSYSTEMS

Much of this paper has focussed on riverine ecosystems and associated biota. In considering bioregional frameworks for the assessment of freshwater biodiversity it is important to recognise that much of it occurs in wetland ecosystems other than the linear drainage networks of river systems. Subterranean and groundwater-associated ecosystems also host biota. Although no attempt is made to address the particular bioregional associations of these systems here, the observation can be made that even subterranean systems are often contained within the hydrological systems of individual catchments and are likely to exhibit some level of bioregional distinction between drainage systems.

Patterns of biodiversity of surface wetlands are likely to reflect both terrestrially and aquatically derived bioregions. The primary means often used to classify surface wetlands are their landform setting and associated vegetation types (Blackman *et al.* 1992; Semeniuk and Semeniuk 1995), both being largely governed by attributes reflected in terrestrial bioregionalisations. Some existing approaches to wetland classification recognise the distinct bioregional drivers of wetland form, function and biodiversity and classify them within the nested hierarchy provided by the existing terrestrially based bioregions (Blackman *et al.* 1992).

However, less vagile aquatic biota within freshwater wetlands are most likely to have distributions confined by drainage basins and hence will be best served by biodiversity assessment frameworks that use bioregions defined on the basis of drainage basins.

DISCUSSION

On the basis of the preceding review of work that has contributed toward defining freshwater bioregions in Australia, the following principles and approaches are proposed as a way forward to the development of an *Interim Freshwater Biogeographic Regionalisation for Australia*:

1. Distribution of aquatic biota should have precedence in the definition of bioregions used to serve biodiversity conservation planning (e.g. Wells *et al.* 2002), with physical attributes used primarily to help define finer scale subregionalisations.

2. The framework should be hierarchical to enable biodiversity assessments and planning to be made at a number of scales.

3. The macro-regions (top of the hierarchy) for a freshwater bioregional framework should be based on riverine drainage systems and various-scale aggregations of drainage systems defined by shared aquatic biota demonstrating historical connectivity. The drainage-based freshwater-fish regions and provinces defined Australia-wide by Unmack (2001) form a robust starting point.

4. The second level of the framework hierarchy should be defined within drainages, with sub-drainage units defined for upper-, middle- and lower-catchment areas. These sub-regions should be defined on the basis of distinctive sub-drainage associations of aquatic biota particularly recognising natural biogeographic boundaries such as escarpment waterfalls, lakes and major breaks in slope where associated changes in hydraulic power are reflected by valley-scale in-stream changes in habitat and associated biota. In the absence of available biota data we suggest that for defining the first interim bioregionalisation, physical and 'river process zone' valley classifications be used (Calvert *et al*. 2001; Thoms *et al*. 2001; Thoms and Parsons in press). Subsequent research effort could then be directed towards identifying the existence and scale of concordance with within-basin biogeographic patterns as defined for both vertebrate and invertebrate biota (Pusey *et al.* 1993, 1995; Gehrke 1997; Doeg 2001; Choy *et al*. 2002).

5. Two lower spatial scales of the bioregional framework hierarchy may also be defined, these being equivalent to the riverine 'reach' and 'habitat patch' association recognised by both geographers and ecologists (Calvert *et al.* 2001; Thoms *et al.* 2001). These associations will usually form discontinuous units and their application would primarily be for within-basin site assessment rather than national, State or regional applications in an *Interim Freshwater Biogeographic Regionalisation for Australia*. More detailed analysis of macroinvertebrate data at a species level and constrained to particular taxa is likely to provide a biogeographic basis for defining some of these smaller-spatial-scale regional associations.

6. The differing vagility of taxa should be recognised in the choice of bioregional frameworks for conservation assessments. This approach acknowledges that more than one bioregional framework is needed to serve conservation assessments for all components of what is recognised as 'freshwater biodiversity', and that a legitimate case can be made for the application of both terrestrial and freshwater-based bioregionalisations in

assessing the status of biodiversity for conservation planning.

7. Where elements of freshwater biodiversity (e.g. freshwater wetlands) reflect both terrestrial and freshwater bioregional patterns, one regionalisation should be used to stratify assessments within the other, with precedence based on spatial scale and the vagility of the biota being examined.

EXAMPLE APPLICATION OF A FRESHWATER BIOREGIONAL PLANNING FRAMEWORK

Representativeness of freshwater-fish regions in protected areas.

Many Australian freshwater conservation biologists do not recognise the potential for applying bioregional frameworks in biodiversity conservation planning, because such approaches have largely been the preserve of terrestrial workers. To illustrate such an application, the freshwater-fish regions of Unmack (2001), have been intersected with the Australian protected area database (Hardy 2001) (Fig. 9). As these protected areas largely contain terrestrial ecosystems, the AUSLIG 250K Australian Drainage Coverage was also intersected to assess the percentage of defined drainage network within each fish region (Table 2) that is included in existing protected areas.

Although the analysis is relatively crude and includes a fallible assumption that riverine systems within terrestrial reserves are protected, there are several significant findings:

1. There are very few protected areas in Australia that are sufficiently large to encompass entire river catchments, exceptions being in Tasmania and Arnhemland. Victoria is the only jurisdiction in Australia that has specifically developed linear protected-area systems to protect the riparian ecosystems of Heritage River basins (Doeg 2001).

2. Even where relatively large percentages of a fish region's drainage network is included in protected areas (e.g. Archer River, south-eastern NSW, south -eastern Victoria (all ~25%), the protected areas predominantly cover upper-catchment areas and do not include lower-gradient mid-catchments or lower-catchment floodplains. This has major implications for components of freshwater

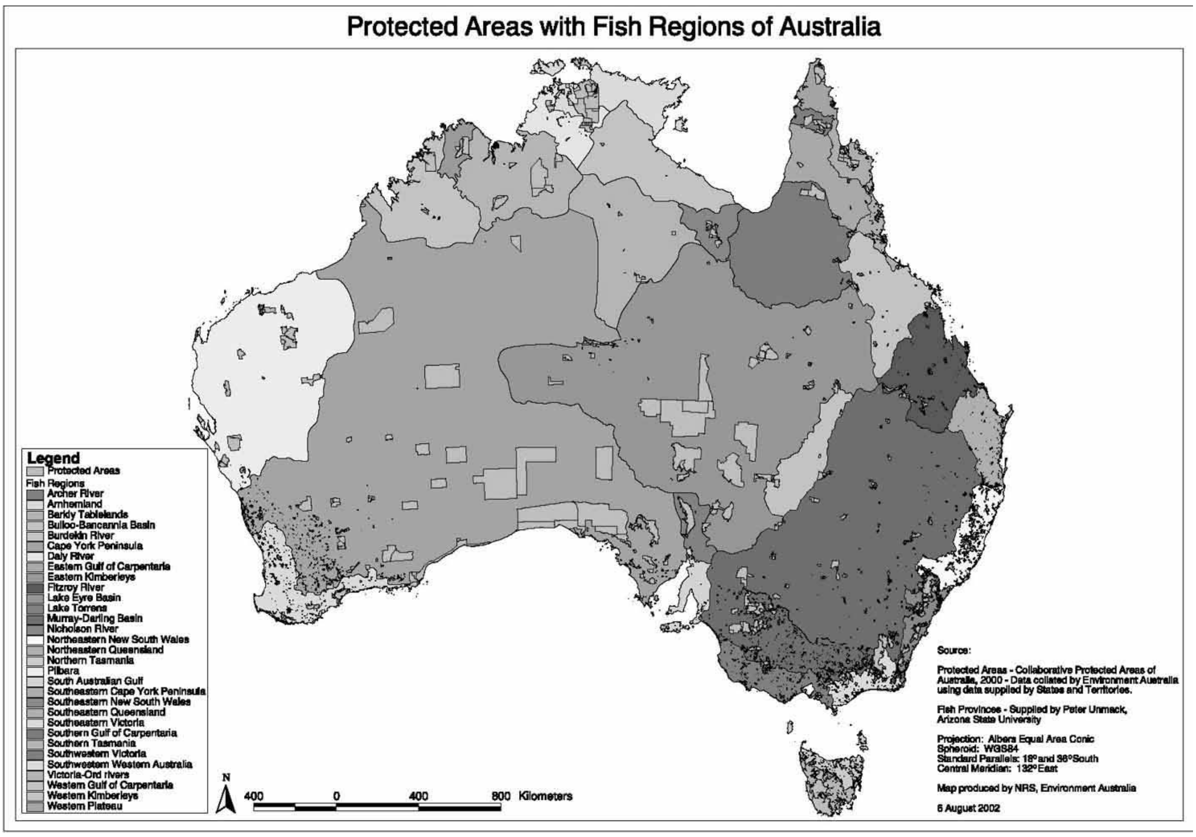

Fig. 9. Overlay of Australian freshwater-fish regions (Unmack 2001) with existing protected areas in Australia (Environment Australia, National Reserve Section).

Table 2. Percent of AUSLIG 1:250000-drainage network within each Australian freshwater-fish region (Unmack 2001) that is included in existing terrestrial reserves.

Fish Region Name	%	Fish Region Name	%
Southern Tasmania	45.2	South Australian Gulf	5.1
Archer River	29.2	Barkly Tablelands	4.9
South-eastern New South Wales	26.3	Lake Eyre Basin	4.7
South-eastern Victoria	23.6	Western Kimberleys	4.7
North-eastern Queensland	16.3	Western Plateau	4.6
North-eastern New South Wales	16.2	Pilbara	4.4
South-eastern Cape York Peninsula	16.0	Bulloo-Bancannia Basin	3.9
Arnhemland	15.8	Fitzroy River	3.7
Cape York Peninsula	13.6	Murray–Darling Basin	3.6
Northern Tasmania	12.8	South-eastern Queensland	3.0
Eastern Kimberleys	11.9	Lake Torrens	2.9
Daly River	9.6	Southern Gulf of Carpentaria	2.0
South-western Western Australia	9.3	Burdekin River	1.6
Nicholson River	9.1	Eastern Gulf of Carpentaria	1.0
Victoria-Ord rivers	8.2	Western Gulf of Carpentaria	0.1
South-western Victoria	7.9		

biodiversity such as freshwater-fish communities, which increase in species diversity with increasing catchment area (Pusey *et al.* 1993, 1995; Gehrke 1997), and for some species that are diadromous (and may not be able to traverse more impacted reaches); this highlights the importance of defining regions within basins to help focus more representative conservation planning.

3. Many of the fish regions with low percentages of their drainage network in protected areas represent Australia's more intensively used river basins (i.e. Fitzroy, Murray–Darling Basin, south-eastern Queensland, Burdekin River (all less that 5% of drainage network within protected areas)). Although opportunities for establishing protected areas within these systems may be limited, they are also the systems under the greatest stress from land use, where protected area declaration may provide a legislative impetus for improved catchment management.

Ecological condition status of freshwater fish regions

Areal representation of fish regions within protected areas is only one approach to defining conservation priorities. The other major input is resource condition. In terrestrial conservation assessments, GIS-based analyses of the status of individual bioregions are often undertaken by intersecting bioregions with vegetation clearing or other measures of biodiversity loss or degradation (NLWRA 2001). With riverine and wetland ecosystems, such analyses are confounded by the fact that the ecosystems may continue to remain physically in the landscape in a range of conditions. Recent Australia-wide integrated assessments of river ecological condition (NLWRA 2002) provide a means to assess the condition of defined aquatic bioregions. Analyses that intersect Unmack's defined fish regions with reach scale output of river condition highlight important biodiversity conservation planning considerations. These include opportunities to secure the protection of better-condition river reaches in Australia's more intensively used basins (and more ecologically impacted fish regions), opportunities for lower catchment / floodplain protection in coastal fish regions, and the prevalence of entire river drainages in relatively good ecological condition in northern and inland Australia. These represent substantial opportunities for large-scale protective management.

WAYS FORWARD

An *Interim Freshwater Biogeographic Regionalisation of Australia* based on concepts developed here should be a 'work in progress' limited only by available data. However, it would provide an interim framework with which to progress assessment and planning initiatives for the development of a representative network of inland aquatic protected areas Australia-wide. An example of the application of freshwater-fish bioregional frameworks defined by Unmack (2001) for protected area planning is presented below. There are a number of key areas in which targeted research could further develop an interim framework.

Closer assessment of existing terrestrial regionalisations

Much more work needs to be done to assess the concordance of freshwater biota and ecosystems with the finer-scale terrestrially based regionalisations that have been developed in many jurisdictions (e.g. Sattler and Williams 1999; Environment Australia 2001). Hypothetically these regionalisations should have application for more vagile aquatic biota and freshwater ecosystems with substantial terrestrial components (e.g. floodplain wetlands). They may also have a useful application as secondary classifiers of defined freshwater regions particularly for biota that form biogeographic patterns at finer spatial scales in relation to terrestrial vegetation of landforms.

More distributional data for aquatic biota

Our ability to define bioregions is constrained by our limited knowledge of aquatic biota. This is demonstrated by the many dedicated surveys of various aquatic taxa that continue to unearth undescribed species, even for conspicuous vertebrate taxa such as fish (e.g. Pusey *et al.* 1995; Unmack 2001). To refine biogeographic boundaries and identify regional concordance between taxa, further dedicated surveys of freshwater aquatic biota are required.

Examining concordance between biophysical and biogeographic patterns

Given that Australia is a very large country and that resources are not readily available to undertake comprehensive inventory across all taxa, effort needs to be made to examine the surrogacy value of geomorphic and physical classification approaches for defining biogeographically meaningful boundaries in the absence of available data for biota.

More detailed analysis of macroinvertebrate data sets

The Australia-wide macroinvertebrate sampling efforts of the National Monitoring River Health initiative provide us with one of the only national data sets for aquatic biota. This data set has been primarily used for riverine condition assessment and has not been used to its full potential for defining bioregional associations and biodiversity values. Assessment of species-level data, where they exist, and bioregional definitions using less-vagile more strictly aquatic taxa may prove most useful (e.g. Wells *et al.* 2002).

Use of molecular tools to define phylogenetic boundaries

Traditionally, biogeographers have defined bioregional boundaries on the basis of concordant cross-taxa discontinuities in the distributions of species defined by a range of methods (e.g. Unmack 2001). One of the key limitations of this approach is the influence of physiological tolerances of individual aquatic species and stochastic events on their occurrence and continued persistence in particular areas, which affects the resolution of defined regional boundaries (A. Georges *pers. comm.).* Phylogenetic approaches to biogeography examine the flow of genetic material between individuals within species (Hughes *et al*. 1996; Avise 2001; Hurwood *et al.* 2001; Georges *et al.* 2001; Ponder and Walker 2001). By these methods, biogeographic boundaries are indicated where there is marked separation in genetic profiles between populations. The advantage of this approach is that biogeographic barriers can be defined confidently where genealogical evidence concurs across a number of taxa; this avoids the confounding influences of stochasticity and variable vagility. Phylogenetic research on a range of key aquatic taxa offers perhaps the most robust method by which to refine an *Interim Freshwater Biogeographic Regionalisation of Australia.*

CONCLUSION

Although efforts to develop a freshwater bioregionalisation in Australia are recent in comparison to advances made for terrestrial and marine ecosystems, advancement in the study of freshwater biogeography puts us in a position to establish an *Interim Freshwater Biogeographic Regionalisation of Australia.* This framework would most logically be based on the natural biogeographic units provided by river basins, which would form the macro-regions of a hierarchical framework, with the second scale of the hierarchy being defined by sub-drainage regions. Biota distribution and phylogeography using molecular techniques should be the primary tools for defining regions across drainages on the basis of demonstrated past connectivity. The variable vagility of different components of aquatic biodiversity should also be recognised as the key determinant of appropriate bioregional frameworks for conservation assessment, which in some cases will legitimately include terrestrial bioregionalisations.

ACKNOWLEDGMENTS

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WHAT DOES LARVAL FISH BIOLOGY TELL US ABOUT THE DESIGN AND EFFICACY OF MARINE PROTECTED AREAS?

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Abstract

Marine Protected Areas (MPAs) can theoretically achieve two Goals: protection of biodiversity, and replenishment of populations both inside and far outside the MPA boundary. The second is supposed to result primarily from larval export from the MPA. Although there is evidence that 'no-take' MPAs protect biodiversity and have higher stocks of larger, older, more fecund fishes, there is scant empirical evidence to support the notion that MPAs actually do replenish unprotected areas, or if they do, over what spatial scale. This notion of replenishment over large scales is largely based on theoretical considerations of larval dispersal and larval biology. Recent research shows that at least fish larvae do not conform to traditional theory: they may have much more control over where they disperse than previously thought. This has important implications for the design and implementation of MPAs, and what we can expect from them as conservation tools. This paper reviews recent advances in understanding larval fish biology and behavioural capabilities and how these impact on the efficacy and design of MPAs. If larvae are as good at resisting dispersal as their behavioural capabilities suggest, then replenishment in ecologically meaningful quantities probably takes place over much smaller scales than previously thought, and MPAs will have to be designed accordingly. These scales, however, are likely to differ spatially, temporally and among species.

Keywords: dispersal, demography, connectivity, larval-fish behaviour, recruitment, settlement

THE GOALS OF MPAS

Marine Protected Areas (MPAs) may be established to achieve one or both of two Goals: 1) to conserve the biodiversity and populations within their boundaries; and 2) to reseed unprotected, exploited areas outside their boundaries (e.g. Bohnsack 1996; Carr & Raimondi 1999; Russ 2002). The first is essentially the same as in terrestrial protected areas. The second is more controversial, and quite different from the normal goals of terrestrial protected areas, so a bit of background knowledge on the life history of marine fishes is required to understand why many marine scientists think it is reasonable that dispersal should take place over much larger scales in the ocean than on land.

REEF-FISH LIFE HISTORIES AND MARINE PROTECTED AREAS

The vast majority of marine bony fishes have a two-phase life history consisting of distinct larval and adult phases. For the kinds of fishes that MPAs are meant to protect, the adult phase is relatively sedentary, often, for example, living on a reef and never leaving it. In contrast, the larva lives in the open pelagic ecosystem, and may disperse long distances before it settles out to become bottom-associated. Few marine benthic fishes give their young any care after they hatch at 1–3 mm in length. The larvae remain pelagic until they settle 2–8 weeks later at 1–3 cm in length (Leis 1991; Leis & McCormick 2002; Fig. 1). Because they are so small, and because studies on Northern Hemisphere temperate fishes such as cod and herring indicate that larvae are poor swimmers, it has been assumed that the larvae have little or no control over where the currents take them during this pelagic phase. Therefore, it seems reasonable to assume that dispersal of fish larvae takes place over wide areas and is adequately modelled by average currents (e.g. Roberts 1997). Most molecular genetic work indicates little genetic subdivision of fish populations over large areas (indicating large amounts of 'connectivity' between fishes in different areas), thus seemingly supporting this view. Few larvae were thought to return to their spawning sites, meaning that these fish populations are demographically open. In other words, any given reef exports the vast majority of its fish larvae to other downstream reefs, and in turn depends on upstream reefs for replenishment (Fig. 2).

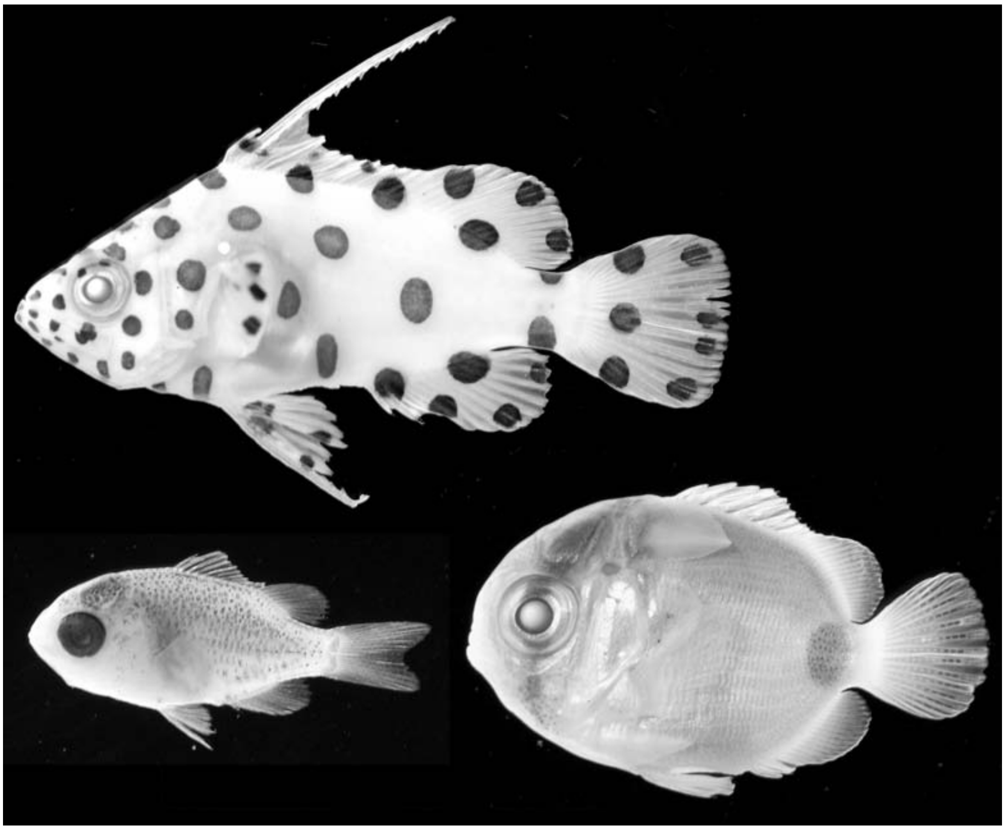

Fig. 1. Settlement-stage larvae of three reef-fish species. Counterclockwise from lower left they are: black-axil chromis, *Chromis atripectoralis* [Pomacentridae] (8 mm SL, Standard Length); bluespot butterflyfish, *Chaetodon plebeius* [Chaetodontidae] (11 mm SL); and barramundi cod, *Cromileptes altivelis* [Serranidae] (14 mm SL). These are preserved specimens. In life, the chromis is silvery laterally and blue-green dorsally, the butterflyfish is yellow with black eye and tail bars, and the cod is transparent with black spots. Photos by Paul Ovenden.

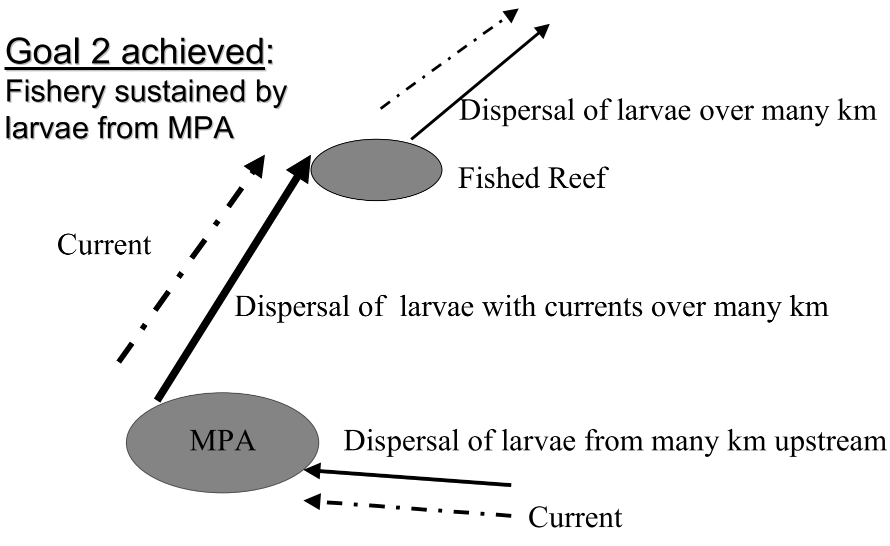

Fig. 2. Passive dispersal of fish larvae: Assumed open population.

EVIDENCE AND ASSUMPTIONS ON HOW MPAS WORK

So, if all this is true, and larvae are dispersed over wide areas, it seems reasonable to conclude that MPAs can replenish exploited fish stocks outside their borders, fulfilling the second Goal mentioned above. However, MPAs might require input from somewhere else for new fish recruits because the larvae spawned in any given MPA would largely have been exported elsewhere (Fig. 2).

Let's look at this seemingly reasonable conclusion a bit more closely for coral reefs, the fishes I know best. There is a reasonable amount of evidence that there are more and larger fishes within well policed MPAs than outside them, and that there is increased biodiversity within MPA boundaries (Russ 2002 for a recent review). This indicates that MPAs achieve Goal 1. There is no empirical evidence, however, that MPAs achieve Goal 2, or, if they do, over what sorts of scales this can be achieved (Russ 2002). The notion that they can achieve this is entirely based on arguments and assumptions from theory, or, in the case of the genetic evidence, misunderstanding of what the evidence is telling us.

GENETIC EVIDENCE

Let's deal with the genetic evidence first. Wide-spread genetic panmixia does indicate that dispersal over wide areas has taken place, but with the electrophoretic and mitochondrial DNA tools that have been used to look at the question, it takes only 1-2 individuals per <u>generation</u> to move between two populations to keep differences from forming by genetic drift (Shulman & Bermingham 1995; Shulman 1998). These few individuals will keep speciation from taking place, but they won't replenish a fishery. So, it is obvious that these genetic tools are appropriate for asking questions over the evolutionary scale, not over the management scale that Goal 2 is supposed to address. In short, genetic evidence of this sort is suitable for providing outer boundaries of management-significant dispersal, but within these genetically identified boundaries there may be many populations that are of management significance.

Given that it is the larvae that are supposed to provide the dispersal that links populations and is required to achieve Goal 2, it is appropriate to ask what we know about the biology of reef fishes that is relevant to this issue, rather than just assuming that larval biology doesn't really matter.

DOES SELF RECRUITMENT EXIST?

There are recent indications that reef-fish populations are geographically more divided and local than we had previously thought. These indications come from genetics, from research on recruitment and from research on larval fishes. Work by geneticists has increasingly shown more population subdivision of benthic fishes and invertebrates than expected (Planes 2002; Barber *et al.* 2002). As explained above, this may generally be subdivision at larger scales than that of management significance. Two types of research have shown directly that a significant proportion of reef-fish larvae are able to either return to or remain near their natal reefs and recruit back to their parent populations: this is called self recruitment. Jones *et al*. (1999) and Swearer *et al*. (1999), using tagging of otoliths, provided empirical estimates of self recruitment for two species of reef fishes in two different island situations. They detected self recruitment at much higher levels than expected under the open population paradigm (Table 1).

Table 1. Summary of characteristics of the species, locations and results of the two empirical estimates of self-recruitment in island populations of coral-reef fishes. (after Leis 2002).

Characteristic	Study 1	Study 2
Authors	Jones *et al*. 1999	Swearer *et al*. 1999
Species studied	*Pomacentrus amboinensis (Pomacentridae)*	*Thalassoma bifasciatum (Labridae)*
Spawning mode	Demersal eggs	Pelagic eggs
Parental care	Eggs brooded by male	None
Incubation period	4–5 days	1 day
Pelagic larva duration	18–21 days	38–78 days
Location	Lizard Island, Great Barrier Reef	St Croix, Caribbean
Dimensions of island (approximate)	7 by 5 km	45 by 20 km
Nearest reef	1–2 km, many within 20 km	100 km
Method	Mark during incubation and recapture at settlement	Otolith microchemistry
% self-recruitment	15–60%	32–89%
Temporal or spatial variation in % self-recruitment?	Not examined	Yes

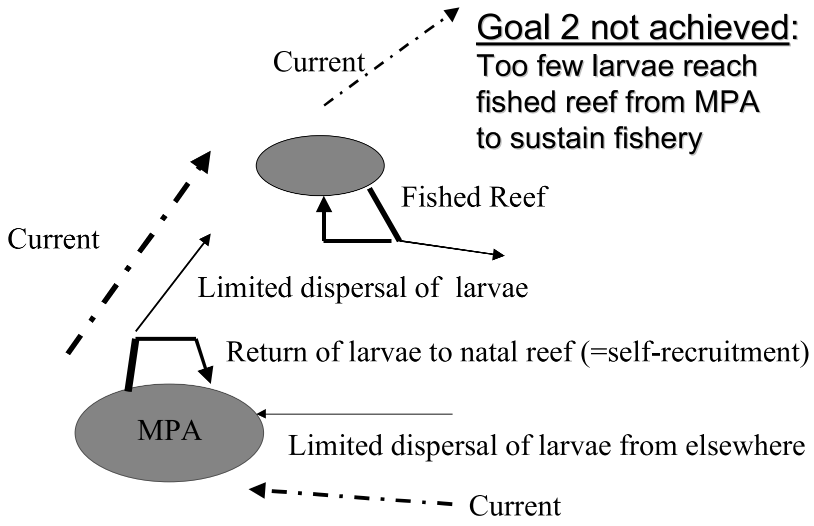

Fig. 3. Non-passive dispersal of fish larvae: Self-recruitment.

Given the differences in the two species studied in terms of spawning type and pelagic larval duration (PLD), the differences in the two environments, the differences in distance to other possible sources for propagules, and the differences in the experimental approaches, it is impressive that the two estimates of self recruitment are so similar. Usually, reef-fish populations are simply assumed to be open (i.e. have insignificant levels of self recruitment), so it is noteworthy that the first two direct estimates of self recruitment are high. In addition, work on larval fishes, otoliths and genetics has shown that some species are capable of completing their life cycle inside the confines of atoll lagoons, thus ensuring a high level of self recruitment at this scale (Leis 1994; Leis *et al*. 1998, in press; Planes *et al*. 1998; Blamart *et al*. 2002). Although this has been shown for only a small proportion of the fish species found on coral reefs, the fact that it appears that lagoonal self recruitment is a facultative ability for some species means that lagoonal self recruitment may be more wide-spread than it first appeared to be. Swearer *et al.* (2002) conclude that there is widespread evidence of self recruitment in marine populations (Fig. 3). It is likely, however, that the proportion of self recruitment will vary among locations, among times and among species. So, the real picture in any situation is likely to be somewhere between those shown in Figs 2 and 3.

HOW SELF RECRUITMENT MIGHT TAKE PLACE: A LOOK AT LARVAL BIOLOGY

If fish populations are more local than previously assumed, then it follows that larval dispersal is more local than previously assumed, and not simply dependent on currents (Fig. 3). In other words, larvae may not meet the 'simplifying assumption' of passive behaviour. In fact, recent research indicates that, at least toward the end of their pelagic phase, the larvae of reef fishes are very competent swimmers with well developed sensory abilities (reviews in Montgomery *et al.* 2001; Kingsford *et al*. 2002; Leis & McCormick 2002). This new insight into what reef-fish larvae are actually doing while pelagic means we have to reassess a number of things, so it is worthwhile to look at what the new research reveals. Nearly all of this work has been done on settlement-stage larvae.

SWIMMING ABILITIES OF LARVAL FISHES

Reef-fish larvae are very good swimmers capable of high speeds over surprisingly long periods of time. The mean swimming speed *in situ* of 50 species of coral-reef fish larvae studied on the Great Barrier Reef and the Tuamotu Islands was 20.6 cm s^{-1} (28 cm s^{-1} = 1 km h^{-1}) with the fastest species swimming at over 60 cm s^{-1} (Leis & Carson-Ewart 1997). The mean current speeds in these areas are 10–20 cm s^{-1}. This means these larvae are 'effective swimmers' (sensu Leis & Stobutzki 1999) capable of swimming faster than local currents. Not only are these larvae fast, they are capable of swimming for many hours at a time at the mean local current speed (Stobutzki & Bellwood 1997) covering many kilometres in the process. The mean endurance for 51 species of coral-reef fishes was 40.7 km (83.7 h), with some families capable of swimming an average of 94 km (194 h) without rest or food in laboratory flumes. Recent work shows these endurance estimates to be very conservative. When provided with access to food, endurance increased nearly two-fold in the larvae of the single species tested (Fisher & Bellwood 2001). Endurance (both time and distance swum) may also increase by several-fold when the larvae swim at slower speeds (Fisher & Bellwood 2002). Swimming speed is frequently 'standardized' to body size (leading to units of Body Lengths, abbreviated BL, per second) to facilitate comparisons. Thus, average speeds *in situ* are 14 BL s^{-1}, and some species swim at speeds up to 40 BL s^{-1} (Leis & Carson-Ewart 1997). A human Olympic swimmer who could maintain 14 BL s^{-1} in the 100 m freestyle would establish a new Olympic record of 4 s (the current record is about 48 s). With swimming capabilities such as these, the average settlement-stage reef-fish larva could easily swim across the width of the Great Barrier Reef near Lizard Island in less than three days. This indicates considerable control over which reef it settles on at the end of its pelagic phase. The limited information available indicates that swimming abilities of larvae of temperate marine fishes are much more mixed (Dudley *et al*. 2000; Jenkins & Welsford 2002; Clark, Hay, Leis & Trnski, unpublished). More work is required to obtain a more complete picture.

In any case, it is clear that the swimming story based on Northern Hemisphere clupeiform and gadiform larvae does not apply to coral-reef perciform larvae (Leis & McCormick 2002). A very interesting finding is that swimming speeds vary depending on location (Leis & Carson-Ewart 1999, 2001, in press). For example, larvae of a species may swim faster in a lagoon than in the open ocean, or faster in open water than near a reef, or faster when swimming away from a reef than when swimming toward or over it. At present, we can only speculate on the reasons for this, but such behavioural flexibility seems to be widespread.

ORIENTATION ABILITIES OF LARVAL FISH

Swimming abilities are of limited use if the larvae don't know which way to swim. However, larvae in pelagic conditions swim in a highly directional way, indicating that they are not simply swimming at random. From 80 to 100% of individual larvae have swimming trajectories that are directional, and when considered at the species level, most species have overall directional swimming trajectories (Leis *et al*. 1996; Leis & Carson-Ewart 1999, 2001, 2003). How is this accomplished?

We now know that reef-fish larvae have good sensory abilities (Montgomery *et al*. 2001; Kingsford *et al*. 2002). They can use the smell of the reef or adult fishes to find settlement sites (Sweatman 1988; Elliott *et al*.1995; Arvedlund *et al*. 1999), they can see well (Shand 1997; Leis & Carson-Ewart 2001) and they can hear the reef (Leis *et al*. 2002*b*). Interestingly, the embryos of anemonefish prior to hatching from their demersal eggs can apparently be imprinted with the smell of their host anemone (Ardvelund *et al.* 1999) raising the possibility that sensory cues unique to the natal reef can be 'learned' by young before they are cast off into the pelagic zone. Thus far, smell as a useful cue for fish larvae has been documented over relatively small scales, but smell has the potential to be an important orientation cue, particularly down-stream of reefs. Reef-fish larvae can use reef sounds as a directional cue. For example, on the Great Barrier Reef, light-traps equipped with speakers that broadcast reef sounds captured 50–200% more larvae of reef-fishes than did quiet light traps (Leis, Carson-Ewart & Hay unpublished). The use of sound as an orientation cue has been documented over scales up to a few 100 m, but as sound travels so well underwater, and as it moves in all directions (not only downstream, as does smell), there is potential for it to be a useful cue over vast distances (Armsworth 2000). So there is good reason to believe that reef-fish larvae use these sensory cues to detect and swim to a reef. Adults of some fish species are known to use magnetic cues in navigation (Montgomery *et al.* 2001; Kingsford *et al*. 2002), so it seems worthwhile to investigate this possibility in larvae of reef fishes.

The swimming trajectories of larvae *in situ* indicate that different species use different cues to maintain their orientation. For example, during the day, it seems that damselfish larvae use a solar compass for orientation. A solar compass would be useful for larvae in the Coral Sea, who would greatly increase their chances of finding a reef (if not any particular reef) by swimming to the west. In contrast, butterflyfish larvae are able to detect the reef and swim away from it. Another damselfish, *Pomacentrus lepidogenys*, swims toward the reef when 1000 m away from it, but not when 100–500 m from it, indicating that it not only knows where the reef is, but also how far away it is (Leis & Carson-Ewart, 2003). Stobutzki & Bellwood (1998) concluded that, at night, damselfish larvae most likely use sound to detect and swim toward the nearest reef.

So it seems clear that reef-fish larvae know where to go when they are swimming in such an impressive way. Some time-dependent (e.g. morning–afternoon, day–night) and location-dependent (e.g. windward–leeward, inshore–offshore) differences in orientation and variation in orientation have been documented (Leis *et al.* 1996; Leis & Carson-Ewart 2003) providing further evidence of behavioural flexibility in reef-fish larvae. Armsworth (2000) concluded that the sensory abilities of larvae are at least as important, if not more so, than are swimming abilities when it comes to finding and reaching reefs for settlement.

WHAT LARVAE CAN'T DO (AS FAR AS WE CAN TELL)

On the basis of first principles, one would not expect that larvae can detect the currents of the pelagic water column in which they swim, and there is no indication that they can do this (Leis & Carson-Ewart 2003). A reference point external to the moving water would be required to do this and, as yet, other than a view of the bottom (useful only when larvae are relatively near the bottom), no such reference point has been suggested that would be accessible to the demonstrated senses of fish larvae.

DEPTH SELECTION BY LARVAE

Currents vary in speed and direction with depth. Therefore, it is important to know what depths fish larvae occupy to determine by what currents they will be influenced. Many numerical models of larval dispersal are two-dimensional: that is, they assume that both currents and larval distributions are uniform with depth. It has long been known that smaller (younger) larvae do not have uniform vertical distributions (Leis 1991), and recent work has described the vertical distribution of settlement-stage larvae (Leis *et al*. 1996; Leis & Carson-Ewart 1999, 2001, and unpublished; Hendriks *et al*. 2001; Fisher & Bellwood 2002; Leis & McCormick 2002). There are large differences among species in vertical distribution, large differences between day and night and, perhaps most surprisingly, large differences in vertical distribution among areas. For example, several species swim deeper in the ocean than in a lagoon, or off the windward side of Lizard Island than off the leeward side.

LARVAL-FISH INTERACTION WITH REEFS: SETTLEMENT

Once a 1–3 cm larva finds a reef, it has to find appropriate habitat, find a way past numerous predators and aggressive residents, and settle onto the reef and transform from a pelagic animal into a benthic one. This is a complex process, fraught with dangers for the tiny larva. Normally, it is assumed in dispersal models that larvae settle onto the first reef they encounter after they are developmentally competent to settle. However, observations *in situ* of settlement-stage larvae released adjacent to reefs show that many larvae reject the reef and swim offshore (Leis & Carson-Ewart 1998, 1999, 2002). This rejection can be 100% in a species of fusilier that will not settle on windward or leeward reefs at Lizard Island, but happily settles on lagoonal reefs. That is clearly a case of a reef-type being unsuitable for a species. However, in other species, a high proportion of individuals may reject reefs that conspecifics have found to be suitable for settlement. Some species may settle only onto live coral, or only onto coral heads with similar-sized recruits (not necessarily of the same species). Even if everything else is suitable, the presence of potential predators or aggressive residents is often enough to cause a larva to abandon attempts to settle and to swim offshore. These sorts of factors are seldom taken into account in dispersal models, but they have a huge potential to influence the distribution of settlement.

The propensity of larvae to abandon settlement attempts and swim back offshore is an indication that it may not be very difficult for larvae to find a reef. If it were difficult, would they so readily swim back into open water?

LARVAL BEHAVIOUR IS FLEXIBLE

Flexibility in the behaviour of larvae is widespread. Settlement has already been discussed, with flexibility in where larvae will settle in response to habitat, predators, reef residents and other, unknown, sources of variation. Swim speeds can vary with habitat and swim direction. Swim depth differs among locations and times. With their ability to be so flexible in behaviour, it should not surprise us that larvae have great control over their trajectories while in the pelagic stage and also over where they settle at the end of it. Interaction of larval behaviour with physical oceanography is very likely to mean that the trajectories of the larvae are significantly different from that of the currents alone (Sponaugle *et al.* 2002; Cowen 2002).

THE SIMPLIFYING ASSUMPTION IS DEAD

Many authors have pointed out that larvae have abilities that would allow them to alter their pelagic trajectories (e.g. Stobutzki 2001). However, until recently, there have been no demonstrations that larval abilities actually result in outcomes different from those produced by the simplifying assumption of passive drift with the current. Most of the species that we worked with at Lizard Island have net trajectories (the result of movement by both swimming and current) that differ in either speed or direction from that of the current alone (Leis & Carson-Ewart 2003). This clearly shows that the simplifying assumption is dead for settlement-stage larvae.

Ontogeny of behaviour: when does behaviour start to make a difference to dispersal?

It seems clear that by the end of the pelagic phase, larvae of reef fishes are behaviourally very competent, and are able to greatly influence, if not entirely control, their trajectories. They have impressive sensory abilities, and the locomotory means to act on the cues they can perceive. They are able to find and settle on the reefs of their choosing over scales of at least 1 km, although at present we don't know whether these scales might be larger.

However, when they initially leave the reef, either as pelagic eggs or newly hatched larvae from demersal or brooded eggs, these reef-fish offspring are very close to being the passive particles that the simplifying assumption would have them be. So, some time during the pelagic period of 10–100 days, these impressive behavioural capabilities develop, but at present we have very limited understanding of when this happens. In other words, we don't know at what time during development the larvae stop being plankton and start being nekton. Most of the limited work on ontogeny of behaviour is on swimming abilities and was done in the laboratory by Fisher (Fisher *et al.* 2000) and shows that by the middle of the pelagic period the larvae are capable of significant speeds and durations. However, Fisher's excellent work was limited to only a handful of species, all of which hatch from non-pelagic eggs. Therefore, research on ontogeny of behaviour needs to be extended to more species, and to aspects of behaviour other than simply swimming in the laboratory.

Larval dispersal: what really happens?

New insights into the behavioural abilities of reef-fish larvae will enable more realistic modelling of larval dispersal (Cowen 2002). Larval dispersal that is demographically relevant is the key to understanding whether and over what scales MPAs will be able to fulfil Goal 2. Realistic dispersal models must be able to incorporate the complex three-dimensional hydrography of coral reef systems at a variety of scales (and possibly at much finer scales within a few hundred metres of reefs than in more open reefal waters). They must include exactly where and when the reef-fish propagules are injected into the near-reef (and over-reef) hydrography. They must incorporate the ontogeny of behavioural and sensory capabilities of the larvae, as well as their capabilities at the settlement stage. They must incorporate realistic estimates of growth and mortality of the larvae (Cowen *et al.* 2000). Finally, they must incorporate the interactions of the larvae with the reefs they are considering for settlement. Early attempts to model dispersal usually used two-dimensional hydrographic models, and the average PLD of the species of interest (and of course, made the simplifying assumption regarding behaviour, e.g. Williams *et al*. 1984; Roberts 1997). They ignored the complex 'hydrographic noise' very close to reefs and stopped at the reef edge. They simply assumed that the larva settled on the first reef encountered once average PLD was reached. Contemporary dispersal models (Wolanski & Sarenski 1997; Wolanski *et al.* 1997; Porch 1998; Armsworth 2000; Armsworth *et al*. 2001; Lindeman *et al*. 2001) are a vast improvement, but still have a long way to go before they can reasonably be expected to realistically represent the complexity that we now know exists.

Our new insights into the capabilities of reef-fish larvae provide an understanding of how and why self recruitment seems to be so much greater and at more local scales than previously assumed (closer to Fig. 3 than to Fig. 2). It now seems more likely that reef-fish populations are more toward the closed than the open end of the open–closed continuum. What are the ecological and management implications of this?

Ecological and management implications of larval biology

The evidence summarized above leads to the conclusion that some level of self recruitment of reef fishes is likely, and therefore that populations are more subdivided at scales relevant to ecologists and managers than previously thought. One expectation that flows from this is that ecologically significant dispersal may well be at very much smaller scales than previously imagined, and many larvae that eventually settle may remain relatively near their natal reef throughout their pelagic period (Fig. 3).

The degree of self recruitment can be expected to vary among reef-fish species. This variation is largely due to differences in larval behaviour among species, not to differences in some vaguely defined 'dispersal ability', usually assumed to be equivalent to PLD (e.g. Carr & Raimondi 1999). Even at the evolutionary scale, PLD has not been a very useful predictor of things like species range, except perhaps at the extremes (Leis 2002). There is little reason to believe it will be any more useful at ecological scales.

The geographic size of management/conservation units is probably smaller than previously thought. If the Lizard Island reef system (Table 1), with an area of about 35 km^2, has a level of self recruitment of 15–60%, management areas on the scale of tens of km^2 may be appropriate. This implies that reserves of this size may be self-sustaining in terms of recruitment for some species. However, like recruitment in general, the percentage of self recruitment undoubtedly varies on a temporal basis. Further, the size of management units will probably vary amongst species. If levels of self recruitment are high, then MPAs might need to include the nearby open-water area where the larvae complete their pelagic phase. Such pelagic MPA additions would be needed if pollution, cooling-water extraction or other threats are likely. At present, the 'radius of return' of larvae of any species is unknown, although the implications of the St Croix study (Swearer *et al*. 1999) are that it could be fairly small (i.e. the width of St Croix 'coastal waters'). At Lizard Island, artificial reefs and light traps caught far more reef-fish larvae near the reef than 500–1000 m away (Leis *et al*. 2002*a*, and unpublished), implying that successful settlers may never move very far from the reef. Research in this area would be particularly useful.

The distance over which reseeding (Goal 2) is supposed to operate is typically unstated, but implied to be large. If self recruitment is high, reserves should be expected to fulfil their first Goal of self-maintenance (e.g. Fig. 3). However, ecologically relevant reseeding over large scales (hundreds to thousands of km) seems unlikely (see also Cowen *et al*. 2000; Barber *et al*. 2002). Over moderate scales, careful research – not assumptions – is needed to determine the reseeding efficacy of MPAs. It is the true (ecologically relevant) dispersal radius that will determine the most appropriate spacing between MPAs. Recent work on mantis shrimps in Indonesia indicates that dispersal may be over larger scales in some regions than in others (Barber *et al.* 2002), implying that MPAs may be more likely to meet Goal 2 in some areas than in others.

If self recruitment is high and, by implication, dispersal distances are small, it follows that for a given area of MPA it may be preferable to protect more smaller parcels of reef than a few large ones. This is because ecologically effective dispersal may not take place over large scales. Thus, the reseeding radius of any one reserve may not be large. However, the critical question of the minimum viable area for MPAs remains unanswered. It probably varies among species, locales, and current regimes. Self recruitment also means that there is increased incentive for local human residents to support MPAs, because the locally produced propagules would not be lost to someone else's reef many kilometres away, but would be replenishing the local reefs.

SOME CAUTIONS

Many of the new insights into larval-fish biology are based on a few studies on a few species. However, unlike the traditional view, they are based on data rather than assumptions. This clearly emphasizes the need for more data in all these areas, and for a less dogmatic view of dispersal and population connectivity not based on untested assumptions (Swearer *et al.* 2002). One area of particular research need is the ontogeny of swimming and sensory capabilities of larvae. Further, what applies to larvae of reef fishes may not necessarily apply to other marine organisms such as sea stars or corals, although it is more likely to apply to invertebrates such as decapod crustaceans with more vagile larvae (Stobutzki 2001; Kingsford *et al.* 2002).

We can expect to find that the proportion of self recruitment varies temporally at any location, and spatially at any time. The two empirical studies of self recruitment (Jones *et al*. 1999; Swearer *et al*. 1999) demonstrate that the proportion of reef-fish settlers derived from local sources can be large, but they also demonstrate that a similar proportion is derived from elsewhere. To fully understand the importance of different sources of settlers, a better appreciation of physiological condition at settlement is needed (Leis & McCormick 2002), because this might vary among sources, especially distant ones, or among environmental factors on the trajectory between source and settlement site. Finally, a better understanding of the levels of settlement and recruitment necessary to maintain populations under different levels of exploitation is needed: if, for example, the external input of propagules is cut off, will self recruitment be sufficient to maintain a given population?

One frequently sees criticisms, usually from fishery biologists, that marine ecologists concentrate their research efforts on 'toy fishes' (i.e. small species like pomacentrids), rather than on the large, commercially exploited species for which Goal 2 is thought to be particularly important. However, this criticism is misguided in the case of pelagic larval stages, because during the pelagic stage, all reef fishes are 'toy fishes'. There is no evidence that the behavioural capabilities of the larvae of 'real fish' species like serranids are demonstrably different than those of 'toy fish' species like pomacentrids (Leis & Carson-Ewart 1997, 1999, 2001). Therefore, we can expect that research on the larvae of both 'types' of fishes will help us understand the dispersal biology of each other.

CONCLUSION

The extent to which MPAs can achieve Goal 2 and over what scales depends where on the open–closed population continuum the fish populations within the MPAs are located. New insights into larval biology and experiments on the source of new recruits to island reefs imply that populations of reef fishes are more toward the closed end and that dispersal distances are smaller than previously assumed (Fig. 3).

The traditional open population paradigm (e.g. Fig. 2) as applied to coral-reef fishes requires reexamination (Leis 2002; Swearer *et al*. 2002). It is based more on assumptions than on data, and attempts to apply it frequently confuse the very different evolutionary and ecological/ management scales. Some of its features were developed based on the dynamics of pelagic fishes such as clupeoids that have ontogenetic, dispersal and demographic characteristics fundamentally different from those of reef fishes. One of these is that propagules of pelagic fishes are immediately placed in the far-field currents, and share the adult habitat, meaning they have no need to find juvenile habitat at the end of their larval phase – they are already in it. Where key portions of the open-population, wide-dispersal paradigm are tested with real data, they frequently fail the test. Recent empirical measures of self recruitment on reefs are at odds with the traditional view, and recent findings on the behavioural capabilities of late-stage reef-fish larvae provide a means by which self recruitment might be achieved and dispersal limited. The view of dispersal outlined herein seems to better fit what we really know about coral-reef fishes. At the very least, the traditional open-population, wide-dispersal view should not be accepted without question but considered simply one extreme view along a continuum, and subject to rigorous testing. In order to get the information we need to design MPAs, we need to apply powerful, contemporary genetic, physical oceanographic and behavioural tools to this testing. We need to recognize the importance to dispersal/retention of larval behavioural capabilities that until recently were inconceivable and of small-scale physical phenomena that were previously viewed as noise to be ignored or filtered out. We also need to make clear distinctions between evolutionary and ecological scales. To treat dispersal this way is much more difficult than required by the traditional assumptions, but it is more realistic and interesting. More importantly, it is more likely to result in effective MPAs that can achieve both Goal 1 and Goal 2.

Our management approach must be adaptive: test and modify MPA design and other measures as more information becomes available (Carr & Raimondi 1999). To postpone management decisions until the final answers to all questions are known means the conservation battle is lost, yet this is just what many opponents of MPAs demand. We must recognize from the start that neither scientists nor managers have all the answers to MPA design at present, but that decisions based on the best evidence (not assumptions), subject to future modification as the 'best evidence' gets better, is the only way forward.

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METAPOPULATION STRUCTURE OF A TEMPERATE FISH IN RELATION TO SPATIAL VARIATION IN HYDRODYNAMICS: IMPLICATIONS FOR SELECTION OF MARINE PROTECTED AREAS

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Abstract

The benefits of marine protected areas to marine species will depend on how the spatial arrangement of marine protected areas relates to the scale of metapopulation structure. One of the major factors influencing metapopulation structure is larval dispersal. For species with a relatively fixed larval duration, the distance of dispersal would be expected to show low variation, and in turn the scale of metapopulation structure would be expected to be relatively fixed. The paper describes a method to investigate larval dispersal of a temperate fish that uses estimated larval duration from daily otolith increments together with a hydrodynamic/dispersal model that is "reversed" from the settlement date. Within one species a high degree of variation in larval dispersal distance can occur over the range that is related to spatial variation in hydrodynamics. Thus, even for a single species, the optimal spatial arrangement of marine protected areas may vary greatly over the species range depending on variation in hydrodynamics.

***Keywords**:* metapopulation structure, marine protected areas, larval advection, hydrodynamic numerical modeling, otolith microstructure

INTRODUCTION

The metapopulation structure of marine species is a primary consideration in the design and implementation of marine protected areas (MPAs) (Man *et al.* 1995; Allison *et al.* 1998; Lipcius *et al.* 2001). Key considerations are the spatial arrangement of the population, and the mobility of organisms through various life-history stages. A critical factor determining the metapopulation structure of marine species, and by implication the design of MPA networks, is larval dispersal (Carr and Reed 1993; Roberts 1997; Botsford *et al.* 2001). Although difficult to measure in the field, understanding larval dispersal must be a high priority for research related to the design of marine protected areas (Carr 2000).

The duration of the larval stage is a major factor that determines the potential for dispersal (Bradbury and Snelgrove 2001). For fish, daily increments in otoliths provide for an accurate determination of larval duration that is not available for most other faunal groups (Victor 1986; Wellington and Victor 1989). Longer larval duration might reasonably be expected to result in greater dispersal distances; for example, larval duration has been shown to be strongly correlated with genetic homogeneity of reef fish populations (Doherty *et al.* 1995). However, in other cases such a relationship has not been found, for example between larval duration and species' geographic range (Victor and Wellington 2000; Zapata and Herron 2002; Sponaugle *et al.* 2002). The actual extent of dispersal will depend on the interaction between larval duration, larval behaviour and regional hydrodynamics; for example, long larval duration might not lead to wide dispersal if physical retention mechanisms are in place (Black *et al.* 1991; Warner and Cowen 2002).

Hydrodynamic modelling, together with knowledge of larval duration and other early life-history parameters, can indicate the extent of larval dispersal between different populations. An example of this approach applies to abalone, where research incorporating larval biology and numerical hydrodynamic modelling has indicated that individual reefs are largely self-seeding and thus supports individual populations (Prince *et al.* 1987; McShane *et al.* 1988).

The dependence of larval dispersal on regional hydrodynamics implies that for a given species the metapopulation structure will vary as a consequence of spatial variation in hydrodynamics. It follows, then, that the optimal arrangement of marine protected areas for a given species or suite of species is likely to vary over the species' range depending on hydrodynamic variability (Carr and Reed 1993). Effects might be expected to be greatest in species with sedentary adults (Sladek Nowlis and Roberts 1999). However, even in more mobile species such variation will be important where critical stages of the life history, such as migration bottlenecks and nursery and spawning areas, are relatively fixed (Roberts 2000; Apostolaki *et al.* 2002).

The King George whiting, *Sillaginodes punctata* (Percoidei: Sillaginidae), is an important commercial species in southern Australia. Post-larvae of King George whiting appear in bays and inlets of Victoria (Fig. 1) from August to November each year (Robertson 1977; Jenkins and May 1994; Jenkins *et al.* 1996), and the bays and gulfs of South Australia (Fig. 2) from June to November (Fowler and Short 1996). This stage is characterised by a full compliment of fin elements but gut coiling has not begun and scales have not formed (Bruce 1995). At this stage, post-larvae are approximately 15 to 20 mm (Jenkins and May 1994; Fowler and Short 1996). The duration of the larval phase up to this point, determined from daily rings on otoliths of post-larvae, is approximately 80 to 150 days, and the approximate spawning period is from May to early July in Victoria (Jenkins and May 1994), and March to July in South Australia (Fowler and Short 1996). This extended larval period gives the potential for wide dispersal during this stage.

There is no evidence that King George whiting spawn in the bays and inlets of Victoria. These populations consist only of sub-adults, and there is an absence of eggs and young larvae (Jenkins 1986; Neira and Sporcic 2002). In South Australia, young larvae are found only near the mouths of gulfs, whilst young juveniles are found deep within the gulfs (Bruce 1989). The only spawning aggregations identified to date in southern Australia were found near Kangaroo Island and south-east Spencer Gulf in South Australia (Fowler *et al.* 1999) (Fig. 2).

Recently, we have used estimates of larval duration based on otolith micro-increments, together with "reverse" hydrodynamic modelling, to identify the possible spawning areas associated with larval recruitment to bays and inlets in Victoria and South Australia (Jenkins *et al.* 2000a; Fowler *et al.* 2000). In this paper we discuss both the methodology and the results from the perspective of the design and arrangement of marine protected areas. We also generalise beyond the species in question to the more general implications for marine species with long larval durations.

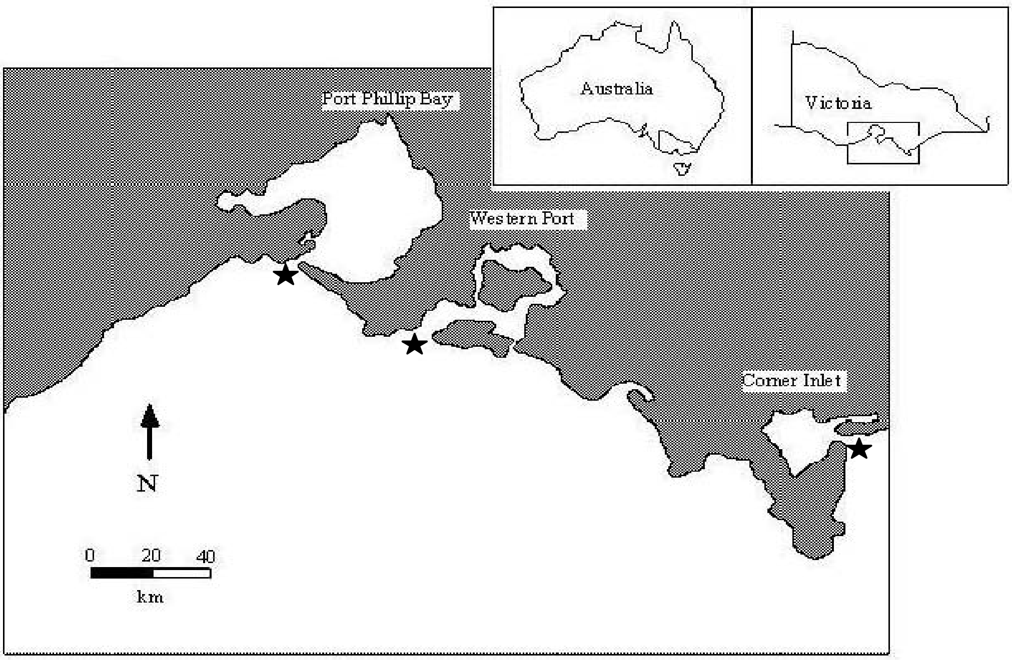

Fig. 1. Map of the central coast of Victoria, including Port Phillip Bay, Western Port and Corner Inlet. Stars indicate initial release point for reverse modelling. Insets: Position of the study area on the Victorian coast, and location of the State of Victoria in Australia.

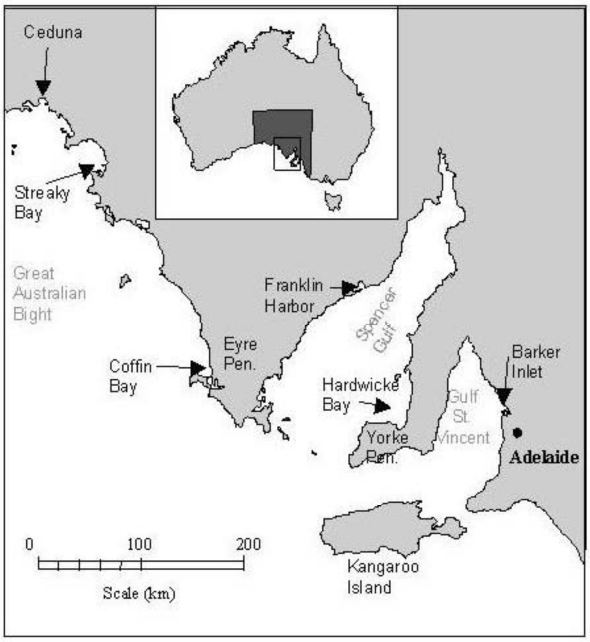

Fig. 2. Map of part of the South Australian Coast including the 4 post-larval sample sites and other geographic features. Initial release points for reverse modelling for Barker Inlet, Franklin Harbour, Coffin Bay and Streaky Bay are indicated by arrows. Inset: Position of the area relative to the Australian coast-line.

MATERIAL AND METHODS

Study area

The region of interest was the coastal waters of Victoria and South Australia (Figs 1 and 2). The continental shelf is relatively wide off South Australia (approx. 200 km), is narrowest off western Victoria (approx. 50 km) and from this point runs south along the west coast of Tasmania. The area off central Victoria is dominated by Bass Strait, consisting of a shallow platform, mostly about 70 m below sea level, flanked by 4–5 km deep ocean to the east and west and by land to the north and south.

Current patterns in the vicinity of the southern Australian coastline are largely influenced by weather systems. In summer, slow moving high-pressure systems are located south of the continent and track from the west to east; as a result, the southern Australian region experiences winds directed from the southeast. In winter, however, the high-pressure systems lie over the continent and cause a predominance of winds from the west (Lewis 1981; Schahinger 1987). The primary determinants of net water movement on the continental shelf are wind-driven currents and coastal-trapped waves (Middleton and Black 1994). Empirical measurements have confirmed that the south-east current flow along the continental shelf is strongest during winter (Hahn 1986).

Sample sites

Sampling for King George whiting post-larvae in Victoria was conducted in winter/spring 1995. Post-larvae were collected from Port Phillip Bay, Western Port and Corner Inlet (Fig. 1). In South Australia, post-larvae were collected in 1994 from Gulf St Vincent, Spencer Gulf, and Coffin Bay and Streaky Bay (Fig. 2). Sampling sites were semi-enclosed, protected areas that support shallow seagrass beds. Details of sampling sites and methods are presented in Jenkins *et al.* (2000a) and Fowler *et al.* (2000).

Laboratory method

To estimate larval duration, sagittal otoliths were dissected from post-larvae, mounted and polished, and increments counted by methods described in detail by Jenkins *et al.* (2000a) and Fowler *et al.* (2000).

Numerical modelling

The western boundary of the model grid was placed near Ceduna, South Australia (Fig. 2), using the boundary-condition techniques proved by Middleton and Black (1994). This involved adding coastal-trapped wave oscillations to the boundary sea levels using measured coastal water levels at Thevenard (Ceduna). Measured winds and sea levels from a range of locations were incorporated into the model and sea-level predictions were calibrated against field measurements (Jenkins *et al.* 2000a).

We used the three-dimensional hydrodynamic model 3DD (Black 1995) and dispersal model POL3DD (Black 1996). The three-dimensional hydrodynamic model had six depth strata, 0–4, 5–14, 15–34, 35–54, 55–74, and 75–6000 m. The model region was based on a grid of 10 by 10 km square cells, 178 cells east–west by 91 cells north–south. In the dispersal model the horizontal eddy diffusivity was set at 0.0015 $m^2 s^{-1}$. The period simulated was from 1 March to 30 November of each year. Larvae were modelled as neutrally buoyant and moving randomly throughout the depth range.

Post-larvae were represented in the model to simulate the actual larval advection based on estimated larval durations and arrival dates. To improve statistical reliability each larva was represented by 10 neutrally buoyant particles that were seeded at the mouth of a bay or inlet on their estimated day of arrival. Particles were then tracked backwards for their estimated larval duration in a "reverse" simulation to the point of hatching. The final (hatching) position was plotted in space for all particles.

To examine the possible influence of larval behaviour on advection pathways and predicted spawning areas, a diurnal vertical migration scenario was imposed on the particles in a second series of simulations. This involved a 'reverse' diurnal migration as has been found for *S. punctata* post-larvae in Port Phillip Bay (Jenkins *et al.* 1998). In this simulation, particles were randomly mixed within the upper 5 m of the water column in daylight, and within the upper 75 m at night.

RESULTS

In Victoria, estimates of larval durations from otoliths increased from west to east with a mean of approximately 120 days for Port Phillip, 130 days for Western Port, and 140 days for Corner Inlet (Jenkins *et al.* 2000a). In South Australia, mean larval duration increased over the sampling period, from approximately 90–115 days in June to 120–130 days in October (Fowler *et al.* 2000). There was also a trend for larval duration for post-larvae in the gulfs to be longer than for those in Coffin and Streaky bays (Fowler *et al.* 2000).

The mean current velocities and directions for the surface layer predicted by the hydrodynamic model are presented in Fig. 3. In the western part of the grid near the South Australian gulfs the currents are generally weak and non-directional. Along the western coast of Victoria, currents are stronger and are uni-directional from west to east. Strong currents run north along the western boundary of Bass Strait and turn to follow the coast from west to east through northern Bass Strait and eastern Victoria. A clockwise gyre is apparent within Bass Strait.

The areas from which the recruits to the individual Victorian bays are predicted to originate are shown in Fig. 4. For Port Phillip Bay the majority of recruits were predicted to originate along the coast from west of Cape Otway to Cape Jaffa (approximately 200 to 600 km). A very low number of recruits were predicted to originate along the western boundary of Bass Strait (Fig. 4). The simulation for Western Port recruits showed a similar pattern to Port Phillip recruits, although a greater number of recruits were predicted to originate along the western boundary of Bass Strait (Fig. 4).

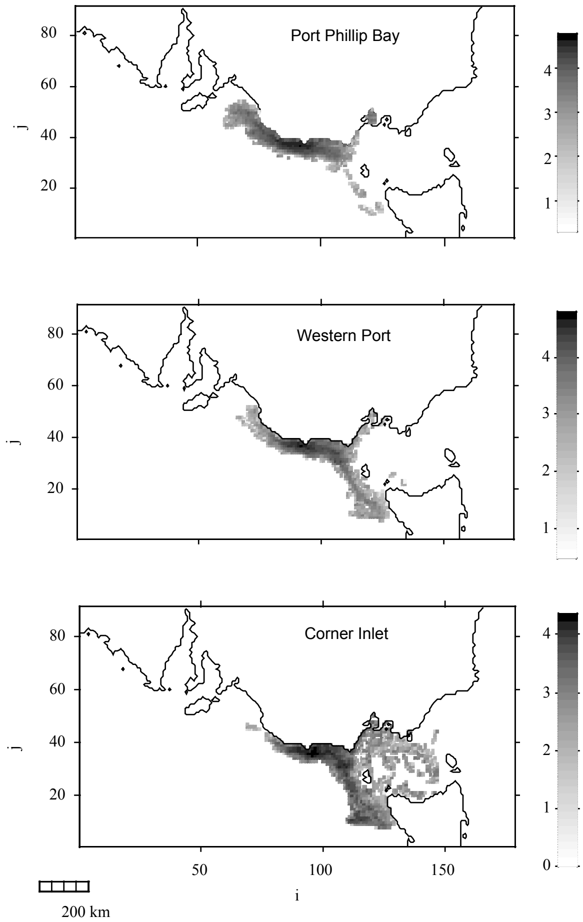

Fig. 4. Predicted source areas for post-larvae of King George whiting recruiting to Port Phillip Bay, Western Port, and Corner Inlet in 1995 based on reverse hydrodynamic modelling. Scale bar: log particle density; 'i': grid cell number from west to east; 'j': grid cell number from south to north.

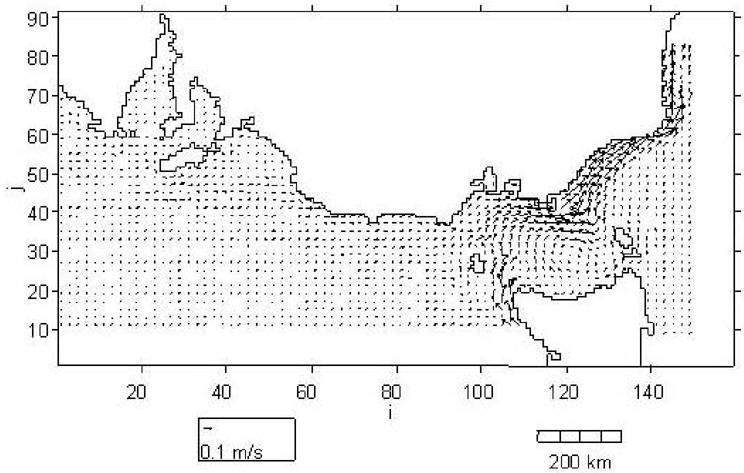

Fig. 3. Mean current velocities and directions for the surface layer predicted by the hydrodynamic model for the period 1 June to 1 October, in 1995.

The simulation for Corner Inlet recruits gave a markedly different pattern, with recruits not predicted to originate as far west along the coast, but showing high levels along the western boundary of Bass Strait and low levels in central and eastern Bass Strait (Fig. 4). For recruits predicted to originate in central and eastern Bass Strait, analysis of the advection pathways for the Corner Inlet simulation showed a pathway that looped around the western boundary of Bass Strait and northern Tasmania to central Bass Strait (Jenkins *et al.* 2000a). This latter pathway suggests that recruits in Corner Inlet could be derived from spawning on the nearby coast, with larval advection in a clockwise gyre down to northern Tasmania, northwards along the western boundary of Bass Strait, and eastwards along the Victorian coast to Corner Inlet. The majority of recruits, however, were still predicted to originate from western Victoria to south-east South Australia, approximately 500 km from Corner Inlet (Fig. 4).

For South Australian simulations the predicted dispersal was much more limited. For Barker Inlet in Gulf St Vincent the simulation was divided into early-, middle- and late-season recruits. As the season progressed the area from which recruits were predicted to originate extended from the middle to lower western side of the gulf to the western head of the gulf for late season recruits (Fig. 5).

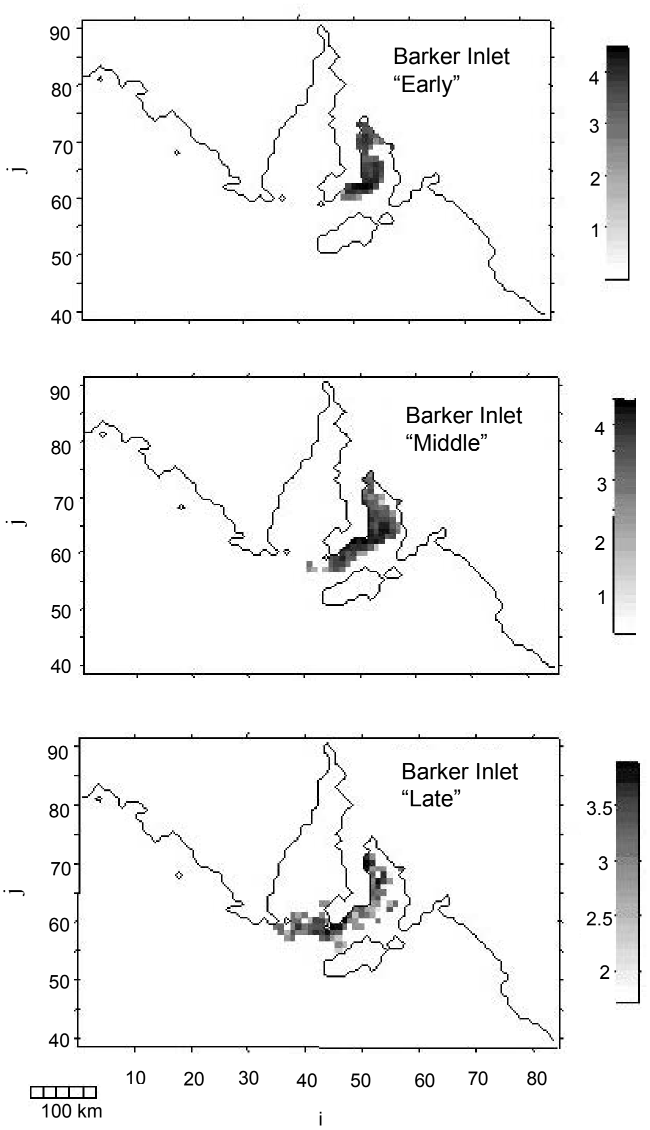

Fig. 5. Predicted source areas for post-larvae of King George whiting recruiting to Barker inlet during the 'early', 'middle' and 'late' parts of the 1994 recruitment season based on reverse hydrodynamic modelling. Scale bar: log particle density; 'i': grid cell number from west to east; 'j': grid cell number from south to north.

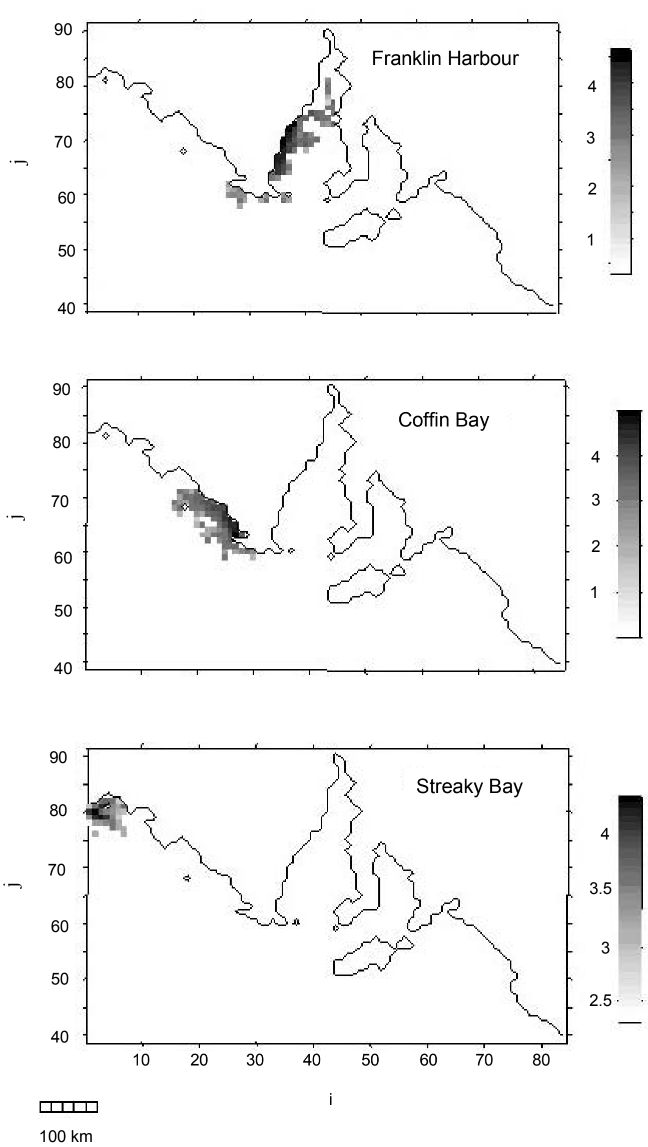

Fig. 6. Predicted source areas for post-larvae of King George whiting recruiting to Franklin Harbor, Coffin Bay and Streaky Bay in the 1994 recruitment season based on reverse hydrodynamic modelling. Scale bar: log particle density; 'i': grid cell number from west to east; 'j': grid cell number from south to north.

Dispersal distances ranged between approximately 50 and 150 km (extending up to 200 km for late-season recruits). Post-larvae at Franklin Harbour in Spencer Gulf were predicted to originate within 100 km of the recruitment site along the south-west coast of the gulf, but with some originating as far distant as up to 200 km at the western head of the gulf (Fig. 6). Recruits sampled at Coffin Bay and Streaky Bay were also predicted to originate locally, occurring primarily within 100 km of the recruitment site (Fig. 6).

The addition of a vertical migration scenario to the simulation for Port Phillip Bay, Victoria, had the effect of moving the predicted spawning area offshore, and reducing the spread of spawning in an east–west direction (Fig. 7). Vertical migration did not result in a quantitative shift of the spawning area along the coast (Fig. 7).

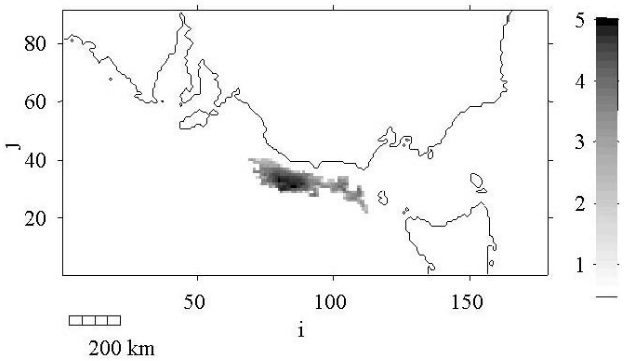

Fig. 7. Predicted spawning area of King George whiting for post-larvae from Port Phillip Bay in 1995 based on reverse hydrodynamic modelling including a diurnal vertical scenario applied to particles.. Scale bar: log particle density; 'i': grid cell number from west to east; 'j': grid cell number from south to north.

DISCUSSION

Understanding larval dispersal is key to the informed design of marine protected areas, given the relatively open nature of marine compared with terrestrial populations (Carr and Reed 1993). Although the openness of populations in marine systems may have been overemphasised in the past (Jones *et al.* 1999; Swearer *et al.* 1999; Cowen *et al.* 2000), it is likely that in many species at least a proportion of larvae undergo significant dispersal. Two of the most useful tools for investigating larval dispersal in fish are the ability to age larvae from otolith micro-increments (Brothers *et al.* 1976), and the simulation of current patterns in numerical hydrodynamic models (Black *et al.* 1993).

The combination of estimated larval duration and "reverse" hydrodynamic modelling has much potential in the design of MPAs, and may be particularly valuable where the aim of the MPA is fishery management. The method could be used to predict the source of recruitment for a proposed or existing MPA, and help in the design of MPA networks so they act as recruitment sources for each other rather than relying on recruitment from exploited areas (Carr 2000). Furthermore, as in the present study, the method could be used to suggest placement of MPAs so that sources of larvae that seed nursery areas can be protected (Roberts 1997; Allison *et al.* 1998).

In this study we found wide variation in dispersal potential across the geographic range considered. In Victoria, long-distance dispersal along the coast is predicted in the western part of the State where strong, unidirectional currents occur on the narrow shelf over the winter/spring period. In terms of MPA placement in this area, they would be unlikely to be self-recruiting for species with relatively long larval durations, and would be dependent on non-MPA areas for recruitment sources. A series of MPAs along the coast may allow for MPAs to receive recruits from those "upstream". In the specific case of King George whiting, one or a few MPAs on the west coast could supply larvae to all the major inlets of central Victoria.

Potential dispersal, however, was much more limited in South Australia, even though the estimates of larval durations were similar to those in Victoria. Generally, particles would be predicted to be sourced from the head regions of the gulfs and bays where recruitment occurred. The simulations suggested that areas such as the heads of the gulfs were characterised by recirculating gyres that tended to trap larvae for long periods. Studies have now shown that recirculation features can lead to unexpectedly high levels of larval retention (Black *et al.* 1991; Werner *et al.* 1997). The implication for the South Australian coast is that MPAs may be located in areas that are largely self-recruiting. This has the advantage that recruits are being sourced from a protected area (Allison *et al.* 1998). The disadvantage is that after a major perturbation of the area, replenishment by recruitment from outside areas would be less likely (Warner and Cowen 2002). Multiple MPAs would provide a hedge against this occurrence. In terms of King George whiting, unlike Victoria, where one or a few MPAs might protect the spawning source for a number of nursery areas, in South Australia individual sub-populations would tend to be sourced from different locations, and this would require a targeted network of MPAs along the coast.

Near Corner Inlet, Victoria, a third pattern of dispersal was predicted for long-lived larvae, that is, wide dispersal but recruitment occurring near the spawning source. This situation is more akin to that in South Australia in terms of the design of marine park networks, even though the dispersal distance is vastly different. The difference results from the size of the recirculation feature, with the Bass Strait gyre occurring on a scale of hundreds of kilometres. Results for Corner Inlet suggest that an MPA in the area, at least for species with a long larval duration such as King George whiting, may receive recruits that are locally produced and also a proportion that are produced from a long-distance source. For such species, this area would be particularly suitable for the siting of an MPA.

The spatial comparisons between Victoria and South Australia are potentially confounded temporally because recruitment data presented here were collected in 1995 in Victoria but were collected in 1994 in South Australia. However, when simulations were also run for Port Phillip Bay recruits collected in 1994 and 1989, these gave a prediction for the origin of recruits very similar to that found for 1995 (Jenkins *et al.* 2000a). Even though estimates of larval duration were longer for 1989, these were compensated for by weaker currents, suggesting a relatively consistent source area (Jenkins *et al.* 2000a). Major temporal changes would be expected, however, if summer spawning species were considered instead of winter. There is a major change in weather patterns between winter and summer when winds change from predominantly westerly to south-easterly, causing major changes to circulation patterns (Lewis 1981; Schahinger 1987).

The accuracy of predictions using these methods depends on a number of assumptions, and the limitations of the method must be recognised. Very importantly, the method gives an estimate of all possible spawning sources for a given settlement site, whether spawning occurs or not. Spawning may be spatially uneven or restricted and a minority of the potential spawning area may provide the majority of the recruits. For example, predicted spawning along the western boundary of Bass Strait seems unlikely, because most evidence suggests that spawning occurs relatively close to the coast (Hyndes *et al.* 1998; Fowler *et al.* 1999). Thus, the results of the modelling need to be interpreted in the context of all other life-history information, including that for the adult stage. It is also possible that the predicted spawning area would increase if additional settlement sites were examined, or if larvae that did not reach settlement were included. Additional techniques to refine the predictions would be most useful. For example, the inclusion of otolith microchemistry in the analysis may allow the prediction of larval source areas to be refined significantly (Swearer *et al.* 2002).

A second major limitation of the technique is that larvae are assumed to be transported passively. Although this assumption might be valid for some weak-swimming invertebrate larvae, the larvae of many fish species are known to undergo diurnal vertical migration (Neilson and Perry 1990) and, towards the end of the larval stage, are capable of directed swimming relative to currents (Leis *et al.* 1996). This may have been less of a problem for King George whiting than for some other species because even at the end of the larval stage their swimming abilities are relatively weak (Jenkins and Welsford 2002). Furthermore, dispersal modelling of late-stage King George whiting larvae in Port Phillip Bay has shown that prediction of dispersal and recruitment based on passive dispersal was highly accurate, and not improved by the inclusion of known vertical behaviours (Jenkins *et al.* 1997, Jenkins *et al.* 1999). The vertical migration scenario included in the present study gave a more defined source area than the passive model, probably because the more restricted vertical distribution meant that larvae experienced less vertical variation in current speed. The scenario also shifted the predicted source area offshore from the coast, which – given present knowledge of spawning areas – is less realistic than the passive case, suggesting that this scenario is somewhat removed from real behaviour. Incorporation of actual larval behaviour is likely to give a much more refined prediction of source areas than the passive case.

Finally, the scale of modelling will need to be appropriate to the dispersal characteristics of the species in question. For consistency, we have used the same resolution across the range; however, the more limited dispersal in South Australia lends itself to higher-resolution modelling over smaller areas. Nevertheless, the present model was able to resolve relatively small-scale retention features.

This paper uses King George whiting as an example of a species for which MPAs might be considered as a management option. It is usually suggested that sedentary species will benefit most from MPAs (Sladek Nowlis and Roberts 1999). However, some studies have shown that migratory species can also benefit (Bohnsack 1998; Roberts 2000), particularly where breeding and nursery areas are relatively fixed (Apostolaki *et al.* 2002). The King George whiting fishery shows high recruitment variation related to climate fluctuations (Jenkins *et al.* 2000b). To date, the fishery has proved relatively resilient, probably because most fishing is for sub-adults in bays and inlets, whereas the spawning adults that occur on the more exposed and less accessible coast are relatively lightly fished. Thus, the spawning individuals have received "natural" protection (Bohnsack 1998). If, however, the spawning individuals were to come under heavier fishing pressure in the future, then marine protected areas might be of benefit.

In summary, estimated larval duration from otolith micro-increments combined with reverse hydrodynamic modelling predicted spatial variation in dispersal that fell into three broad patterns. In western Victoria, strong uni-directional shelf currents meant that significant, long-shelf dispersal to the east occurred, suggesting that MPAs in the area would mostly receive larvae from areas to the west. For an MPA to receive larvae from a protected area, more MPAs would need to be located "upstream". In contrast, in South Australia and near Corner Inlet, Victoria, although at markedly different scales, retention mechanisms meant that MPAs may well be self-recruiting to some extent. Although this means that recruits may be produced in a protected area, there is also a greater risk from external perturbations, and therefore a network of MPAs might be desirable. If MPAs were used to manage King George whiting specifically, then only one or a few MPAs might be required to protect part of the spawning area that supplies a number of juvenile nursery areas in central Victoria. Conversely, in South Australia, a network of MPAs would be required to protect spawning areas of sub-populations. Overall, this technique shows considerable promise in the planning of MPA networks, and could be further refined by the addition of a "tagging" technique such as otolith microchemistry.

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MULTISCALE DECISION SUPPORT FOR AQUATIC PROTECTED AREA PLACEMENT

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Abstract

Successful placement of aquatic protected areas (APAs) not only relies on a myriad of local social factors relating to 'resource' users and jurisdictional boundaries, but also on the relevant-scale spatial distribution and movement of species, ecosystems and habitats for which protection is sought. Equitable protection can be viewed as a problem with many spatial scales, and conventionally for some there have been few data available. A data-mapping process (SimMap) was developed to support a large study on the effects of fishing on marine ecosystems ('The Sea Around Us' project; http://seaaroundus.org); this provides critical data at several scales, linking fine-scale ecosystem models (EcoPath/EcoSim/EcoSpace) in a nested fashion to whole ocean-basin, and even global distributions of taxa and oceanographic processes. This system supports the spatial ecosystem modelling used to evaluate the impacts of APAs, and also allows work within these ecosystems to be extrapolated over larger areas and allows investigation of temporal changes.

Keywords: fisheries, global, spatial, statistics, marine

INTRODUCTION

Most marine planners charged with the responsibility of delimiting aquatic protected areas (APAs) would not usually envisage using large-scale fisheries data in the decision-making process. Nor is this information considered useful to ecosystem modelers in the form in which it is usually supplied. The problem is one of mismatched scale. Fisheries data reported by national and regional commissions is often reported as port landings, with no spatial origin confirmed, or by statistical areas that are much too large to be directly useful. The problem of spatial precision can be addressed by new approaches that reduce the scale of available fisheries landing data using a process of spatial subtraction. Areas that do not fit with known biological or jurisdictional/access information are removed as potential catch locations from the statistical areas from which the catch is reported. After evaluating where catches could not have been taken within the reporting area on the basis of the distributional limits of the taxa landed, and the access agreements in effect between the reporting country and coastal states in the area, the catch can then be 'allocated' differentially to the remaining portion of the statistical reporting area on the basis of habitat suitability and primary productivity levels. This process results in relatively fine-scale maps of fisheries catches where only vague landings data officially exist.

In addition to problems of spatial scale, many areas of the world's oceans are outside the jurisdictional waters of coastal states, with no statistics available except those supplied voluntarily to the Food and Agriculture Organization (FAO) of the United Nations by fishing nations. This leaves many areas vulnerable to over fishing with newer technologies (Roberts 2002), and leaves critical habitats such as seamounts without representation in management processes because offshore 'high seas' areas usually have not been subject to governmental protection. Large-scale fisheries data made available from the FAO and regional bodies may be used to fill these gaps once the spatial allocation process has been completed.

METHODS

The example described here starts with global fisheries data available from FAO (FishStat). The production data-set (which does not separate capture landings from aquaculture production) starts in 1950. After 1970, a separate capture data series is available from the web (*http://www.fao.org/fi/statist/fisoft/FISHPLUS.asp*). There are, however, limitations to these data that need to be understood, if they are to be used. Firstly, the identity of the taxon reported can be vague, and 15% or more of FAO's world catch is currently reported only by larger aggregations such as 'miscellaneous marine fishes', rather than by species. Secondly, though regional bodies supply data by smaller statistical areas, the majority of the world's statistics are reported by large areas averaging 19 million square km in size

(Fig. 1). Thirdly, fisheries data are supplied voluntarily to the FAO and though not without error they represent the 'official' statistics. In some instances FAO staff must make educated breakdowns of data supplied by statistical areas or taxa.

As daunting as these limitations are, there are ways of overcoming them. The taxonomic identity of catches from reporting countries can often be deduced by the more detailed reports produced by neigbouring countries and by lists of common taxa known to occur in the area in a process referred to as 'taxonomic disaggregation' (Watson *et al.* 2001; Watson 2001). The process of identifying the spatial location of fisheries landing reports is a process we call 'spatial allocation' (Fig. 2) and will be discussed in more detail below. Once the catch has assigned to spatial cells by use of defined taxonomic groups it is possible to investigate and correct aberrations in the 'official' data. For example, computer models identified over-reporting by China in recent years (Watson and Pauly 2001; Watson 2001).

Marine planners should have access to the best possible time-series data rather than those constrained by official sanction or blurred by vague reporting schemes. While all means should be exploited to secure better, more comprehensive data, it is necessary to make better use of the existing data sources.

The spatial allocation process relies on supporting databases and rule-based procedures to locate reported landings from large statistical areas into the most probable distribution of catch amongst a global system of approximately 260,000 spatial cells measuring 30 minutes latitude by 30 minutes of longitude (Fig. 2). There are two main types of databases involved. The first relates to the global distributions of the reported taxa (be they by species, family or higher levels of aggregation). Fishbase (Froese and Pauly 2000), SpeciesDab from FAO (FAO 2000; *http://www. fao. org/ fi/statist /fisoft/SPECIES.asp*) and other sources provide general information on the distributional range of fish taxa based on depth, latitude, presence or absence by FAO statistical area, etc.

Secondly, it is also possible to use maps detailing proximity to critical habitats such as coral reefs or seamounts identified by the World Conservation Monitoring Centre and other sources to constrain potential catch areas. Information for invertebrates is also available from sources such as CephBase (http://www.cephbase.utmb.edu). In total, information was compiled on the distribution of all taxa reported in FAO landings statistics and hundreds of others reported by other agencies.

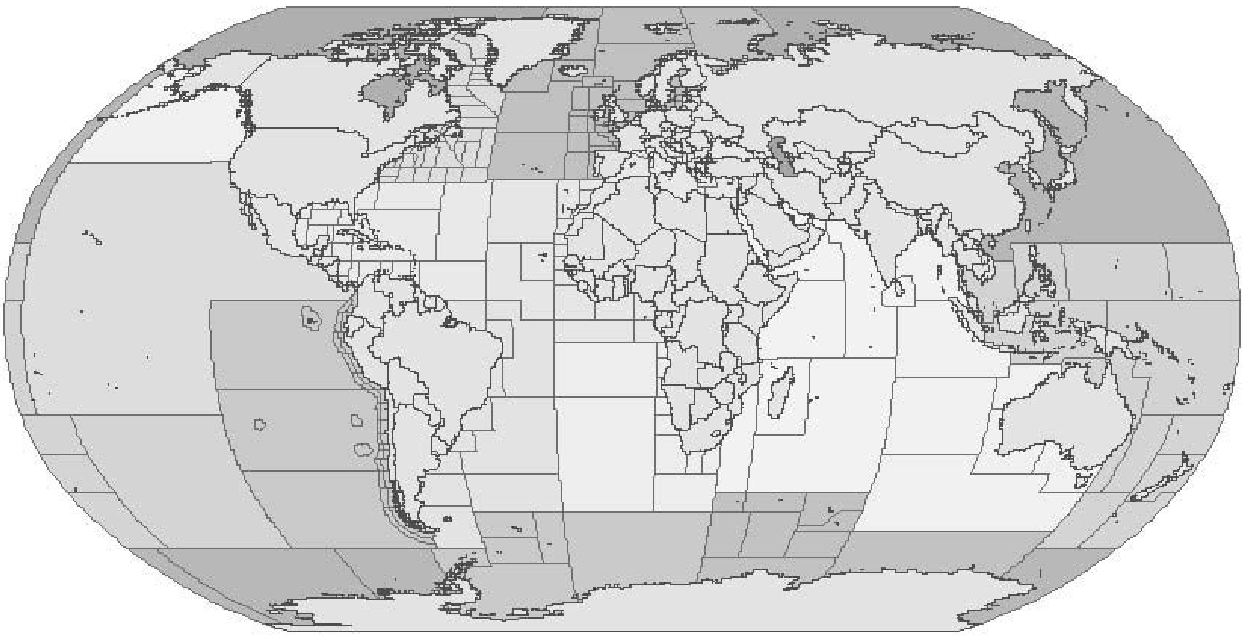

Fig. 1. Fisheries statistics mapped by FAO statistical areas (catch from 1950 to 2000 combined with darker areas representing higher catches).

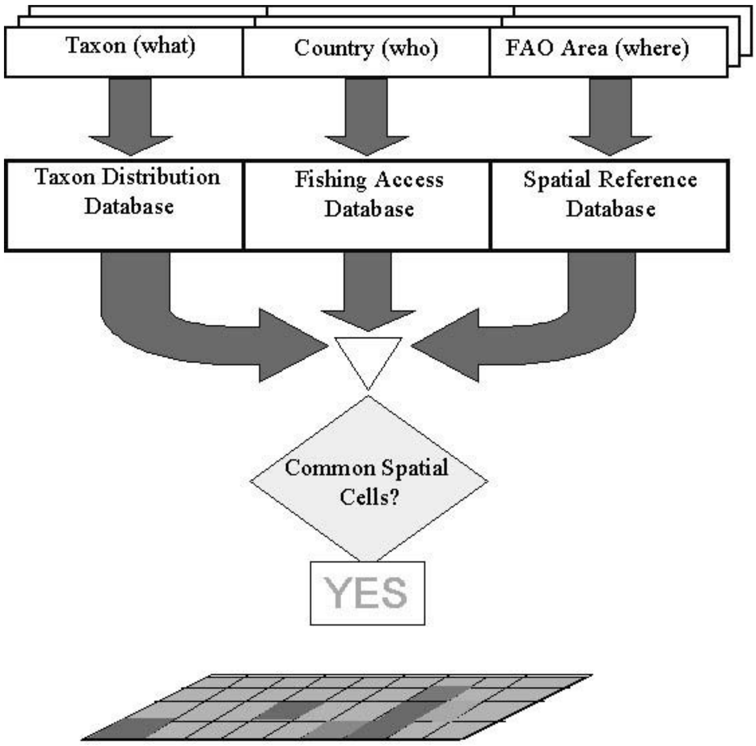

Fig. 2. Allocation of fisheries landings statistics to spatial cells within SimMap program.

More challenging even than compiling information on species distribution is gathering information on the fishing access arrangements and known areas of fishing access by the nations reporting statistics to FAO and other bodies. For some years, FAO has kept a registry of these agreements (Farisis) that documents many existing arrangements. Unfortunately, this database does not include all arrangements, because these are viewed as confidential by many parties and are not always reported in trade papers. Therefore, in addition, it has been necessary to research the known fishing patterns of all major maritime states in all literature available, grey and otherwise, in order to document where reporting nations may have fished. While some nations fish almost exclusively in their own waters, others, e.g. Japan, Russia and Korea, have bilateral fishing arrangements covering the Exclusive Economic Zone (EEZ) waters of other countries. Knowledge of these arrangements is very useful because most taxa are taken within continental-shelf areas that lie almost exclusively within the EEZs of nations.

In the spatial allocation process, the distribution of each taxon was used, in combination with the known access of the reporting fishing nation, to determine which part of the large statistical area reported in the landing statistics could have yielded the reported catch and in what proportion. By processing fisheries database records in this fashion, many inaccuracies were discovered in the reported distributions of the different taxa, in the identification of each taxon, and in the accounts of fishing arrangements. Each statistical landing record that could not be spatially allocated because of these inconsistencies was investigated and the underlying databases improved or the data in the landing report modified to reflect the most likely circumstance (these are carefully documented). This process of refinement is ongoing but at present nearly 99.5% of global landings (by weight) reported by FAO can be spatially allocated. That is, the reported landing can be distributed amongst the ½° spatial cells. FAO's capture database comprises more than 7000 records annually (more than 235,000 since 1950). Though computationally demanding, each is spatially allocated and composite maps of global landings are developed. The resulting spatial databases allow queries to produce maps of such attributes as trophic levels, catch composition or catch value.

RESULTS AND DISCUSSION

It has been possible to produce maps of global fisheries landings with a resolution of ½° latitude by ½° longitude (e.g. Fig. 3). At such scales, historical catch time series have been used to investigate catch reporting anomalies (Watson and Pauly 2001), to allow ecosystem models of the North Atlantic to be extrapolated to basin-wide studies of biomass and fishing intensities (Christensen *et al.* 2002), to look at diet overlaps between marine mammals and commercial fishing (Kaschner *et al.* 2002), to partition global coastal catches by large marine ecosystem (http://seaaroundus.org/lme/lme.asp), to examine changes in the trophic level of commercial landings (Pauly and Watson, 2003), and to estimate catch values by each nation's EEZ.

Marine planners face tremendous challenges. While commercial fisheries desperately attempt to expand and maintain profitability, their biological assets dwindle (Pauly *et al.* 2002). Public support for APAs to maintain marine systems has been growing but the hidden nature of marine resources makes denial relatively easy compared with terrestrial systems. Our impact on global climate, dangerous in its own right, is held by some to be the probable cause of biomass declines, and unfortunately to be sufficient justification for delayed action. Single-species management, long held to be sufficient to protect our marine resources, now appears incapable of capturing many critical processes such as trophic cascades. Unfortunately, 'management by ecosystem' has yet to be realized by most agencies. Ecosystem management is still believed by many managers to be too hard, unproven or unnecessary. Jurisdictional conflicts make decisions about large-scale marine closures difficult, and high-seas areas have little protection now that technologies exist for their exploitation (Roberts 2002). As always, it is easy to blame lack of political will or public ignorance for slow progress. Today, the signs of large-scale change can be seen in every ocean and sea. Trophic level decline in our capture fisheries is now well documented (Pauly *et al.* 1998; Pauly and Watson 2003). The challenge is for marine researchers to 'retool' quickly. Program suites such as Ecopath/EcoSim/Ecospace (Walters *et al.* 1999; Watson *et al.* 2000) now have many practitioners worldwide, and our experience and expertise is building.

Unfortunately, to provide managers with the tools to perform 'what if'examinations of management scenarios will require more than better tools. It will require better data. Many of us believe that there is precious little time to wait for this, all the worse because time-series data are required to provide likely trajectories for biomass and other vital measures. The approach described here provides a way of using currently available large-scale data to study spatial process on the scale of proposed APAs. These datasets were designed decades ago to track only broad economic development of nations' fishing industries, but must now serve other more demanding purposes. By coupling powerful existing databases on biological taxa with others dealing with fishing access arrangements, it is possible to improve the value of existing fisheries landing data, and to provide input of the type that ecosystem modelers urgently need to provide managers with vital information required for planning. The processes of taxonomic disaggregation and spatial allocation available within the SimMap program will assist by sharpening the spatial resolution of landings data sufficiently to be useful in the smaller-scale management decisions being investigated in the context of APAs.

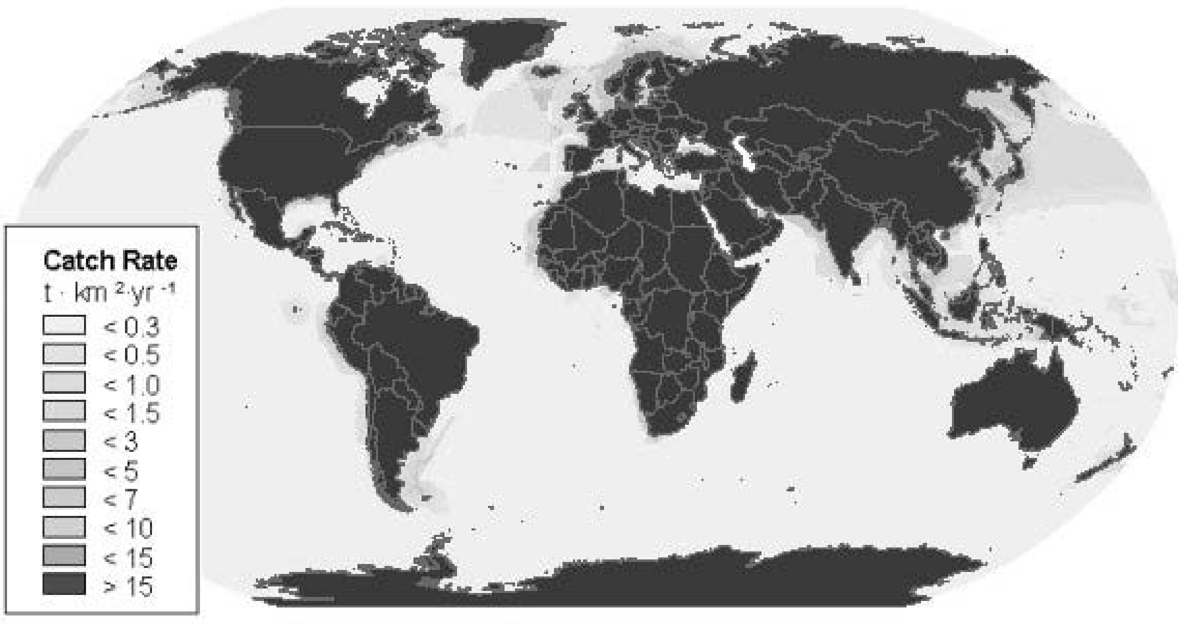

Fig. 3. Catch Rates (all taxa) based on spatial allocation of FAO fisheries data for 1999.

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DECLARATION OF MARINE PROTECTED AREAS – THE CASE OF THE BALLENY ISLANDS ARCHIPELAGO, ANTARCTICA

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Abstract

Historically, the protected-areas system in the Antarctic Treaty area has focussed on terrestrial and near-shore areas. No general marine protected areas exist. However, now that Annex V (Area Protection and Management) to the Protocol on Environmental Protection to the Antarctic Treaty has entered into force, it is possible to designate any marine area as either an Antarctic Specially Protected Area, or an Antarctic Specially Managed Area. This paper explores the processes for designating a marine protected area under the Protocol, and the role of the Convention for the Conservation of Antarctic Marine Living Resources, which has a mandatory role in such a proposal.

This paper uses the Balleny Islands archipelago in the northern Ross Sea as a case study to examine the options for establishing a marine protected area in Antarctica. Until now, only one small island (Sabrina) in the group has been accorded protected area status. The issues posed here are relevant to marine protected areas anywhere in Antarctica: the relative claims of access for marine harvesting, tourism and scientific research; the arguments for restricting access to secure environmental or scientific values; and the question of appropriate size of the area, etc.

Keywords: Antarctica, Marine Protected Areas, Balleny Islands, Antarctic Treaty, CCAMLR, Ross Sea, marine biodiversity, ASPA, ASMA

INTRODUCTION

By virtue of its remoteness and harsh natural conditions Antarctica (the continent, islands and marine area bounded by the Antarctic Convergence), constituting some 10% of the surface of the earth, has remained one of the least modified parts of the earth. Yet, that historic isolation and inhospitable climate can no longer be relied on to ensure that Antarctica will remain in its present state. There is growing evidence that global climate changes are affecting the Antarctic environment (Anisimov and Fitzharris 2001), and an increasing awareness of the environmental impacts of science, tourism and marine harvesting activities on and around the continent (Hansom and Gordon 1998; Waterhouse 2001).

Since 1964 there has been an Antarctic protected areas system under the auspices of the Antarctic Treaty. The system was initially based upon two categories: Sites of Special Scientific Interest (SSSIs) and Specially Protected Areas (SPAs). Subsequently, other categories were added, and SSSIs and SPAs were in a few instances able to include marine areas on the margins of their core terrestrial areas. A capacity to extend some level of protection to marine areas was, however, granted under two later Antarctic agreements – the 1972 Convention for the Conservation of Antarctic Seals (CCAS), under which Seals Reserves may be designated, and the 1980 Convention for the Conservation of Antarctic Marine Living Resources (CCAMLR), which enables the establishment of CCAMLR Ecosystem Monitoring Programme (CEMP) sites. Only a few of each have been designated – and these only for the narrow purposes that the names suggest.

As a result, the Antarctica protected areas system over the past forty years has been largely terrestrially focussed. It has also been driven by the specialist interests of individual scientists, and has tended to secure protection of single species and local assemblages of immediate interest to these scientists. This approach has lead to a largely *ad hoc* system of protection of an extremely small percentage (about 0.003%) of the Antarctic continent and the Southern Ocean below latitude 60°S.

A recent review of knowledge describing the biodiversity of the Ross Sea region (Bradford-Grieve and Fenwick 2001, 2002) revealed a paucity of information for all marine habitats. Many of our assumptions about the biodiversity of the Ross Sea are based on startlingly scant scientific information. However, recent research has clearly indicated that, for at least the benthic fauna, there is high species diversity (Page *et al.* 2002). There are presently few threats to marine biodiversity in the Ross Sea. However, there is increasing marine activity in the region via science activities, vessel-based tourism and an exploratory fishery for toothfish (Waterhouse 2001).

THE LEGAL FRAMEWORK FOR DESIGNATING MARINE PROTECTED AREAS IN ANTARCTICA

Two international agreements are relevant for the designation of marine protected areas in the Southern Ocean, the Protocol on Environmental Protection to the Antarctic Treaty and the CCAMLR. Traditionally, CCAMLR has dealt with marine issues, while the Antarctic Treaty Consultative meetings (ATCM), including in the past decade implementation of the Environmental Protocol, have focused on the terrestrial environment.

The overall purpose of the Antarctic Treaty is to encourage the peaceful use of Antarctica as a giant scientific laboratory; it entered into force on 23 June 1961. Forty-five nations, representing about two-thirds of the world's human population, abide by the treaty, either as consultative parties or as observers. However, some key maritime nations are not party to the Treaty, for example Panama and Mauritius. The Treaty covers the area south of 60°S and encompasses around 10% of the world's land surface and 10% of the earth's oceans (approximately 85 million km^2). The ATCM operates a consensus-based system whereby all consultative parties must agree to any resolutions, measures or decisions.

The CCAMLR entered into force in 1982; all signatories are entitled to be Members of the Commission, which at present has 24 Members. CCAMLR, like the ATCM, operates on a consensus basis. The Commission is primarily concerned with the rational use and management of living resources in the Southern Ocean; in doing so it strives to follow both a precautionary and an ecosystem approach to management (CCAMLR 2001). CCAMLR applies to all marine living resources south of the Antarctic Polar Front at about 50°S (except seals south of 60°S and all whales); it represents an area of approximately 35 million km^2. Several working groups provide advice to the CCAMLR Commission, including the Scientific Committee and the Working Group on Environmental Monitoring and Management (WG-EMM). While CCAMLR could designate marine protected areas via a Conservation Measure pursuant to Article IX of the Convention, so far it has only done so in the restricted context of CEMP sites.

In 1991 as part of the new Protocol on Environmental Protection to the Antarctic Treaty, the protected-area system was revised. This revision – "Area Protection and Management", Annex V to the Protocol – finally entered into force in 2002, some four years after the Protocol and its first four Annexes. Annex V creates two categories for protection: Antarctic Specially Protected Areas (ASPA), and Antarctic Specially Managed Areas (ASMA). Critically for our purposes, both categories may include "any marine area".

An ASPA can be designated to protect outstanding environmental, scientific, historic, aesthetic or wilderness values, any combination of those values, or scientific research. ASPAs will subsume previous SSSI and SPA designations. An ASPA establishes a legally binding obligation on states to regulate activity and to require a permit for entry. Activities within an ASPA can be strictly controlled to protect areas from human impact and to ensure that any environmental impacts are minimised.

An ASMA may be designated to assist in planning and coordination of multiple activities. No permit is required for entry. The Management Plan can be used as a guide, but use of it to control activities is limited because there is no legally enforceable requirement for a prior permit. An ASMA may have any number of ASPAs nested within it. There are at present no ASMAs designated under Annex V.

The Treaty's Committee on Environmental Protection (CEP) considers all ASPA and ASMA Management Plans submitted. In formulating its advice for the ATCM, CEP must take into account any comments on Management Plans provided by the Scientific Committee on Antarctic Research (SCAR). Annex V stipulates that no marine area can be designated as an ASPA or an ASMA without the prior approval of the CCAMLR.

THE BALLENY ISLAND PROPOSAL

In 1999 New Zealand put forward a proposal to establish an ASPA around the Balleny Islands. The goal of New Zealand's proposal was to create an integrated marine/terrestrial biodiversity reserve including and around the Balleny Islands.

The Balleny Islands archipelago is an island chain, 160 km long, in the northern Ross Sea NNW of Cape Adare in northern Victoria Land (Fig. 1). The archipelago is orientated NW–SE, between 66°15'S and 67°10'S, and between 162°15'E and 164°45'E, and it straddles the Antarctic Circle. The archipelago consists of 6 islands, 3 large (Buckle, Young and Sturge Islands) and three small (Row, Borradaile and Sabrina)(Fig. 2). Volcanic in origin, the islands rise sharply from the ocean floor, with depths of 2000 m being found within 5 nautical miles of the coast. All the islands are thickly ice-covered.

The only oceanic islands in the Pacific section of the Southern Ocean, the Balleny Islands are distinct from any neighbouring areas. They are a rare oasis of land bisecting the Antarctic Divergence, and their position is far enough north to be directly in the path of circumpolar ocean currents. Consequently, their presence creates upwellings, which bring nutrient-rich deep water to the surface; this in turn makes the area biologically productive (Knox 1994).

Only one small island in the group has been accorded protected-area status: Sabrina Island, with an area of around 0.4 km^2. The designation of Sabrina as an SPA in 1966 was on the grounds that the Balleny Islands support a flora and fauna that reflects many circumpolar distributions of the latitude, and that Sabrina provides a representative sample of that flora and fauna.

The key objectives of the proposal to expand the protection to the entire archipelago were as follows: to avoid degradation of the values of the area by preventing unnecessary human disturbance; to preserve the natural ecosystem as a reference area; to contribute to the protection of biodiversity in the Ross Sea region; to allow for appropriate scientific research; to minimise the risk of unwanted species introductions; and to allow for visits for management purposes. To achieve these objectives, the proposal included a substantial marine area around the islands. The proposal was based on the premise that the marine ecosystem, and much of the associated biota, of the Balleny Islands is unique to the geographic area.

Fig. 1. Ross Sea Region (taken from Waterhouse, E.J. (Ed) 2001. Ross Sea Region 2001: A State of the Environment Report for the Ross Sea Region of Antarctica. New Zealand Antarctic Institute, Christchurch, New Zealand.)

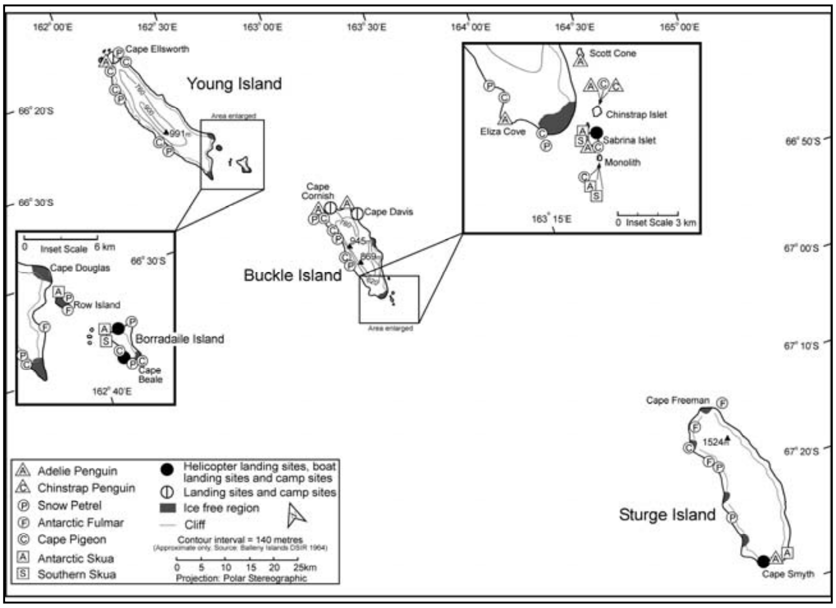

Fig. 2. Balleny Islands (courtesy of New Zealand Antarctic Institute)

Certainly, seabird diversity exceeds any other site in the Ross Sea region (Robertson *et al.* 1980). In total, 20 species of seabird have been reported from the Balleny Islands (Bradford-Grieve and Fenwick 2002), with at least seven of the species having been recorded as breeding, including a small colony of chinstrap penguins (MacDonald *et al.* 2002). This colony represents the only site in 264° of longitude where chinstrap penguins breed.

Between 1899 and 1992, only 79 stations were sampled in the waters around the Balleny Islands from a mere nine collecting events (Bradford-Grieve and Fenwick 2002). A review of the available information from those events suggested that rather than being unique and special, the marine biota of the Balleny Islands is representative of that found in equivalent habitats; however, much of the material collected during those surveys has never been processed (Bradford-Grieve and Fenwick 2002).

However, a more recent survey of the benthic biodiversity of the Ross Sea, including the Balleny Islands, suggests that at least the benthic macrobiota of the Balleny Islands is somewhat distinct from that found in the coastal waters around Cape Hallett, northern Victoria Land and the Possession Islands in the western Ross Sea (Page *et al.* 2002). Six dredge stations around the Balleny Islands resulted in the collection of 151 benthic species, of which 90 were new records for the archipelago and 12 were new or previously undescribed species (Page *et al.* 2002). Video footage recorded from two small reefs to the south of Sabrina Island revealed isolated patches of high diversity and abundance of benthic macrobiota in areas protected from episodic iceberg scouring by their position on the sides of the reefs (Page *et al.* 2002).

In addition to the ecological, scientific and aesthetic values of the archipelago, the New Zealand proposal identified the low level of human impact as an important reason for special protection of the islands and the surrounding marine area. There are no established stations on any of the islands, and landing on the islands is extremely difficult. Although there is interest from the tourism industry in visiting the islands, the present scale of tourist-industry activity is limited in the region (Waterhouse 2001). There is no known fishing activity around the archipelago. The proposal, therefore, would have little impact on present human activities in the region.

Over the past three years, various drafts of the New Zealand proposal have been considered by both the CEP and the CCAMLR. The boundary and extent of the area to be included in the ASPA has been one of the key variables in the proposal, ranging from 10 nmiles to 50nmiles from the mean low-water mark of all islands. To date, no agreement has been reached within these fora on the proposal. At present there is no proposal under consideration.

WHY DID THE PROPOSAL NOT SUCCEED?

The Balleny Islands proposal was the first ASPA to be proposed with a substantial marine component. As such, it was a test case for both the process and the substance of a proposal of this nature, and raised several issues related to the identification and designation of marine protected areas in Antarctica.

Probably the most important issue that the proposal highlighted was the lack of process within both CCAMLR and ATCM to assess and approve proposed management plans for marine areas to be designated under Annex V. Under Annex V, any area, including any marine area, may be designated as a protected area (Article 3(1)). Any party to the Environmental Protocol, the CEP, SCAR, or the CCAMLR Commission can propose areas for protection (Article 5(1)).

New Zealand submitted the Balleny proposal to CCAMLR, where both the Scientific Committee and the WG-EMM considered it. However, there was no clear understanding of the role of each of those groups in providing advice to the CCAMLR Commission, and how CCAMLR would "approve" such proposals. Nor was there any clarity on what aspects of the proposal the groups should provide advice on, and the timeframes within which advice should be provided.

Consequently, it took well over a year from receipt of the first proposal for CCAMLR to finally make a recommendation. Much to New Zealand's frustration, CCAMLR recommended that New Zealand consider resubmitting the proposal as an ASMA rather than an ASPA. Meanwhile, New Zealand has kept a holding pattern at the ATCM because a proposal cannot be considered within that forum until there is approval for the marine component from CCAMLR.

Alongside the process issues, opponents of the proposal raised questions about the substance of the proposal. They sought clarity on the values to be protected and the objectives of the proposal, and argued that more scientific information was needed about the islands, in particular the marine environment. The boundary issue was hotly debated and seemed to be the major stumbling block in reaching any consensus.

Issues such as right of free passage and access for both tourism and marine harvesting were widely discussed, and were points of disagreement. There was also considerable debate as to whether an ASPA was at odds with the CCAMLR concept of rational use. Opponents of the proposal argued that the marine area around the Balleny Islands was an important area in respect to potential future fisheries and that the designation of an ASPA would limit future rational use of the marine living resources.

As a result, there is still no consensus on the designation of the Balleny Islands as an ASPA. However, in keeping with the spirit of the proposal, New Zealand (for each of the past three years) proposed and had accepted a conservation measure through CCAMLR, which prohibits toothfish fishing within 10 nautical miles of the island chain. In addition, New Zealand fishing vessels operating in the exploratory toothfish fishery in the western Ross Sea have had as a condition of their permit a closed area of 50 nmiles around the archipelago. South Africa also voluntarily applied that provision in the 2000/2001 fishing season.

However, this may have inadvertently fuelled suspicion over New Zealand's true motives for the proposal. There has been a perception that the main underlying reason for the proposal was to protect Patagonian and Antarctic toothfish (*Dissostichus eleginoides and D. mawsoni*) from the potential effects of overfishing (Harris 2001). This lead to confusion over whether the proposal was in fact intended as fisheries management tool, which should rightly be dealt with under the auspices of CCAMLR rather than the Treaty.

WAY FORWARD

Given the lack of consensus, New Zealand has gone back to the drawing board to develop a three-pronged strategy for progressing the proposal. Key elements of the strategy are: to increase understanding about marine protected areas (MPAs); to resolve process issues related to the designation of marine protected areas within the Antarctic treaty system; and to recast the Balleny proposal within this context.

To increase understanding of MPAs, the strategy identifies three main actions. The first is to host and attend appropriate meetings and conferences with relevance to MPAs both within the Southern Ocean and globally. The second action is the development of a database containing key publications and documenting other MPA proposals of relevance. The final action is to pursue opportunities to advocate for marine protection within the wider international community (including the science community and the fishing industry.

To work towards a resolution of the process issues related to the designation of MPAs, New Zealand will work cooperatively with other countries to resolve the process issues within CCAMLR. New Zealand will also encourage other countries to take a lead in resolving key aspects of marine protected area issues within the Antarctic Treaty System and the CCAMLR. Critically, New Zealand will support other initiatives for specific MPA proposals in Antarctica.

New Zealand is developing a revised proposal for the protection of the Balleny Islands archipelago. In doing so it will review the values to be protected and the specific objectives of the protection, particularly in light of recent research findings. These are also relevant to the consideration of the size of the area for protection, along with scientific merit, application of the precautionary approach and the practical issues of ease of monitoring and compliance. Options for protection will also be identified, including the designation of the area as an ASMA containing a number of smaller embedded ASPAs.

RECENT PROGRESS

In May 2002, CCAMLR outlined a process with timelines for consideration of proposals. Proposals submitted to the CCAMLR Secretariat will be immediately forwarded to the Scientific Committee, relevant issues will be considered by the Working Group EMM and the Working Group on Fish Stock Assessment.Advice will be developed by the Scientific Committee for consideration and decision making by the CCAMLR. Finally, discussions and decisions will be reported to the ATCM. It is intended that proposals will be reviewed within one calendar year of receipt. However, there are still unresolved issues relating to the timing of proposals between CCAMLR and ATCM.

Significantly, 2002 will see four proposals for MPAs go forward to both CCAMLR and the ATCM. The Italian proposal for an ASPA in Terra Nova Bay, Ross Sea, is a new proposal. The proposals for Bransfield Strait and Dallmann Island in the Antarctic Peninsula region and Cape Royds in the Ross Sea are existing SSSIs that are being redesignated as ASPAs under Annex V. These proposals will test the new CCAMLR process.

Since the original proposal for a Balleny Island ASPA was submitted, New Zealand has developed increasingly strong domestic policy related to the protection of the marine environment. In February 2000, The New Zealand Biodiversity Strategy (NZBS) was launched. The purpose of the NZBS is to "establish a strategic framework for action, to conserve and sustainably use and manage New Zealand's biodiversity" (Department of Conservation 2000).

The NZBS contains a number of objectives directly regarding the protection of the marine environment; including a target of protecting 10% of New Zealand's marine environment by 2010 through a network of representative MPAs. This commitment to marine protection extends to Antarctica, with a stated action of the NZBS to 'advocate for the conservation and sustainable use of marine biodiversity in areas subject to international jurisdiction, including the Ross Dependency and other Antarctic areas" (Department of Conservation 2000).

As an action under the NZBS, the Department of Conservation and the Ministry of Fisheries have been jointly developing a Marine Protected Areas Strategy for New Zealand. This strategy will provide for the operational aspects of the NZBS, including defining what is meant by the term 'Marine Protected Area'. The definition of the term differs in a New Zealand context to that applied in many other countries, the key difference being that MPAs in New Zealand are not created as fisheries management tools, they are created to protect biodiversity values.

In June 2002 the New Zealand Government released its Statement of Strategic Interest for Antarctica, which commits New Zealand to the conservation of the intrinsic and wilderness values of Antarctica and the Southern Ocean. This sentiment is carried through in New Zealand's advocacy for the establishment of MPAs in the Ross Sea region of Antarctica. These policy developments have provided useful principles upon which to recast the basis and justification for the Balleny Island proposal, including the values to be protected and the specific objectives.

CONCLUSIONS

The geopolitical policy framework for the designation of ASPAs and ASMAs with a marine component is complicated. There are two international agreements (ATCM and CCAMLR) with different foci, advised by numerous working groups and committees.

Despite the difficulties with the Balleny Island proposal, the Protocol and CCAMLR have the potential to be quite creative in the development of marine protection in the Southern Ocean. However, the potential of ASMA and ASPA to provide protection has yet to be fully explored. In particular, the Balleny Islands provides an opportunity for innovative application of these two potentially powerful management tools. New Zealand intends to continue to pursue the designation of the Balleny Islands as an integrated terrestrial and marine protected area under the Antarctic Treaty System.

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DESIGNING REPRESENTATIVE AND ADEQUATE MPAS IN A STRUCTURED ENVIRONMENT

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Abstract

Knowledge of the spatial structure of marine ecosystems and the processes that affect this structure are required before MPAs can be considered representative and adequate. Spatial structure can be measured on many different scales. This is illustrated for the southeast Australian continental shelf ecosystem, which is structured by: oceanography; geology; plankton, invertebrate and fish communities, fisheries and invasive marine species at different scales in the space and time domains. The design of representative MPAs first requires a clear enunciation of the management aims so that a suitable spatial scale (or scales) can be identified. Once the representative scale(s) has been determined, a hierarchy of management measures, with objectives and performance measures at each scale will be required if the MPA is to be adequate.

Keywords: adequate, representative, invasive marine species, fisheries management, ecosystem structure and function

INTRODUCTION

Setting aside a portion of the environment for conservation purposes has a long history on land, where for centuries land has been set aside for game parks and, more recently, natural parks, a wide variety of conservation areas, and smaller areas including streamside buffer zones and even hedgerows to reduce soil erosion. The values and functions of land set aside in this manner have been clearly identified. There is less of an exclusive relationship between habitat and species under water than on land. In contrast to terrestrial vertebrates, most species of fish are carnivores with highly flexible diets and more flexible growth rates, suggesting that a variety of areas can provide suitable habitat (Larkin 1978). In addition, the frequently diverse life histories of marine organisms, which often include a widely dispersed pelagic stage followed by settlement or recruitment to an area that can be quite distant from that of the parents, means that there is not as clear a link between habitat and organisms as there is on land. Consequently, it is harder to identify areas of the seabed that have high biological conservation value.

A perceived solution to this lack of clearly identified conservation values is to set aside representative areas. Australia is implementing a National Representative System of Marine Protected Areas (NRSMPA). The primary goal of the NRSMPA is to commit jurisdictions to establishing and managing a comprehensive, adequate and representative system of MPAs (ANZECC TFMPA 1999a). 'Comprehensive' implies recognising the full range of ecosystems; 'adequate' implies developing MPAs of sufficient size and appropriate spatial distribution to ensure the ecological viability and integrity of populations, species and communities; and 'representative' implies that selected areas should reasonably reflect the biotic diversity of the marine ecosystems of which they are part.

A hierarchy of scaled ecological units has been proposed for the NRSMPA. These scaled ecological units are: bioregion, ecosystem, habitat, community/population and species/individual (ANZECC TFMPA 1999b). Twofold Shelf, a 32,198 km^2 bioregion, is one of 60 bioregions identified in the NRSMPA. Only a small percentage of the Twofold Shelf bioregion has been protected to date. This bioregion is the area of greatest fishing effort in Australia's most important domestic finfish fishery – the South East Fishery (SEF) – where more than 100 species of teleosts and elasmobranchs are caught and marketed, although only 20 species or closely related species-groups, forming >80% by weight of total catches, are managed directly. MPAs will directly affect this fishery.

We have previously suggested (Williams and Bax 2001), on the basis of results of extensive surveys of fish and invertebrate communities in the Twofold Shelf bioregion, that to be representative, distinct management units would need to represent the distinct fish and invertebrate communities found there. In this paper, we first review the spatial structure of the Twofold Shelf bioregion, concentrating on what would be necessary for a representative system of MPAs in this region, and then address the question of what additional management measures would be necessary for an adequate system of MPAs that would assist the ecological viability and integrity of populations, species and communities.

Spatial structure in the Twofold Shelf bioregion

The Twofold Shelf bioregion is an exposed and current-swept area (Fandry 1983; Morrow and Jones 1988) that extends to about 200 m depth on the south-eastern Australian continental shelf (Fig. 1). The area is a faunal transition zone, or biotone, containing cool- and warm-temperate faunas – a major cross-shelf faunal disjunction occurs near Cape Howe, as the shelf broadens and orientates more to the east–west, coinciding with an overlap of temperate and subtropical currents.

Water masses

Three main water masses affect the bioregion (Figs 2a and 2b): the East Australian Current (EAC) and its eddies flow southwards, carrying warm, high-salinity, nutrient-poor water; high-salinity, cool Bass Strait water flows eastwards driven by the prevailing westerly winds; low-salinity, cool subsurface sub-Antarctic water flows slowly from the south. There is strong seasonality in the presence of the water masses in the study region (Figs 2a and 2b show spring water mass structure at the surface and at depth). Nutrients are generally low in the study area, making the flow of nutrient-rich sub-Antarctic water from the slope onto the outer-shelf distinctive. The mechanisms that drive this deep upwelling – an interaction of EAC eddies, wind and topography – result in an uneven and seasonally variable enrichment leading to small-scale variability in productivity (Newell 1961; Cresswell 1994; Church and Craig 1998); deep upwelling is particularly evident at the Horseshoe, the largest arm of the Bass Strait canyon.

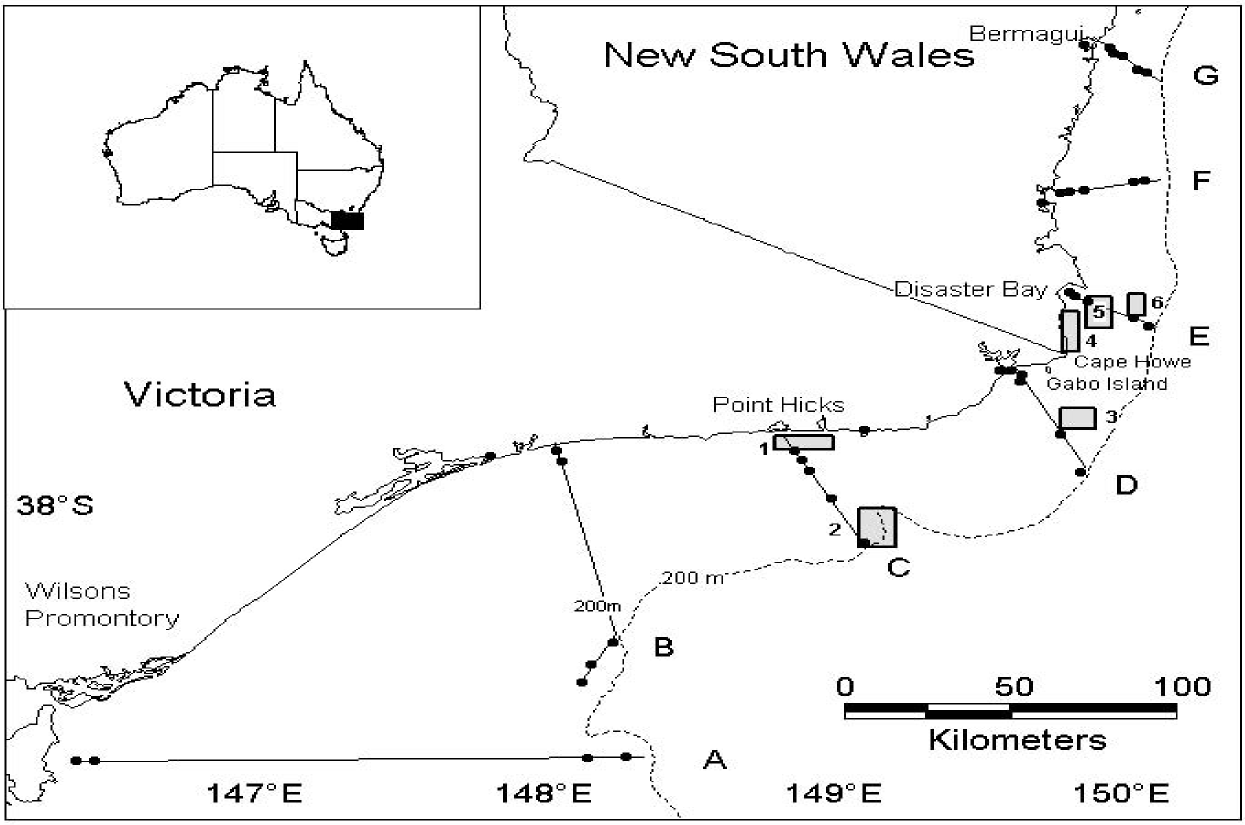

Fig. 1. Study area on the Twofold Shelf bioregion on the south-eastern Australian continental shelf, showing transects (lines), positions of depth-stratified stations (dots) and sites of intensive sampling (boxes) for the surveys from 1993-1996. Intensive sampling areas are: (1) Point Hicks; (2) The Horseshoe; (3) Gabo Reef; (4) Black Head; (5) Disaster Bay; and (6) Big Gutter.

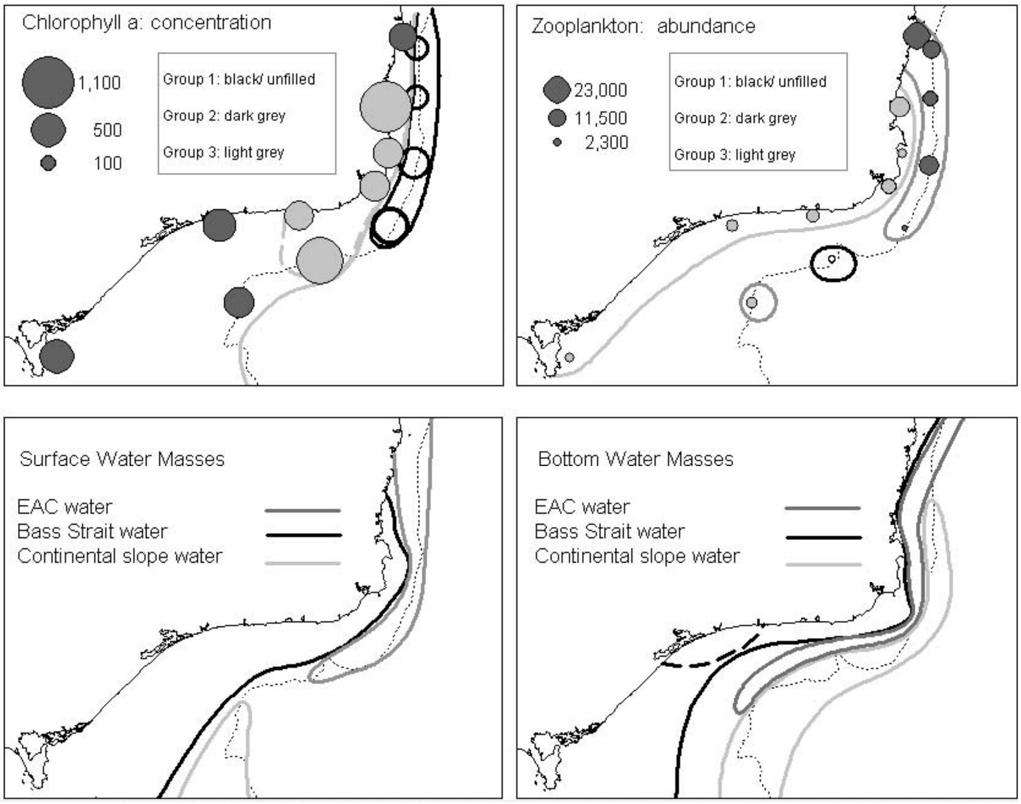

Fig. 2. Summary of oceanography in spring 1996, showing schematic of water-mass structure for (*a*) surface and (*b*) bottom waters based on multivariate analysis of *in situ* oceanographic measurements of salinity, temperature, nutrients and dissolved oxygen; (*c*) phytoplankton groups based on multivariate analysis of photoreactive pigments, and; (*d*) zooplankton groups based on multivariate analysis of zooplankton (from Bax and Williams 2000).

The seabed

The seabed today results from a complex interaction of marine and terrestrial processes superimposed on tectonic processes that stretch back hundreds of millions of years, and modified by previous climate changes (Bernecker *et al.* 1997). Sea levels have been at approximately their present level for only the past 6000 years. When sea levels were lower, presently submerged inner-shelf rock formations were exposed to karst weathering, leading to their very irregular topography of pinnacles and depressions, the latter becoming filled with sediment when sea levels subsequently rose.

The Twofold Shelf bioregion consists mainly (89%) of massive sediment plains ('soft grounds'), with dispersed patches of reef, bedrock and consolidated sediments ('hard grounds') making up the remaining 11%. Additional smaller outcrops of reef (biogenic and bedrock) and patches of cemented hard grounds occur, particularly at the shelf-break, but have not been mapped (Bax and Williams 2001).

Biological communities

Results reported below are derived primarily from a 5-year study of the Twofold Shelf bioregion (Bax and Williams 2000).

PRIMARY PRODUCTION

Primary production in the Twofold Shelf bioregion is principally planktonic in origin, with only very limited contribution from macroalgal growth in shallow waters (Bax *et al.* 2001). Compared with other continental shelf regions, the waters of the shelf off south-eastern Australia have low chlorophyll concentrations – highest recorded value of chlorophyll *a* in a spring bloom in this area was 1.3 $\mu g\ L^{-1}$(Bax *et al.* 2001). Blooms in the area develop when EAC eddies provide upper-water-column stability at the same time as Ekman forcing uplifts nutrient-rich sub-Antarctic water (Bax *et al.* 2001).

Broad regional groupings of pigments denoting algal groups (Fig. 2c) are linked with oceanography (cf. Fig. 2a). However, in this area, where EAC-driven currents along the outer shelf and slope may be 30 cm s^{-1} or more, production reaching the sediment may have originated more than 500 km upstream (cf. Boon *et al.* 1998). This can lead to local enriched zones (Josefson and Conley 1997). The areas of fine sediments at the head of branches of the Bass Canyon – video observations show marine snow moving up over the lip of the shelf – suggest that oceanographic uplifting along the shelf break provides nutrients and other potential food from slope waters.

ZOOPLANKTON

The zooplankters in the study area were consistently divided into two communities (e.g. Fig. 2d). There was a highly diverse, species-rich, north-east and offshore community associated with warmer surface waters, higher nutrients and lower dissolved oxygen (especially at depth) and a relatively low-diversity, species-poor south-west and inshore group, occurring in areas with cooler surface waters, lower nutrients and higher dissolved oxygen (especially at depth). The northern extent of the inshore zooplankton community appeared to well match the discontinuities in surface temperature associated with the EAC eddy dominating the oceanography off New South Wales (Figs 2b and 2d).

INVERTEBRATES

Distinct differences in invertebrate communities were found in different habitats, although they had many taxa in common. Four clear trends in invertebrate communities were observed associated with changes in depth, latitude, sediment characteristics associated with hydrology (Figs 3a and 3b), and presence of hard substrata (not shown)(Bax and Williams 2000).

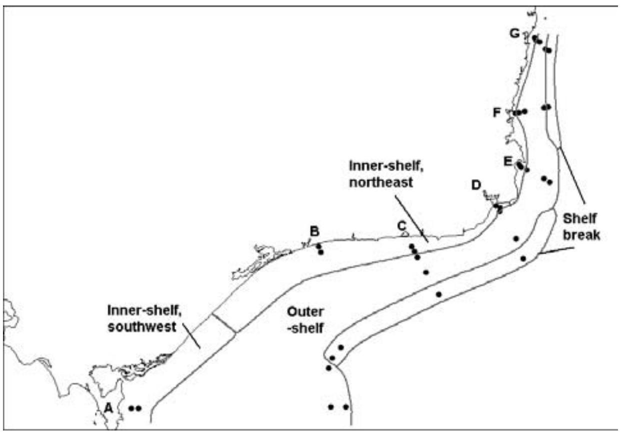

Fig. 3. Map showing invertebrate communities determined from (*top*) epifauna and (*bottom*) infauna collected with a benthic sled on soft-sediment substrata (sites given in Fig. 1).

The relationship between invertebrate fauna and habitat type is clearest for the offshore sites, where rough habitat is associated with a high coverage of sponges and bryozoans, whereas softer habitat is associated with bivalves and echinoids. A distinct invertebrate community characterized by stalked crinoids is found in areas of poorly sorted sediments of high biogenic activity occurring, for example, at the head of the Bass Strait canyon.

FISH

The four major correlates of spatial variation in demersal fish community structure for this bioregion are latitude, depth, seabed type and hydrography (Fig. 4)(Williams and Bax 2001), similar to the factors driving spatial organization in benthic invertebrate communities. Spatial variation associated with seabed type and hydrography is nested within latitude and depth. The interaction of prevailing currents with larger topographic features (e.g. deep upwelling at the head of arms of the Bass Strait Canyon) influences the production sources for demersal shelf fishes (Bax *et al.* 2001; Bulman *et al.* 2001).

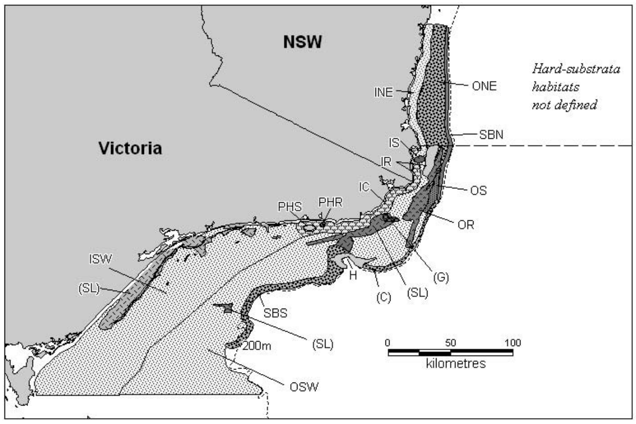

Fig. 4. Preliminary map of biophysical substructure on the south-eastern Australian continental shelf based on the conjunction of fish communities and distribution of substrata. Codes for sampled and mapped areas: INE – inner northeast; IC – inner central; ISW – inner southwest; ONE – outer northeast; OSW – outer southwest; SBN – shelfbreak north; SBS – shelfbreak south; IS – inner soft; IR – inner rough; PHS – Port Hicks soft; PHR – Port Hicks rough; OS – outer soft; OR – outer rough. Codes for areas mapped but unsampled for fish: (SL) – sandstone outcrop/limestone reefs; (G) – granite outcrop; (C) – bryozoan-consolidated sediments supporting colonies of stalked crinoids (from Williams and Bax 2001).

Demersal fish communities are generally species-rich (Williams and Bax 2001), comprising up to 80 species, reflecting both the high overall richness of the temperate Australian ichthyofauna (Paxton *et al.* 1989; Yearsley *et al.* 1994) and the location of the Twofold Shelf bioregion in a faunal transition zone, where elements of cool-temperate and warm-temperate faunas overlap (IMCRA 1998).

Implications for representative MPAs

We have shown that within the Twofold Shelf bioregion, latitude, hydrology (at several scales) and depth influence the overall distribution of sediments, biological communities and different size classes of individual species. The overall pattern of biological communities could be represented as three distinct depth-structured communities on the shelf, replicated over two latitudinal regions (separated by the distribution of the three water masses in the bioregion). Superimposed on these larger-scale patterns are the smaller-scale patterns created by significant seabed and hydrological features – reef complexes; scattered hardground outcrops; hardground mosaics of little vertical relief; soft-sediment areas of high biological activity associated with river outflows or deep upwelling of nutrient-rich slope water; high-current areas. Biological diversity is as much a function of these small- and variable-scale features as a function of the overall large-scale patterns. These small-scale features are also frequently the target of the commercial fisheries.

Distribution of organisms with habitat is a multiscale problem where the scale is dependent on the species and processes being considered (Garcia-Charton and Perez-Ruzafa 1999). The continental shelf of the Twofold Shelf bioregion is one of the few offshore areas around Australia where sufficient information has been collected to understand the scale-dependent structure that affects the distribution of biological communities. Given clear definition of the objectives and values behind the NRSMPA, it should prove possible to define particular areas, or a network of areas, that are(is) able to be representative of many of the different communities in this bioregion.

IMPORTANT PROCESSES IN THE TWOFOLD SHELF BIOREGION

Marine communities are not fixed in time and space – even sessile invertebrates frequently have a pelagic life-history stage that can lead to a wide dispersal of reproductive products and early life-history stages. Although representative MPAs can be defined through understanding the spatial structure of the bioregion, adequacy can be achieved only by understanding the processes that produce that spatial structure over the short and long term. In this section we discuss some of the fundamental processes in the Twofold Shelf bioregion that have helped determine the observed spatial structure in biological communities and then discuss more modern anthropogenic processes that will influence this structure in the future.

Life-history migrations

Complete life histories, including habitats occupied at each life-history stage, are unknown for almost all invertebrate and fish species in the Twofold Shelf bioregion. However, the majority of species with identified life histories have a planktonic stage where they may be widely dispersed.

Fish communities in the Twofold Shelf bioregion are structured by depth-related processes, with defined ranges across the shelf being seen in most individual fish species. All major commercial fish species show an increasing length with depth for sediment-flats habitats; however, this increase can be masked by large variation in length between habitats for some species (Bax and Williams 2000).

For sessile invertebrates, or other species that show little long-range movement once settled out from the water column, seabed habitats occupied may represent sources or sinks for subsequent populations. It seems likely that in the Twofold Shelf bioregion, which is strongly structured by seasonal alongshore currents, there are areas of the seabed that, despite containing healthy populations of sessile invertebrates, may not be able to sustain future generations of their species – they are dependent on 'upstream' populations for future generations. A good example of the directional nature of invertebrate recruitment is the northwards spread of the New Zealand screwshell (*Maoricolpus roseus*) from Tasmania, where it is thought to have been introduced in the 1920s, to Sydney where it was found in 1999, with no comparable westwards expansion.

Seasonal migrations

The habitat experienced by organisms in the Twofold Shelf bioregion is the intersection between a stable topographic substratum and the mobile water masses above. Whereas sessile invertebrates and the majority of fish species that depend on the benthic food web may remain associated with particular benthic habitats, mobile fish species, especially the majority of commercial species that depend on the midwater food web (Bulman *et al.* 2001), may remain associated with water-column habitats. As water-column habitats change with the seasonal progression of water masses in the area, it is to be expected that many fish species will undertake seasonal migrations.

Commercial catches in the Twofold Shelf bioregion show a seasonal pattern as fish move between depths and along the shelf. Some species are only available for limited periods of the year during spawning migrations (e.g. gemfish, orange roughy); other species are available year-round but in different parts of the shelf (Klaer and Tilzey 1994). Experienced local fishers report distinct seasonal movements of some abundant species including the economically important redfish, jackass morwong and blue warehou over the deep shelf (Williams and Bax 2001). These movements are also related to depth, substratum type and time of day, and may vary inter-annually.

Fishing effort

Mobile fishing gear has measurable short-term impacts on the structural components of habitat (reviewed by Auster and Langton 1999) – removing or damaging epifauna, smoothing sedimentary bedforms, and removing taxa that create structure. The limited available data suggest that dragging fish traps, longlines and gillnets across the seabed has similar short-term impacts, although restricted to a much smaller area (Auster and Langton 1999).

Fishing effort in the Twofold Shelf bioregion is increasing – the time that trawls are on the bottom has increased steadily since comprehensive monitoring of effort began in the early 1980s, additional licences will expand fishing by autolongliners, and the adoption of advanced navigational aids (track plotters and GPS) and gears that fish rough ground effectively has enabled effort in all sectors to be increasingly targeted at the 'hard-ground' habitat features that attract fish (Bax and Williams 2000). Effort in the fishery continues to rise, despite recent management attempts to reduce it (Fig. 5; Larcombe *et al.* 2001).

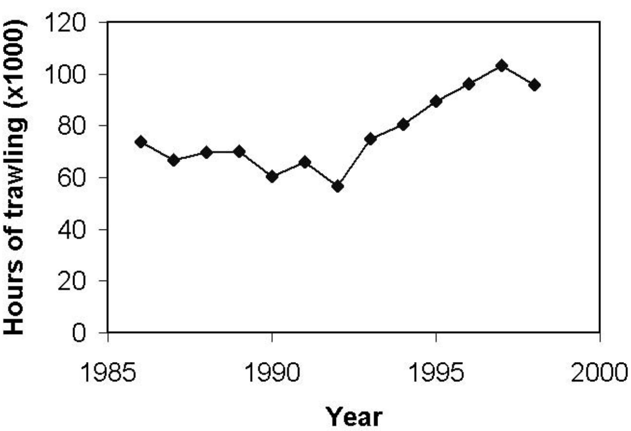

Fig. 5. Total effort in hours (1986–98) for the South East Fishery trawl sector. Retained data only (From Larcombe *et al.* 2001).

Introduced marine pests

Introduced marine pests threaten Australia's coastline, and a few of these marine pests threaten the continental shelf. Recent port surveys have identified more than 250 introduced marine species in Australian waters and, on the basis of their arrival over the past century and a half, the rate of arrivals is increasing exponentially. A detailed study of marine introductions of Port Phillip Bay in Victoria, adjacent to the Twofold Shelf bioregion, identified more than 150 introduced species (Hewitt *et al.* 1999) – the three most abundant species in the bay are introduced species. On the basis of historical trends, a new species arrives every 41 weeks. Two introduced marine species are of particular concern to the biodiversity of the Twofold Shelf bioregion – the New Zealand screwshell *M. roseus* and the North Pacific seastar *Asterias amurensis* – because they are already here and have the capacity to invade much of the continental shelf in this bioregion.

M. roseus inhabits depths from the shoreline to at least 80 m in the Twofold Shelf bioregion (Bax and Williams 2000). In its native New Zealand it occurs down to 130 m and reaches densities in excess of 1000 individuals per sq m. It is the only known introduced marine species, anywhere in the world, that has successfully invaded the continental shelf from a port environment. Very little is known about the biology of *M. roseus*, its impacts on sediment structure or its competition with other invertebrates. Even the empty shells may have substantial impact as homes for hermit crabs, as indicated by the crabs' frequent occurrence in areas where *M. roseus* is abundant. From its densities, it is likely that *M. roseus* may well be the environmentally most damaging of the introduced marine species present in Australia, though largely out of sight and hence unknown to the general public or conservation managers.

The North Pacific seastar, *A. amurensis*, arrived in the Derwent estuary in the 1980s but it was not recognized until the 1990s when the population was estimated to number in the tens of thousands. Nothing was done to reduce the risk of the seastar spreading outside the Derwent, and in 1996 the first few specimens were collected from Port Phillip Bay, Victoria, presumably transported there by a commercial or recreational vessel. Numbers increased from a few occasionally collected specimens in 1996 to over 115 million individuals in 2001 (Fig. 6); it now covers 1500 km^2 and its biomass equals the total biomass of fished species in the bay. Prevailing currents can now spread it northwards along Australia's east coast at least as far as Bermagui. *A. amurensis* is a dominant invertebrate predator that occupies habitats from the subtidal to 200 m depth in its native habitat. In its presence, the abundance of shellfish is greatly reduced.

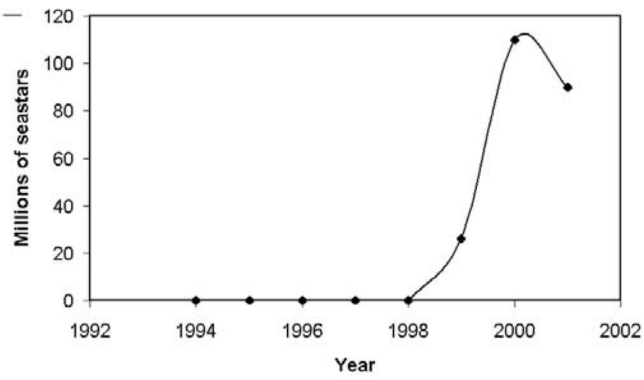

Fig. 6. Numbers of *Asterias amurensis* estimated to be in Port Phillip Bay (Victoria, Australia). Early data are numbers caught by scallop dredgers (Don Hough, Victorian DNRE, *pers.comm.*)

Implications for adequate MPAs

Marine protected areas can be delineated by fixed lines drawn on a two-dimensional representation of a four-dimensional habitat. One of the missing dimensions – time – is critical to the adequacy of a MPA for achieving management objectives. The other – the water column intersecting with the substratum and extending above it to the ocean surface – provides the medium for time-varying processes to operate. For an MPA to be adequate it must address the four dimensions of habitat.

Over time, nearly all marine invertebrates and vertebrates will cross the lines drawn on a map – through seasonal migrations, changes in habitat with maturity, through release of gametes into the overlying water masses. Once outside the MPA, organisms will be susceptible to deleterious events, such as fishing or loss of habitat. All an MPA can protect, by itself, is the physical habitat and the limited number of self-recruiting populations within it. Even this protection may be limited over the long term, as fishing effort continues to increase and introduced marine pests continue to spread through Australian waters – lines drawn on a map will provide no deterrent.

For a MPA to be adequate in protecting marine life, it must be part of a larger process of an integrated management strategy that controls external events including introduced marine pests, fishing effort, marine pollution and climate change.

CONCLUSION

Fundamentally different processes shape structure in landscapes and 'waterscapes', especially 'seascapes' (Larkin 1978; Holling 1992; Link 2000). Higher connectivity, life-history-mediated changes in habitat, seasonal migrations, and habitat occurring at the interface of two physical processes at different scales (the seabed and the impinging water masses), suggest a greater openness of marine systems than their terrestrial and freshwater counterparts, with implications for the adequacy of natural reserves and other spatial management options. It is already being recognized on land that 'off-reserve conservation' is critically important, and it may be even more important in the marine environment to look outside the area being protected to ensure that larger-scale processes are being managed adequately; time and geographic scale determine the questions, or management objectives, that can be addressed (Larkin and Gazey 1982).

A structured hierarchy of models, representing processes occurring at several levels of scale, can be used to restrict complexity at each scale to achievable levels (May 1973; O'Neill *et al.* 1986). A scale higher than the process of interest provides a context and top-down constraints on the focal level, while the lower level provides mechanisms and imposes bottom-up constraints (Wu and Loucks 1995). Hierarchical management programs, containing objectives, performance measures, indicators, reference points and decision rules for each level, may be necessary to adequately manage the open marine environment, where processes occur at many different scales.

Without adequate attention to management of broader-scale processes, small (as a fraction of the bioregion occupied by key species) marine protected areas surrounded by a hostile landscape that removes or adds new species will lose their distinctive species. Conservation of biodiversity needs to be addressed at all levels in the bioregion. Overemphasis on MPAs should not be used as a substitute for effective management of fisheries, invasive species and other human-related impacts at the broader scale.

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INVOLVING FISHERS' DATA IN IDENTIFYING, SELECTING AND DESIGNING MPAS: AN ILLUSTRATION FROM AUSTRALIA'S SOUTH-EAST REGION

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Abstract

Commercial fishers are the most frequent observers and users of many marine environments – especially in offshore regions. They have mapped, named and sampled many features of the continental shelf and slope that the rest of the community is unaware of, and collectively have knowledge for large areas. Scientists use relatively sophisticated equipment and methods for mapping, but usually over relatively small areas because they do not have the resources to do more. Scientific observation using hydroacoustics, cameras and physical samplers can provide fine-scale spatial resolution of the environment, while larger-scale information is provided by archival data on biota and geology together with remote-sensing. Fishers' information, based on repeated observations over long periods, charts the environment at an *intermediate* level of resolution. At this intermediate level, the types and boundaries of habitats are defined at scales of tens of kilometres – 'megahabitat-scales' – over large areas of the seabed, and are complementary to scientific data. CSIRO and sectors of the offshore fishing industry are working together to map the seafloor of south-east Australia. Prospectively, information gathered in this way will contribute to understanding representativeness and adequacy at the broad-area identification phase, and will also contribute to MPA selection and design phases during the present development of a CAR system of MPAs in the South-East Region of Australia. The data also provide a first means of identifying both the fishery implications of MPAs, and the links between conservation and fishery management goals that are unclearly specified for this region at present. As importantly, the process of involving the fishing industry at all stages of the map development provides them with the information they require if they are to insist that proposed spatial management of their working environment is appropriate and based on sound environmental principles.

Keywords: South-east Region, CAR, MPA, fishers, seabed mapping

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INTRODUCTION

Australia is developing integrated management of its marine resources through Australia's Oceans Policy, launched in December 1998. Principal drivers for the policy are: ecosystem-based management; integrated oceans planning and management for multiple use; promotion of ecologically sustainable marine-based industries; and management for uncertainty (Commonwealth of Australia 1998). Implementation of these planning concepts will be through Regional Marine Plans (RMPs) developed by the National Oceans Office (NOO 2002a). The development of marine protected area (MPA) proposals under the Commonwealth (federal) component of the National Representative System of Marine Protected Areas (NRSMPA) will be developed as part of the regional marine planning process.

The marine environment off south-east Australia, the South-east Region (SER), is the test case for regional marine planning in Australia – it forms the first of 13 'large marine domains' that will eventually be covered by management plans. Details of the draft operational criteria and process for identifying and selecting a representative, comprehensive and adequate system of MPAs in the SER were released in July 2002.

Commercial fishers are key stakeholders in the SER. Offshore, the largest geographical overlap is with the South East Fishery (SEF) – a complex, multi-species, multi-sector fishery that operates on the continental shelf and slope adjacent to mainland Australia, and on some offshore seamounts and rises (Tilzey and Rowling 2001). It is Australia's largest scalefish fishery, and the most important source of scalefish for domestic markets.

There are many reasons for involving fishers in the process of identifying, selecting and designing MPAs (Baelde *et al*. 2001). Most obvious is that MPAs are likely to affect fishers' access to fishing grounds, and, as a single stakeholder group, commercial fishers are often most affected by MPAs (Hall 1999). However, commercial fishers are also usually the most informed about the broad structure and condition of the marine environment – they are out there fishing most days of the year – and have the potential to substantially improve the process of MPA selection. The aim of this paper is to examine this second aspect of fishers' involvement in MPA development: the relevance, and prospective contribution, of their data and knowledge to the CAR process of identifying and selecting candidate areas. We start by providing an overview of the process, the data needs and data availability for MPA development in the SER, and illustrate this with reference to one area – the Twofold Shelf bioregion (see also the companion paper, Bax and Williams, these Proceedings). We then show at what level fishers' data can enhance this process, and finish by detailing how we are working with fishers to collate their data and provide them with the capacity to actively participate in MPA design in Australia's 'South east Region'.

MPA DEVELOPMENT IN THE SER: ECOSYSTEM UNITS

Protection of biodiversity in the Australian marine environment will be implemented partly through a comprehensive, adequate and representative national system of MPAs – a systematic 'CAR' approach. In simple terms this means reserving areas that reflect the biodiversity of particular marine ecosystems (representative), of sufficient size and spatial distribution to ensure their ecological viability (adequate), for the full range of ecosystems (comprehensive). MPA development in the SER has two interactive phases: identifying broad candidate areas based on regionalisations of biological and physical data to differentiate major ecosystems, and selection of reserve sites from, or within, candidate areas based on human and scientific considerations.

The many guidelines and actions needed to implement the NRSMPA (ANZECC TFMPA 1999) rely on a variety of spatial data to define ecosystems. However, for the offshore seascape, data that are both detailed and wide-ranging are rarely available. Unlike spatial management on the land – which has benefited from numerous datasets available from visual observation of the landscape – similar information is not available for the seascape because it cannot be observed directly (except at the shallowest depths).

Implementation decisions for MPAs will be have to be made using the best information available, which in many instances will be limited in time and space and sometimes based on surrogates (ANZECC TFMPA 1999). The available information for identifying bioregions and smaller-scale spatial units is usually a combination of broad-scale datasets, such as bathymetry and physical oceanography gathered for the entire region, and archival (museum) data such as taxonomic and geological inventories assembled over decades, together with fine-scale data on habitat types and their associated species for a selection of (usually) isolated locations.

Intermediate-scale data that provide habitat and biological community distributions at scales of tens of kilometres within bioregions are not available for most areas of the Australian shelf or slope. Techniques for using surrogate variables to reliably predict the distributions of habitats and components of biodiversity at intermediate scales are under active development (Kloser *et al.* 2001b). These methods are typically based on single-beam or multi-beam acoustics in conjunction with cameras and physical samplers (Kloser *et al.* 2001a, 2001b, respectively) and are providing increasingly detailed and accurate 'pictures' of seabed habitats and biodiversity. However, substantial resources are required before scientific mapping at intermediate scales can be extrapolated over large areas.

Bioregionalisations have been completed for the SER continental shelf (<200 m depth) (IMCRA 1998) and deeper regions (NOO 2002b). However, intermediate-scale habitat distributions have been mapped for only one area – the Twofold Shelf bioregion (Bax and Williams 2001; Williams and Bax 2001). In this area, at least six distinct biological communities were identified on the shelf alone.

MPA DEVELOPMENT IN THE SER: SPATIAL FRAMEWORK AND HABITAT CLASSIFICATION

The multi-scale structures and functions of (marine) ecosystems (e.g. Langton *et al.* 1995; Garcia-Charton and Perez-Ruzafa 1999; Roff and Taylor 2000) necessitate that a classification scheme for habitats and a spatial framework of habitat or biological community distributions be developed, before spatial management of resource use can be implemented. This is exemplified by the process of bioregionalisation that underpins the development of a network of MPAs, where large areas are sequentially subdivided into units that represent either identifiable ecosystems or areas that are amenable to management (both usually at large spatial scales).

A hierarchal classification of "habitats" is effectively used as a surrogate for the hierarchy of ecological units and processes that are the subject of MPA development. The scheme applied to the SER recognises a series of nested, pseudo-spatial 'Levels' for the structure of habitats, each reflecting the influence of characteristics and processes acting at different scales (Table 1). It is under development (mainly by V. Lyne and P. Last of CSIRO Marine Research) but is presented with illustrations, and examples from the Twofold Shelf bioregion, in Kloser *et al.* (2001b). The bioregionalisation for the offshore regions of the SER (>200 m) differentiates bioregions at Level 3, i.e. as a set of biogeomorphological units (Table 1) (NOO 2002b).

Because different natural systems are not delineated at spatial scales that are either clearly defined or repeated, the boundaries between levels are rarely sharp or unequivocal (Allen and Starr 1982). Hence, the scheme of Lyne and Last (Table 1) is pseudo-spatial: ecosystems defined at one level may not all be at the same spatial scale, while ecosystems at one level may not be 'smaller' than others at the next higher level. Nevertheless, in most systems there are discontinuities that can be recognised, and these have allowed the development of a number of classification schemes for different purposes. A useful example for classifying deep seabed habitat is Greene *et al.* (1999). These authors (e.g. Greene, pers. comm.) are not wedded to the fine details; what they stress is the importance of the hierarchical view, and the need for an agreed classification scheme as a working language for their particular purposes.

An illustration for the SER is provided by (Bax and Williams, these Proceedings) from a study of the continental shelf portion (25–200 m depth) of the Twofold Shelf Bioregion (IMCRA 1998). Several ecosystem features in that region including the distribution of sediments, biological communities, and size classes of abundant fishes, were influenced primarily by latitude, hydrology and depth at 'provincial' scales (*sensu* Greene *et al.* 1999) of hundreds of kilometres. At a finer scale, biological patterns were due to substratum type, geomorphology and locally modified hydrology at 'megahabitat' scales of one kilometre to tens of kilometres, or less (Williams and Bax 2001). Ideally, then, the development of MPAs in a marine system would have management objectives, performance measures, indicators, reference points and decision rules that take all spatial scales into account, even if the MPAs will only operate at one particular scale in the hierarchy.

WHAT ARE FISHERS' DATA AND HOW ARE THEY RELEVANT TO MPA DEVELOPMENT?

Habitat distributions at megahabitat-scale are not known for the vast majority of the continental shelf and slope seabed around south-eastern Australia. Information at this scale is available for the Twofold Shelf region because habitat distributions were mapped and sampled in several surveys over five years (Bax and Williams 2001). A vital component of that mapping process was to integrate fishers' spatial information on habitat distribution with survey data (Williams and Bax 2003).

Habitat mapping in this way was an iterative process over the life of the study. At the project's commencement, navigation around the bioregion was based on third-party, coarse-scale bathymetry data and navigation charts – primarily point-source depth soundings, the approximate positions of key depth contours including the continental shelf edge at ~200 m, and the positions of some near-surface rocky banks identified as shipping hazards. This information was used in combination with limited existing survey data, and some rapid exploration by echo-sounding during surveys, to fix a set of transects and sampling sites, stratified by depth and latitude, for trawl surveys (Bax and Williams 2001, Fig. 1).

These sites provided broad-scale information across the Twofold Shelf, but only for soft-sediment substrata. It was dialogue with knowledgeable local fishers that enabled targeted sampling of consolidated substrata, mostly rocky reefs, to be progressively built into the field surveys. What evolved at the end of the study was an intermediate-scale map of habitats – the 'fisher map' – a hybrid mix of fisher-delineated geomorphological features at scales of tens to hundreds of kilometres (such as sediment plains and rocky banks), ground-truthed with physical samples and photographs from surveys that identified biological facies – patches of substratum and their dominant faunal elements or characteristic community types at scales of metres. The map is reproduced at coarse resolution in Fig. 1, with a zoom view and detail for selected areas in Fig. 2.

Geomorphological features at scales of tens to hundreds of kilometres, such as sediment plains and rocky banks, (Level 3), and details of their primary substrata and biota (Level 4) were mostly defined by fishers (Fig. 1). Ground-truth physical samples and photographs from survey identified secondary biotopes (Level 5), and biological facies (Level 6) (Fig. 2).

A CLOSER LOOK AT THE NATURE OF FISHERS' DATA

Fishers' data – maps and names

Fishers have names for large numbers of a great variety of seabed features at a range of spatial scales, including small scales (tens of metres to a few kilometres). These enable navigation around the spatial framework: visualizing and interpreting patterns in data at a variety of spatial

Table 1. Overview of the hierarchical scheme used to classify the structure of marine habitats in the South-east Region (under development by V. Lyne and P. Last, CSIRO Marine Research, version 1.2, February, 2001). For more detail see NOO 2002b)

Level	Brief description
1 - Provincial	Biogeographic units.
2 – Biomes/ sub-biomes	Large areas with characteristic collections of species: the biotic communities of the coastal region, shelf, slope and abyss differentiated by depth and latitude.
3 – Biogeo-morphological units	Easily identifiable geomorphological subdivisions, usually with distinct biotas. Typical units on the continental shelf include sediment plains, rocky banks, and valleys and cliffs at the shelf-break, while continental slope units include canyons and seamounts.
4 – Primary biotopes	Biotic assemblages associated with broadly different substrata (soft, hard or mixtures) and modified by hydrological variables such as wave exposure, turbidity, tidal effects and current speed.
5 – Secondary biotopes	Generalised types of biological and physical substrata within the soft/hard/mixed types (e.g. igneous, calcareous, silts, sands, gravels, seagrasses, sponges) together with geological, biological and ecological interpretation (community structure and composition or biodiversity) provided by biological and physical sampling.
6 – Biological facies	Identifiable biological and physical units defined by a biological indicator, or suite of indicator species, used as surrogate for a biocoenosis or community. They include, for example, a particular species of seagrass, or group of corals, sponges, or other macro-fauna that generally occur together.
7 – Micro-communities	Assemblages of species that depend on member species of the Facies (e.g. communities associated with kelp holdfasts).

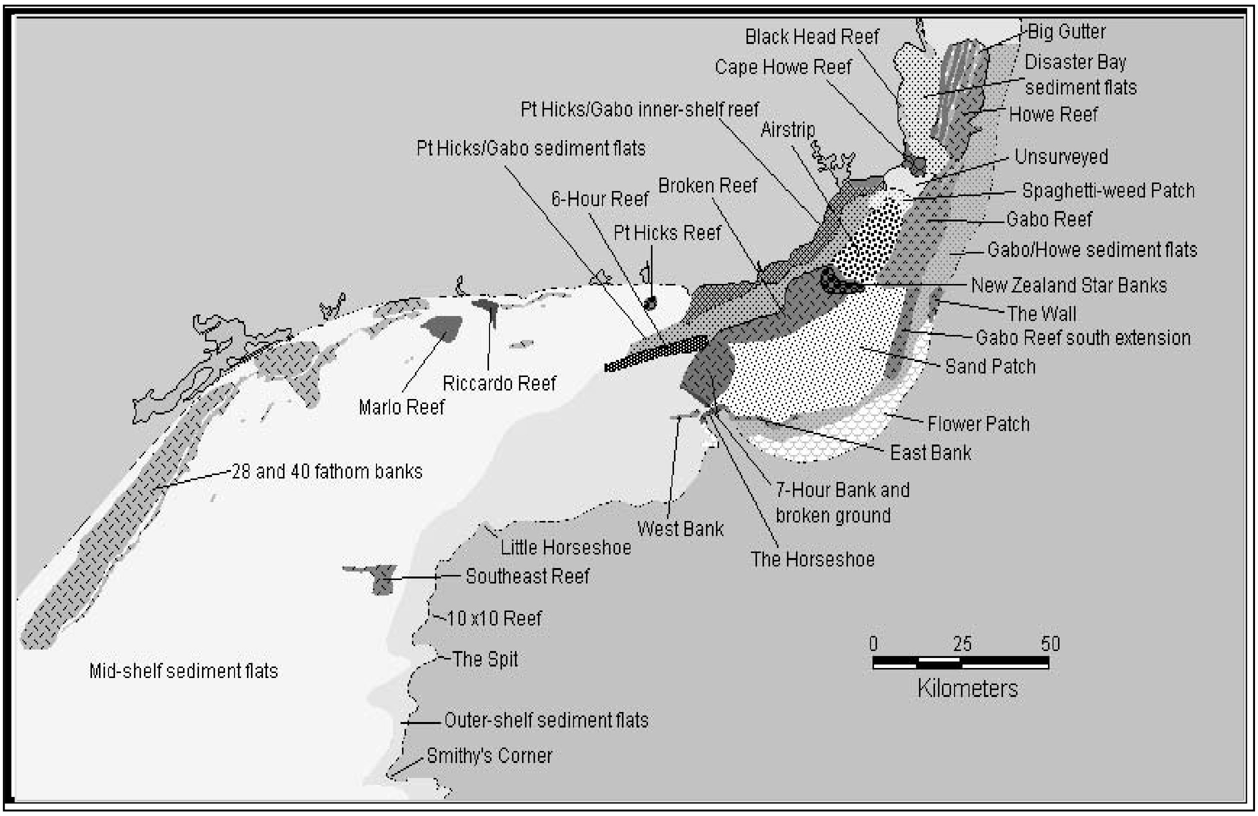

Fig. 1. A coarse-scale map of habitats – the 'fisher map' – made for the Twofold Shelf Bioregion (from Bax and Williams, 2001, Fig. 4). The map combines is a mix of fisher-delineated geomorphological features (mostly sediment plains and rocky banks) ground-truthed with physical samples and photographs from surveys (Fig. 2).

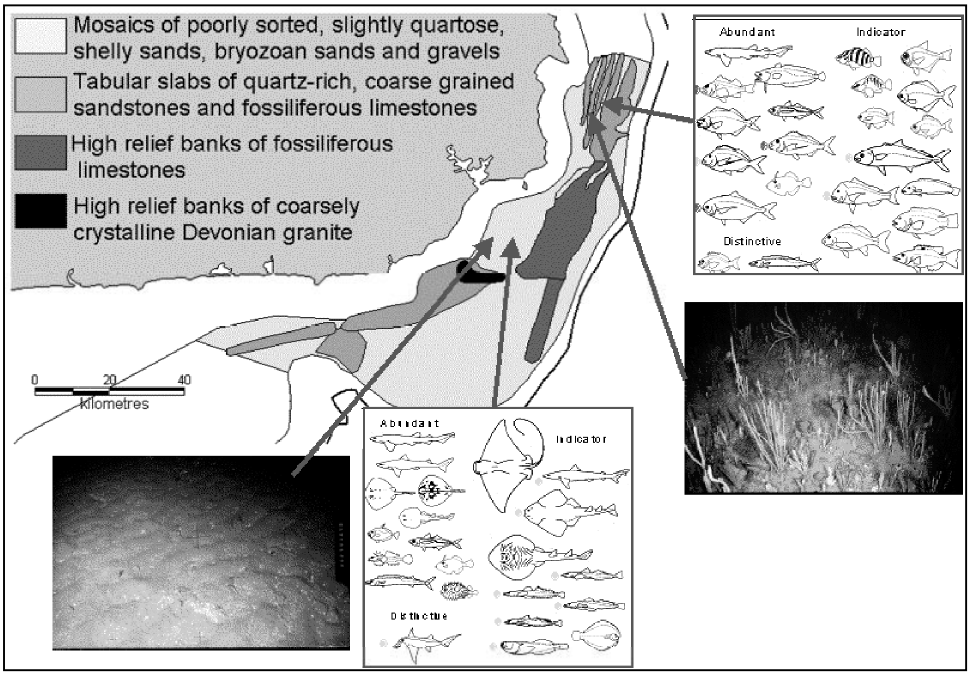

Fig. 2. Zoom view of a section of the outer continental shelf within the Twofold Shelf Bioregion with ground-truth detail for the "Airstrip" and inner "Gabo-Howe Reef complex" 'megahabitats' (see Fig. 1). Data on fish communities and habitats from Williams and Bax (2001) and Bax and Williams, 2001 respectively.

scales and providing a common language for discussing system properties. In contrast, scientists are usually restricted to navigating by a limited range of names from navigational charts – mostly coastal features such as headlands or towns, near-surface rocky reefs identified as shipping hazards, and major features of seabed topography such as the shelf edge and offshore platforms – these may have little to do with the spatial units that describe biological communities. Occasionally, the better-known names given by fishers to features visible only on echosounders are also included in scientists' vocabulary. One example for the SER is the naming of seamounts in a survey (Koslow *et al.* 2001) that led to the establishment of the Tasmanian Seamounts Reserve. The 'fisher map' exemplifies this for the Twofold Shelf bioregion: 33 names for major seabed features (megahabitats) can be attributed to fishers while only three (two near-shore reefs and one shipping hazard) are found on navigation charts of the area). At a finer scale, fishers also give names to individual habitat patches or geomorphological features such as rocky reefs and the 'gutters' between them.

Fishers' data – physical sampling

Fundamental differences between observations made by fishers during commercial fishing and by scientists during survey are related to the timing, frequency and coverage of sampling (Williams and Bax 2003). In offshore regions, fishers sample frequently, often targeting particular topographical features, current regimes, or periods of day and night. As a result they gain good, although unquantified, knowledge of local ground types and their species-mixes or 'taxonomies' of fishes and benthic invertebrates. In contrast, scientists tend to gain highly detailed data from a variety of specialized sampling tools, but usually from relatively few samples that are often untargeted. Because fishers and scientists tend to observe marine ecosystems at different spatial and temporal scales, their observations have the potential to be complementary. Unfortunately, and as is usually the case, in the absence of adequate communication and cross validation between scientists and fishers, these different observation scales lead to different system views and the potential for divisive debates.

Fishers' knowledge may permit scientific observing to be better targeted and more insightful, while survey data can provide detail that leads to more rigorous interpretation of fishers' knowledge. The comprehensive scope of fishers' exploration and fishing provides the means to extrapolate the point sampling of scientists to larger scales, and to locate unique areas of biodiversity that may remain undetected by survey or surrogate-based approaches. In the context of MPA development, this means that fishers collectively will frequently have knowledge about biodiversity and spatial structuring of which the broader community, including scientists, is unaware. In this respect, their knowledge is relevant to both systematic (CAR) and to targeted (iconic area) approaches to MPAs in the SER.

CONTRIBUTING INDUSTRY DATA TO MPA DEVELOPMENT IN THE SER

The need for MPA declaration in the SER, as part of the Regional Marine Plan which is to be completed by 2003, means that lines must be drawn on the water that will, firstly, identify broad areas of interest from which, secondly, draft candidate MPAs are selected before, thirdly, MPA sites are chosen. The utility of 'fisher map' style mapping data, if collected in the right form and to meet the above timetable, becomes obvious. The data set would prospectively provide interpreted habitat information (distribution, boundaries, sizes, generalized geology and community types) at 'megahabitat' scale or finer, with near-complete coverage for the continental shelf and slope (from about 100 m out to about 1300 m depth), over all SER provinces.

The data are relevant in two ways. Firstly, and with regard to the conservation goals of the NRSMPA, megahabitat-scale data with provincial-scale coverage are a unique contribution to understanding representativeness and adequacy under the CAR approach (Bax and Williams, these Proceedings). Their inclusion is therefore prospectively beneficial (for managers *and* industry) to the broad identification phase by defining the essential fishing grounds, and may be the best available for the selection and design phases by providing megahabitat data – especially the areas, shapes and boundaries of habitats. Secondly, with regard to fishery management goals in the SEF, the data provide a first means of identifying the fishery implications of any area management (such as effort displacement by area closures) and the scope for integration of spatial planning by conservation and fishery managers. Presently, the linkage between spatial management planning for biodiversity conservation and for fishery purposes is not clearly specified (Baelde *et al*. 2002), benefits to fisheries from MPAs are not well established (Ward *et al.* 2001), and prospects for integration of conservation and fishery management goals in the SER remain largely unexplored.

A joint project between CSIRO and the trawl and non-trawl sectors of the offshore fishing industry (detailed below) was started in 2001 with the explicit aim of incorporating fishers' knowledge of the seascape into strategic management planning. Industry executives supported the project primarily because they viewed it as a way to participate directly in the forthcoming, but then unspecified, spatial management process. It was argued that, with their information systematically collected and rigorously evaluated, fishers would be positioned to critically evaluate proposed spatial management plans, such as the placement of MPAs, and require management agencies to have clearly defined and measurable aims for their proposed management options. In this way, fishers could reduce the likelihood of inappropriate MPAs holding little conservation advantage and only a cost to industry. However, support at executive and grass-root levels was not unanimous, and remains that way, in large part because many fishers fear that their information will be used against them, especially for closing off valuable fishery areas.

Nonetheless, at the time of writing, a large volume of data (some 550 separate electronic files) had been contributed and processed, and maps made at various levels of refinement for most of the shelf and slope in the SER. There is momentum to introduce these data in time to contribute to both the initial identification and subsequent selection of MPA sites. However, although involvement of industry data in this way has clear prospects for enhanced conservation outcomes, fishers remain uncertain about the outcomes for them and therefore uncertain about how, or indeed whether, to contribute their data. The consultative process will need to clarify key issues that remain unclear at this stage of the planning process: the likely negative impacts of MPAs on commercial fisheries, particularly those stemming from effort displacement; the links of systematic MPA development defined by conservation goals to spatial management actions defined for fishery goals; and the tangible benefits that will come from sharing their knowledge.

OVERVIEW OF CSIRO–INDUSTRY MAPPING PROJECT

A list of the main project features and structures to address the issues of involving fishers' data in the spatial planning process is shown in Table 2. Importantly, a high degree of transparency gives fishers a high degree of control over the form (spatial scale, information content, overlays of other data sets) and timing of any outputs, and authority is required from individual contributors and the relevant associations for release of information. This is anticipated to be a step-wise and adaptive process because it will be necessary to determine, firstly, what industry is confident to release, and secondly, what specific products are needed for an MPA development process that is evolving rapidly.

Table 2. Overview of CSIRO–industry mapping project in the Australian South-east Region: issues and project structures

Issue	Project structure
Data collection	Collection in ports and at sea by project leader known to fishers.
Spatial data and maps	Mainly based on electronic data from fishing vessel track-plotters.
Habitat attribute data	Terrain and bottom types, species mix and fishing patterns collected using a questionnaire developed with industry input together with fishery logbook data.
Verification and validation	Procedures in place to ensure data are scientifically rigorous.
Data management	Storage and map production with a customised spatial database for spatial and attribute data.
Formal arrangements	Responsibilities for CSIRO and industry set out in a memorandum of understanding, and data security and IP agreements.
Field sampling	From industry vessels with a high-tech camera system designed and built as part of the project.
Other data	Scientific survey and other data (geology/ oceanography/ video/ logbook/ socioeconomic) for GIS overlays.
Industry consultation	Continued involvement of industry through peak associations, Steering Committee and individual operators.
Agency consultation	Steering Committee with multi-agency and cross-sector industry representation.
Release of industry maps/ information	Step-wise and adaptive with clear arrangements for industry review and approval procedures.

CONCLUSIONS

Fishers' mapping data, if collected in the right form and to meet the MPA development timetable for the SER, could provide interpreted habitat information (distribution, boundaries, sizes, generalized geology and community types) at 'megahabitat' scale or finer, with near-complete coverage for the continental shelf and slope (from about 100 m out to about 1300 m depth), over all SER provinces. This is relevant to all phases (identification, selection and design) of MPA development, as well as other forms of spatial management for fishery goals. Spatial management – including MPA declaration – based on coarser levels (bioregion and depth) increases the risk of unnecessarily restricting fishing activity, while not increasing conservation benefits.

Including fishers' knowledge in defining spatial management of a seascape best known to them is perhaps the best way to gain their acceptance and understanding of conservation objectives. Achieving this understanding is likely to provide benefits in the subsequent operational stages of spatial management, e.g. compliance, surveillance, performance assessment and monitoring (Baelde *et al*. 2002). Active and successful participation of fishers in this process for the SER could provide a blueprint for industry participation in future phases of the NRSMPA.

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WHAT FEATURES MATTER WHEN DESIGNING PROTECTED AREAS FOR FISH IN BEDS OF SEAGRASS: A REVIEW

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Abstract

There has been considerable research in recent years on the factors determining fish abundance and diversity in beds of seagrass. This research needs to be communicated to managers and designers of marine estuarine protected areas (MEPAs) so that the information is used. A review of the literature finds some general features of a seagrass bed that can be favourable for the abundance and diversity of fish. Small beds of seagrass can have greater species richness than larger beds and must be included within a network of protected areas. This network must also include both patchy and uniform beds because they were found to have different assemblages of fish. Similarly, beds of varying distance from the estuary mouth were found to have different assemblages of fish. So beds from all regions of the estuary must be included to ensure the protection of all fish species. The protection of adjoining mangrove habitat as well as seagrass habitat is recommended, because beds proximal to mangrove had significantly greater species richness than beds without adjoining mangrove. The effects of the following on fish abundance and diversity were not well covered by the literature: the interconnectedness of seagrass habitat, the shape of seagrass beds and seagrass heterogeneity. The existence of edge effects for fish in beds of seagrass is yet to be determined, and an understanding of the interactions of adjoining habitat with beds of seagrass is required. The eventual aim should be an understanding of the seascape processes that determine the diversity of fish within estuaries.

Keywords:: seagrass, fish, SLOSS, connectivity, edge

INTRODUCTION

Coastal regions are among the most rapidly urbanizing places on earth (Crooks and Turner 1999; Ehrenfeld 2000) and this has resulted in unprecedented habitat loss. The establishment of marine and estuarine protected areas (MEPAs) in Australia is generally regarded as a means to ameliorate this destruction of natural habitat. The estimated loss of seagrass habitat in estuarine communities is thought to be significant although few studies have observed and empirically recorded this loss (Kirkman 1978; Larkum and West 1990). This is of concern because seagrass beds are important habitats for the fauna in estuaries, especially for juvenile fish and invertebrates (Heck and Thoman 1984; Middleton *et al.* 1984; Orth *et al.* 1984).

A landscape or seascape approach is considered the most appropriate model when designing MEPAs (Agardy 1994; Robbins and Bell 1994). The ocean and surrounding environs are considered to be open systems with a great degree of connectedness due to the action of waves, currents and tides. Yet many MEPAs in Australia are multi-use areas (including the largest, the Great Barrier Reef Marine Park) that allow for commercial and recreational use in designated zones of protected areas. Designers of a protected area rarely have the opportunity to conserve the entire estuary and its catchment; they are required to make informed decisions about which regions should be protected or used.

At present there is little consensus on criteria such as the size, shape and connectedness of protected areas (McNeill 1994). Nevertheless, there is evidence that some general features of a seagrass bed can favour abundance and diversity of fauna. Within an estuary there is usually a region of seagrass that contains greater diversity and abundance of fish and invertebrates than the rest of the estuary (McNeill *et al.* 1992). This paper reviews the most common features of a seagrass bed identified by researchers to contribute to fish abundance and diversity; the aim is to identify consistent patterns that could be used by managers and designers of protected areas for fish in beds of seagrass. This review will be limited to more large-scale features of the bed such as size, shape and patchiness. The effects of small-scale seagrass structure such as shoot density, leaf length and biomass will not be considered.

QUESTIONS TO BE CONSIDERED

Designers of a MEPA need to know what influence on fish abundance and diversity the following features of a seagrass bed may have: size, shape, edge effects, spatial structure (i.e. patchy or uniform), distance from the estuary mouth, and proximity of adjacent habitat (e.g. mangrove).

INFLUENCE OF SIZE AND SHAPE OF SEAGRASS BED

It is unclear how the size and shape of an aquatic habitat affects species diversity and abundance. Ever since Macarthur and Wilson presented their Island Biogeography theory in 1967, there has been considerable debate on how large and how many habitats are required to conserve species diversity. Macarthur and Wilson (1967) hypothesised that larger habitats or islands had greater diversity (species richness) than smaller habitats or islands. Conversely, other ecologists (Diamond and May 1976; Connor and McCoy 1979, Simberloff and Abele 1976) hypothesised that many smaller habitats would contain greater species richness than a single habitat of the same total area (This debate has been termed SLOSS, single large or several small habitats.

The response of fauna to the size of a bed of seagrass has been investigated by only a small number of researchers (McNeill and Fairweather 1993; Irlandi *et al.* 1995, 1999; Irlandi 1996, 1997; Eggleston *et al.* 1998, 1999; Bologna and Heck 1999; Frost *et al.* 1999; Bell *et al.* 2001; Hovel *et al.* 2001, 2002). Most, but not all, of these studies investigated the response of invertebrates. Only three studies (McNeill and Fairweather 1993; Eggleston *et al.* 1999; Bell *et al.* 2001) considered the effects of bed size on fish.

McNeill and Fairweather (1993) found that a combination of two small beds consistently contained more species of fish than one large bed of the same area. When using artificial seagrass beds, however, they found these results were not upheld. This could have been due to the small size of the artificial beds, which did not replicate the scale of natural beds. In contrast, Bell *et al.* (2001) found no consistent effects of bed size on resident fauna, including fish. Yet Bell *et al.* (2001) also found that the small beds of *Halodule* seagrass were not of poor quality in terms of fish densities; the densities of fish in the small and large beds were often similar and no species found in the large beds were missing from the small beds

Eggleston *et al.* (1999) considered the response of fish and macro-invertebrates to different-sized plots of artificial seagrass. The only groups that responded to plot size were the grass shrimp (*Palaemonidae*) and other mobile crustaceans (isopods and amphipods), which were in greater densities in the smaller plots. Other studies of invertebrates in seagrass habitat have had varied findings. Hovel and Lipcius (2001) found the survival of juvenile blue crab was higher in smaller beds of seagrass than in larger beds, whereas Eggleston *et al.* (1998) found the reverse. Hovel and Lipcius (2001) found that the density of seagrass shoots as well as bed size influenced predation on juvenile blue crabs. Further investigation found connectivity of beds to be more influential than bed size or structural complexity in determining rates of predation on the juvenile crabs (Hovel and Lipcius 2002), with crabs in isolated patches being more vulnerable. In contrast, in a mark–recapture experiment with juvenile hard clams (Irlandi 1997), larger beds (5–10 m across) of seagrass had higher survivorship than smaller beds (~1 m across). With controls for below-ground biomass and shoot density (using artificial beds) there was no significant difference in the proportion of clams recovered live from large (4 x 4 m) and small (1 x 2 m) beds. In this experiment the scale of the artificial plots was similar to that of the natural beds surveyed. Similarly, Irlandi *et al.* (1999) found no long-term effects of patch size on the growth and survivorship of juvenile bay scallops.

Generally, when considering the effect of bed size on invertebrates it seems other factors are more influential, such as shoot density, or connectivity of patches.

Larval processes are considered crucial for the abundance and diversity of juvenile and adult fish in most habitats. The advantage of many small beds of seagrass over just one bed is an increase in the probability of interception by larvae and recruits (Paine and Levin 1981; Sousa 1984; McNeill and Fairweather 1993). This increases the overall colonization of the network of beds, as compared with a single larger bed of seagrass (Bell *et al.* 1987; Sogard 1989; Worthington *et al.* 1992a; Eggleston *et al.* 1998). The recruitment process is variable in both time and space (Sogard 1989; McNeill *et al.* 1992; Worthington *et al.* 1992b) although the supply of larvae is quite predictable spatially (McNeill *et al.* 1992; Jenkins *et al.* 1998). This suggests that many smaller beds will have the advantage over one large bed in terms of intercepting larvae and increasing overall recruitment. The size of a bed may indirectly affect recruitment through other mechanisms such as changing predator distribution, abundance and foraging behaviour (Irlandi 1997; Bologna and Heck 1999; Irlandi *et al.* 1999; Michelli and Peterson 1999; Hovel and Lipcius 2001, 2002) and modifying water flow (Eggleston *et al.* 1998).

The shape of a seagrass bed could also influence the diversity and abundance of fish by similar mechanisms. Some researchers propose that the high perimeter:area ratio of smaller habitats may offer more advantages than one large habitat with a low perimeter:area ratio (Paine and Levin 1981; Sousa 1984; McNeill and Fairweather 1993). A long, narrow bed (with a high perimeter:area ratio) may have an increased likelihood of intercepting more larvae than a rounder bed (with a low perimeter:area ratio).

Further investigation is required to determine if the smaller beds have higher numbers of prey organisms than larger beds. Seagrass beds are commonly used by macro-invertebrates. Dietary studies of fish assemblages in seagrass show that crustaceans are the major food item in fish diets (Burchmore *et al.* 1984; Pollard 1984; Robertson 1984; Edgar and Shaw 1995a, 1995b). Few fish species are capable of directly using plant material (Edgar and Shaw 1995a). Some amphipods, isopods and polychaetes sometimes swim above bed of seagrass at night in large numbers (Robertson and Howard 1978). They can also travel long distances from seagrass habitat (Virnstein and Curran 1986), so a small bed may attract macrofauna from a larger surrounding area. This macrofauna can then attract predator fish species to the bed or support larger numbers of fish than the size of the bed would suggest.

In summary, the gaps in research concern the effects of the size and shape of seagrass beds on the diversity and abundance of fish; this includes the recruitment processes of fish to beds of seagrass and the interaction of prey invertebrate species with predator fish.

ARE THERE EDGE EFFECTS IN BEDS OF SEAGRASS FOR FISH?

Ecotones, where two habitats meet such as sand and grass, are considered to be areas of high biodiversity. They provide two habitats for shelter, enhanced biotic interactions (such as predation or competition) and allow the mixing of two biotas from two separate habitats (Fox *et al.* 1997). An ecotonal effect occurs when the abundance of organisms can change about the edge of habitat (Lidicker 1999). The edges of seagrass beds may influence the abundance and diversity of fish. There are changes in beds of seagrass as a function of the distance from the edge, such as a decrease in water flow from the edge to the centre (Fonseca *et al.* 1982; France and Holmquist 1997).

Numerous studies have considered the infaunal response to the edges of seagrass habitat (Summerson and Peterson 1984; Irlandi 1994; Bologna and Heck 1999; Bowden *et al.* 2001; Hovel and Lipcius 2002; Tanner in press). Hovel and Lipcius (2002) found that the densities of juvenile blue crabs were greater in the interior of seagrass beds than at the edges (independent of shoot density). Tanner (in press) also studied some epifaunal organisms (not fish) and found that they tended to show a relatively strong edge effect within 1 m of the edge; however, no distinctive edge-associated fauna was detected. Bowden *et al.* (2001) also found some differences in the assemblage structure of small epifauna between the centre and edge of seagrass patches. In a review of the literature on faunal response to fragmentation in seagrass habitats, Bell *et al.* (2001) suggested a preferential use of the edge or interior by seagrass taxa. Some studies found increased survival and growth for the taxa investigated on the edges of seagrass beds (Irlandi 1994; Bologna and Heck 1999) although the risk of predation was greater too (Bologna and Heck 1999; Hovel and Lipcius 2002). The edges of seagrass meadows were found by Sanchez-Jerez *et al.* (1999) to be relatively important for epifauna distribution, depending on taxon and period of the year.

McNeill and Fairweather (1993) hypothesised that one of the reasons smaller beds have greater species richness than large beds is the increased likelihood of sampling an edge in a small bed. In fact, small seagrass beds could be considered to be all edge. An investigation of the existence of edge effects for fish in beds of seagrass is required.

HABITAT HETEROGENEITY OR SPATIAL STRUCTURE OF A SEAGRASS BED

Seagrass beds are considered to be simpler than terrestrial ecosystems in terms of species diversity and structural complexity, and this simplicity may be useful for testing theories of habitat heterogeneity (Robbins and Bell 1994). Heterogeneous environments are considered to promote diversity (Heck and Orth 1980; Parrish 1989; Irlandi and Crawford 1997) and a positive correlation has been found between habitat heterogeneity and the number of fish species in shallow waters off the southern Bothnian sea in Sweden (Thorman 1986).

In terms of structural complexity, some seagrass beds have an even grass cover and can be described as uniform. Other seagrass beds contain numerous patches of sand so that they appear to be broken up and can be described as patchy. The uniform seagrass beds could be considered to be a homogeneous environment (i.e. only one habitat), whereas the patchy seagrass beds could be considered to be a heterogeneous environment (i.e. containing two habitats: seagrass and sand). The spatial structure of a seagrass bed (such as patchy or uniform) may be important in determining fish abundance and diversity (Ferrell *et al.* 1992).

A mosaic of sediment, i.e. sand and seagrass, may directly or indirectly alter the assemblages of fish by numerous means (as outlined by Eggleston *et al.* 1999) including

1. An alteration of predator distribution, abundance and foraging behaviour (e.g. Coen *et al.* 1981; Leber 1985; Main 1987; Bell and Hicks 1991; Danielson 1991; Edgar and Robertson 1992; Irlandi 1994; James and Heck 1994; Irlandi *et al.* 1995);
2. A modification of the hydrodynamics, which can influence settlement of larvae (Eckman 1983; Bell *et al.* 1995);
3. An influence on the accumulation of algae (Kulcycki *et al.* 1981; Holmquist 1994; Bell *et al.* 1995); and
4. Creation of changes in animal behaviour (reviewed by Heck and Crowder 1991); an organism's response to habitat heterogeneity depends on features specific to that organism such as body size and functional group (Eggleston *et al.* 1999).

Irlandi (1994) considered how the percentage cover and spatial arrangement or patchiness of a seagrass bed affected predation on *Mercenaria mercenaria* (hard clams). Predation was higher in the more patchy beds and in the beds with less percentage cover. These findings were supported by further studies using another bivalve, the bay scallop (Irlandi *et al.* 1995). The pink shrimp *Penaeus duorarum* was found to be more abundant in low-energy, continuous seagrass beds than in high-energy, patchy seagrass beds (Murphey and Fonseca 1995). When investigating the differences in infaunal macroinvertebrates in patchy (fragmented) and unfragmented beds of *Zostera marina*, Frost *et al.* (1999) found there was no difference in abundance of organisms or taxonomic groups but there was a difference in the community composition. Similarly, the assemblage of mysids in shallow waters was strongly affected by the heterogeneity of the seagrass habitat (Barera'-Cebrian *et al.* 2002), including two different species of seagrass and patches of sand. Given the influences on the macroinvertebrate community from numerous studies, it is likely that the patchiness or heterogeneity of a seagrass bed could also influence the vertebrate community, i.e. fish. One study examining the fish found within patchy and continuous beds of seagrass found that different fish species vary in their response to the patchiness of a seagrass bed (Crawford *et al.* 1995). More research is required to discern the influences of habitat heterogeneity on fish in seagrass beds.

POSITION OF SEAGRASS BED WITHIN THE ESTUARY

The position within an estuary can influence the abundance and diversity of fish found in a seagrass bed (Bell *et al.* 1988; McNeill *et al.* 1992; Jenkins *et al.* 1996) although the measured effect of that influence can vary. For instance, between June and March of the years surveyed, one seagrass bed was found by McNeill *et al.* (1992) to have up to 73 times the abundance of five species of fish than the other 15 beds surveyed. Yet during the rest of the year, there was no significant difference between this bed and the others. The supply of larvae to this site was thought to have caused the recruitment of large numbers of individuals. In another study, there was a correlation between whiting abundance and distance from the bay entrance (Jenkins *et al.* 1996). Whiting spawn outside the bay, and hydrodynamic modelling demonstrated that a large amount of the variation in abundance at different sites could be attributed to two processes: the variation in currents delivering the larvae, and the exposure of the site to wave action that either kills or relocates the larvae (Jenkins *et al.* 1997). Bell *et al.* (1988) found that the location of the seagrass bed had a significant effect on the abundance of some fish species. They found that the fish were distributed in zones, with some species being more common close to the estuary mouth and others more common in the deeper reaches of the estuary. The estuary surveyed does not possess strong temperature or salinity gradients so they attributed this zoning to different patterns of spawning, larval dispersal and settling behaviour. Similarly, in another study (Hannan and Williams 1998), newly settled juveniles of ocean spawners were concentrated near the entrance of a marine lagoon. The distance of the seagrass beds from the ocean limited the larval distribution. In contrast, newly settled juveniles of lagoon spawners were widely distributed within the lagoon. Therefore, a protected area within an estuary needs to include a number of beds from all regions to ensure that a full suite of fish is protected. Furthermore, the ecological processes that ensure successful spawning, larval dispersal and recruitment need to be identified. Many estuarine fish spawn outside the estuary and use other habitats during stages of their life cycle (Boehlert and Mundy 1988; Hannan and Williams 1998). Hydrodynamic processes can influence seagrass landscape patterns (Fonseca and Bell 1998) and the larval flows that reach a bed (Boehlert and Mundy 1988; Jenkins *et al.* 1996; Hannan and Williams 1998; Etherington and Eggleston 2000;

Smith and Suthers 2000). The hydrodynamic processes that influence a seagrass bed can be a function of its location within the estuary (Kjerfve *et al.* 1992; Cox *et al.* 1993; Kingsford and Suthers 1996; Smith and Suthers 2000). Further research is needed to determine how the hydrodynamic processes within an estuary influence fish abundance and diversity in seagrass beds.

PROXIMITY TO OTHER HABITATS, E.G. MANGROVES

The effectiveness of a MEPA is thought to be influenced by the distance between habitats within the MEPA, their degree of interconnectedness, and the dispersal ability of individuals from other marine habitats (Goeden 1979; Sammarco and Andrews 1988). Studies have found that the adjacent habitat can play an important role for species associated with seagrass (Heck 1979; Howard 1989; Ferrell and Bell 1991; Fortes 1991; Irlandi and Crawford 1997; Micheli and Peterson 1999). The type of adjacent habitat and its distance from the seagrass bed can affect the diversity of seagrass species (Heck 1979; Sogard 1989).

Seagrass beds and mangrove forests in an estuary are often found within close proximity of one another and it may be predicted that some biotic interchange could occur in nutrients, sediments, larvae, post-larvae recruits, or adult fish and invertebrates. Mangrove habitat is considered to be important for many fish species as habitat for the post-larva and juvenile stage (Robertson and Duke 1987, 1990a, 1990b; Little *et al.* 1988; Laegdsgaard and Johnson 1995). Laegdsgaard and Johnson (1995) found that the majority of the juvenile fish in mangroves in summer were non-residents and therefore not confined to the mangrove habitat. One may predict that seagrass habitat close to mangrove could benefit in terms of species abundance and diversity from the proximity of the mangrove. Some researchers have found that the mangrove habitat had more fish and/or species of fish than the adjoining seagrass habitat (Robertson and Duke 1987; Thayer *et al* 1987; Laegdsgaard and Johnson 1995), but they compared the mangrove with seagrass only at high tide. It is conceivable that at low tide, when the mangrove is exposed, the fish might reside in an adjacent seagrass bed until the tide immerses the mangrove habitat again. A study investigating the abundance of Caribbean reef fish found that seagrass beds close to mangroves had a greater species richness of nursery fish than beds with no adjacent mangrove habitat (Nagelkerken *et al.* 2001); those authors suggested that the mangroves enhanced the species richness of the seagrass by an unknown interaction.

However, the interconnectedness of habitats can have negative effects on some organisms. For instance, the species richness of macroinvertebrates on intertidal oyster reefs separated from seagrass and saltmarsh was higher than on reefs connected with either of these habitats. The seagrass and saltmarsh were shown to act as corridors for the movement of predatory blue crabs and hence to facilitate higher predation rates in the reef habitat (Micheli and Peterson 1999).

More studies considering the interaction of seagrass with adjacent habitat is required, including the effect of varying distances between the two habitats.

IMPLICATIONS FOR DESIGNING ESTUARINE PROTECTED AREAS

Smaller seagrass beds cannot be overlooked when protected areas (PAs) within estuaries are designed, although further research is required to confirm the influence of size of seagrass bed on fish. On the basis of the findings of McNeill and Fairweather (1993), however, a network of PAs should include small beds. Both patchy and uniform beds must be included to conserve the different assemblages of fish found in each (Crawford *et al.* 1995). Since different species assemblages are found in beds of varying distance from the estuary mouth, beds in all regions must be included to ensure the protection and survival of all fish species. More research, however, is required to understand the processes by which these different patterns of fish distribution exist.

The protection of adjoining mangrove habitat as well as seagrass habitat is recommended, although the interactions of adjoining habitat with beds of seagrass are not yet understood. Furthermore, research is required on the influence of the interconnectedness of seagrass habitat, the shape of seagrass beds and the heterogeneity of seagrass beds on fish abundance and diversity. Similarly, the existence of edge effects for fish in beds of seagrass also needs to be investigated. Finally, more experimentally designed projects are required to answer specific questions on the influence of seagrass beds on fish.

Seagrass habitats are ideal for the application of research considering landscape ecology theories (Bell and Robbins 1994). Seagrass beds occur in meadows that can extend over kilometre-wide areas (i.e. historically defined landscape) and it is at this scale of patchiness that marine studies are relatively scarce (Robbins and Bell 1994). An understanding of the seascape processes that determine the diversity of fish within seagrass beds and estuaries is where we should be aiming.

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ESTABLISHING MARINE PROTECTED AREAS IN VIETNAM: A CAPACITY-BUILDING APPROACH

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Abstract

Vietnam currently three legislated protected areas with marine components (Cat Ba NP, Con Dao NP and Ha Long Bay World Heritage Area). The Government of Vietnam has prepared draft legislation providing for the declaration and management of a national system of 15 marine protected areas (MPAs) by 2010. The first of these, the Hon Mun Pilot MPA, was developed in cooperation with IUCN The World Conservation Union, Vietnam Office and theGreat Barrier Reef Marine Park Authority (GBRMPA), with primary funding from the Global Environment Facility.

The Hon Mun project aims to develop a multiple-use MPA that protects coastal ecosystems whilst enabling local communities to improve their livelihoods. The project will provide long-term environmental and socio-economic benefits by developing an effective Provincial Marine Parks Authority and a system for co-management with local resource users, through the following: participatory planning and management by stakeholders; development of alternative income sources to discourage activities associated with excessive resource use; capacity building through management training and public education; and a financially self-sufficient management system.

The paper reviews the status and development of Marine Protected Areas (MPAs) in Vietnam, examines those factors likely to influence the effectiveness of Hon Mun and other MPAs, and identifies potential directions for the future development of a national system of MPAs in Vietnam.

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Keywords: Vietnam, Hon Mun MPA, capacity building, governance, poverty alleviation

INTRODUCTION

With approximately 3260 km of coastline (excluding island coastlines), more than 3000 inshore and offshore islands and an Exclusive Economic Zone (EEZ) of almost one million square kilometres, marine and coastal resources constitute an important asset for Vietnam. Vietnam's maritime territory forms part of the East Asian Seas Marine Region defined by the IUCN Commission on National Parks and Protected Areas (now the World Commission on Protected Areas) (Kelleher *et al.* 1995). The East Asian Seas Marine Region also includes marine and coastal waters of Brunei Darussalam, Cambodia, Indonesia, Malaysia, Philippines, Singapore, Thailand, and Vietnam, and arguably supports the most diverse marine flora and fauna in the world.

The diversity of Vietnam's coastal and marine natural resources provides the human population with an abundance of benefits, such as marine products (fish, invertebrates, algae, etc.), energy (crude oil, gas and fuel wood), raw material (mineral resources), tourism, recreation and coastal protection (NEA 1994; MOSTE 1995; ADB 1999). About 50% of Vietnam's provinces and cities are located along the coastline. The majority of Vietnam's rice production occurs in the Mekong River Delta and Red River Delta areas, and these two delta systems, which comprise areas of mangroves, saline lagoons, coastal wetlands and streams, underpin much of Vietnam's economy (UNDP/MPI 1999).

Exploitation of these marine resources has accelerated over time, mainly as a consequence of the high densities of human populations in the coastal zone, which are highly dependent on these resources for their economic subsistence. Destructive and unsustainable fishing practices, uncontrolled collection of endangered marine organisms, high pollution levels, and sediment load from land-based activities have had adverse effects on some of the most important ecosystems and habitats in Vietnam.

The fisheries sector has witnessed substantial economic growth during the past decade. Total fisheries production and the contribution of fisheries to the national GDP have both grown substantially and approximately 30–40% of protein consumption is supplied from marine products. Most fisheries occur in the inshore and near shore areas (<30 m deep), where the majority of recent growth in fisheries productivity has come from increased aquaculture. Capture fisheries have either stabilised or decreased, with catch per unit effort and fish stocks having declined in near shore fisheries. There are plans to extend into deep-water fisheries. Destruction of coastal wetlands and mangroves from coastal development, agriculture and aquaculture is affecting the long-term viability of the fisheries sector (NEA 1994; UNDP/MPI 1999; ADB 1999).

The alarming rate at which coral reefs and other marine ecosystems are declining has not yet been fully acknowledged by decision-makers and the general public. The present national system of protected areas has mainly focused on the protection of terrestrial flora and fauna. At present, the information on other valuable and representative marine areas is scattered among different agencies, research institutions and organisations, and is not easily accessible. This often results in a lack of direction among the end-users and an inefficient regulatory system.

A representative system of marine protected areas (MPAs) is an important step toward the conservation of marine biodiversity in Vietnam, especially with regard to the protection of coral reefs, seagrass beds and other critical habitats for endangered species. Ideally, this system should be enmeshed with a larger Integrated Coastal Zone Management framework.

STATUS OF COASTAL AND MARINE BIODIVERSITY IN VIETNAM

Vietnam's 3,200 km coastline stretches over thirteen degrees of latitude and includes both subtropical and tropical environments which host a wide range of inter-linked coastal and marine ecosystems, including mangroves, seagrass beds, coral reefs, lagoons, dunes, beaches, estuaries, and upwelling areas (NEA 1994; Vo Si Tuan 1998).

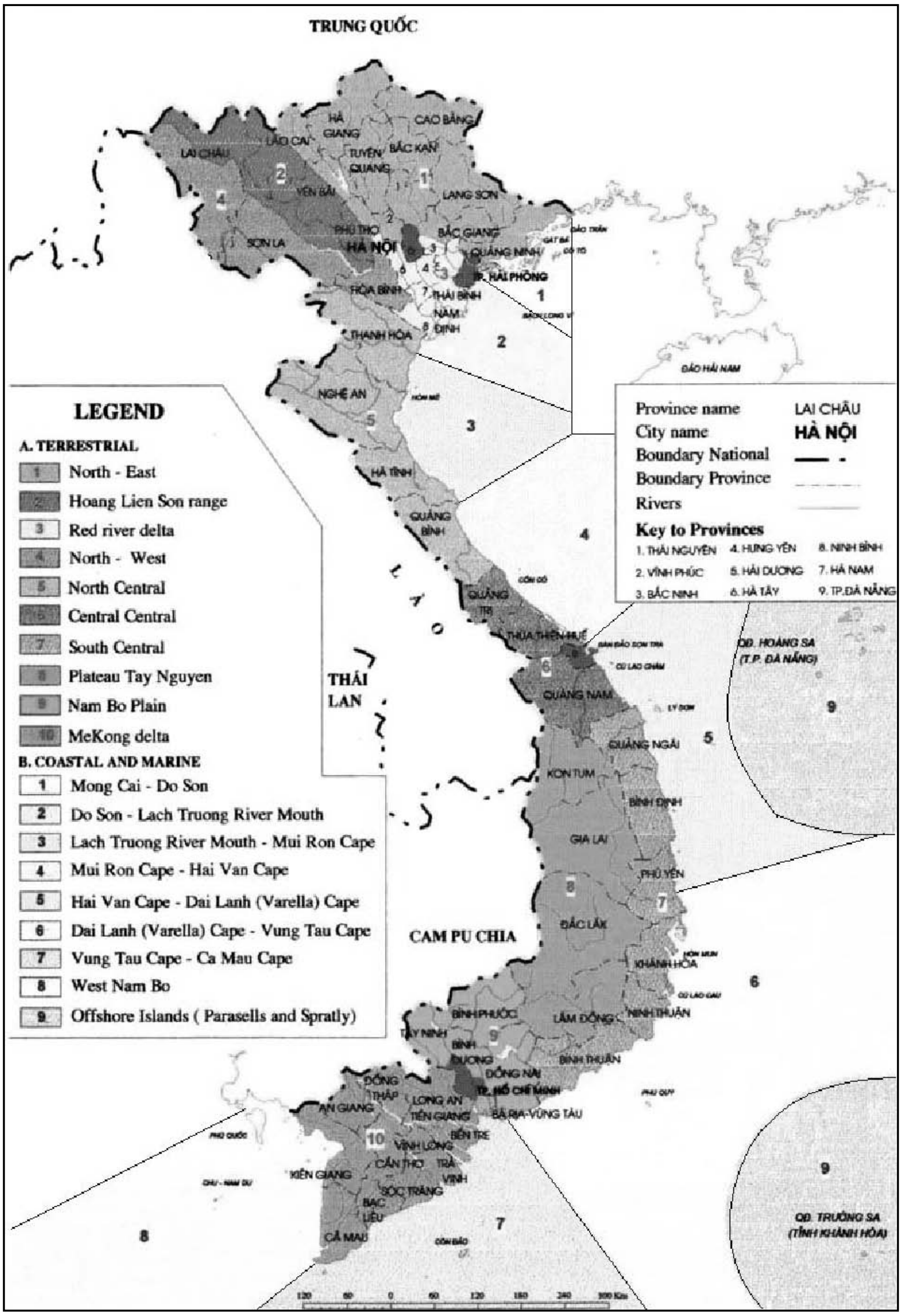
Vietnamese marine research institutions have identified 11,000 marine species (Nguyen Chiu Hoi *et al.* 2000). An estimated additional 1,290

species of plants and animals also occur on islands. Species diversity increases from north to central and southern regions, and fish abundance is higher in offshore than inshore coral reefs (Chou 2000). Vietnam's 350 species of hard corals is a slightly lower number than those found in Indonesia and the Philippines, which have 450 and 400 species respectively (Chou 2000). There are key gaps in information available on marine biodiversity, such as the distribution of coral reefs and seagrass beds in central Vietnam. The lack of comprehensive information hinders establishment of a bioregional framework for Vietnam. However, initial studies have identified nine coastal and marine bioregions (Fig. 1) in Vietnam (UNDP/MPI 1999).

The main threat to coastal and marine biodiversity in Vietnam appears to be the over-exploitation of marine fishes and invertebrates. A large portion of Vietnam's coral reefs, especially in the more accessible areas, are severely degraded. An important contributing factor to the declining health of coastal and marine biodiversity is the harvesting of coastal marine resources under an open and unregulated access regime; this generally results in resource over-exploitation. Additional threats include: destructive fishing methods (e.g. dynamite and cyanide); sedimentation; loss of mangrove habitat caused by illegal logging, salt production and aquaculture; plantation of new mangrove forests; conversion of wetlands to agriculture or aquaculture; land-based pollution; uncontrolled tourism development; mineral exploitation; coastal construction; and mining of corals (NEA 1994; ADB 1999; UNDP/MPI 1999).

The conservation of coastal and marine resources has been recognised as a high priority in the National Biodiversity Strategy (MOSTE 1995; Chou 2000; MOSTE/NEA 2000). MOSTE has identified 83 rare, threatened or endangered marine species listed in the Vietnam Red Book. The green (*Chelonia mydas*), hawksbill (*Eretmochelys imbricata*) and olive ridley (*Lepidochelys olivacea*) sea turtles have traditional nesting sites along the Vietnamese coast. However, the leatherback (*Demochelys coriacea*) and loggerhead (*Caretta caretta*) sea turtles have not been documented in Vietnamese waters for many years. The endangered dugong (*Dugong dugon*) has been reported in the waters of Con Dao National Park and the Phu Quoc Islands (Vo Si Tuan 2000, Marsh *et al* 2002).

Marine & Coastal Bioregions of Vietnam



1. *Mong Cai to Do Son:* Dominant tidal dynamic; estuarine coastline with mud sediments.
2. *Do Son to Lach Truong River Mouth:* Riverine flows; deltaic river mo uth, sand-mud sediments.
3. *Lach Truong River Mouth to Mui Ron Cape:* Riverine flows and wave action; the seashore is a sandy plain.
4. *Mui Ron Cape to Hai Van Cape:* Seashore currents and waves; the coast consists of sand dunes and trapped lagoons.
5. *Hai Van Cape to Dai Lanh Cape:* The land-sea interaction relatively balanced; the coastline consists of capes, small deltas, small lagoons and bays.
6. *Dai Lanh Cape to Vung Tau Cape:* The land-sea interaction relatively balanced; the coastline consists of capes, small deltas, small lagoons and bays.
7. *Vung Tau Cape to Ca Mau Cape:* Dominated by flows from the Mekong River; coastline is deltaic with mangrove forests and sand-mud sediments.
8. *Ca Mau Cape to Ha Tien (Gulf of Thailand):* South-westerly waves; coastline is deltaic with mangrove forests and sand-mud sediments.
9. *Offshore Islands (Parasells and Spratly Archipelago):* Almost all islands are coralline.

Source: UNDP/MPI 1999.

Fig. 1. Marine and coastal regions of Vietnam (UNDP/MPI 1999).

Table 1. Marine species in Vietnam (ADB 1999; Chou 2000 ; Nguyen Chu Hoi *et al.* 2000).

Marine Species in Vietnam	
• 21 marine mammals	• 1647 crustaceans
• 2175 marine fish	• 350 echinoderms
• 5 marine turtles	• 2500 molluscs
• 12 sea snakes	• 700 polychaetes
• 350 hard corals (Scleractinia)	• 15 seagrasses
• 120 soft corals	• 653 macroalgae
• 73 Gorgonian sea fans	• 94 mangrove species
• 150 sponges	

NATIONAL SYSTEM OF MARINE PROTECTED AREAS

Depending on the source of information, or the definition of a MPA, the number of (existing) protected areas in Vietnam, with a coastal or marine component, varies between two (Kelleher *et al*. 1995), three (Chou 2000), twenty (ADB 1999) and twenty-two (Nguyen Chu Hoi 2000). There is no legal framework for the establishment of MPAs in Vietnam, and areas referred to as 'marine' protected areas, including terrestrial protected areas that claim to include marine zones, are without the necessary legal basis to support such a designation.

Two designated protected areas, Con Dao National Park (14,000 ha marine area and 20,500 ha marine buffer zone) and Cat Ba National Park (5,400 ha marine area) contain small marine areas. These National Parks are managed by the Forest Protection Department (FPD) of the Ministry of Agriculture and Rural Development (MARD).

A number of other protected area categories with significant marine areas exist in Vietnam. These comprise Ramsar Sites, World Heritage Sites, and Man and the Biosphere (MAB) Reserves:

- Xuan Thuy (Red River Delta) Ramsar Site
- Vinh Ha Long
  (Ha Long Bay) World Heritage Area
- Can Gio (Ho Chi Minh City) MAB Reserve

Since the mid-1990s, the Vietnamese National Government and the international community have invested considerable resources in establishing and managing marine protected areas in Vietnam. During 1998–99 the Hai Phong Institute of Oceanography, under the guidance of MOSTE/NEA, prepared a shortlist of 15 proposed MPAs to be included in a national MPA system (Nguyen Chu Hoi *et al*. 1998). These areas were selected on the basis of biodiversity and wilderness values, severity of threats to conservation in each area, and feasibility, including socio-economic factors. The Ministry of Science Technology and Environment/National Environment Agency (MOSTE/NEA) has been allocated the responsibility for the establishment a national system of 15 MPAs by 2010 (MOSTE/NEA 2000; MOFI 2001).

An important role for central Government agencies, in particular the Ministry of Planning and Investment (MPI), will be the management of national and international finances through the determination of priorities for MPAs. The central role of MPI in coordinating the national system of MPAs will also provide political support to the designated management agencies to enable them to implement their responsibilities for priority MPAs (MPI/Danida 2000). Through systematic planning for a national system of marine protected areas, the Government of Vietnam aims to link the development of MPAs with national development priorities and to allow for management decisions on a system-wide basis – including the development of an appropriate institutional framework for management of MPAs (MPI/Danida 2000).

From an ecological standpoint, the MPA program will embrace concepts of representativeness, natural integrity, biodiversity and replenishment. From a socio-economic standpoint, the program aims to ensure the long-term sustainability of local communities through the ecologically sustainable use of renewable natural resources and a reduction in the over-exploitation of resoures that is characteristic of many coastal and marine areas.

The proposed National System of Marine Protected Areas (Fig. 2) comprises

- 6 areas in the North of Vietnam: Tran Island, Co To Island, Cat Ba Island, Bach Long Vi Island, Hon Me Island and Con Co Island;
- 6 areas in the central region: Hon-Son-Tra (Hai Van area), Cu-Lao-Cham, Dao Ly Son, Hon Mun (Bich Dam), Hon Cao (Vinh Hao), Phu Quy;
- 1 area in south-east Vietnam: Con Dao;
- 1 area in south-west Vietnam: Phu Quoc; and
- 1 area in the Truong Sa-Hoang Sa (South China Sea): Truong Sa (Spratlys Archipelago).

(Nguyen Chiu Hoi 2000 ; MOFI 2001).

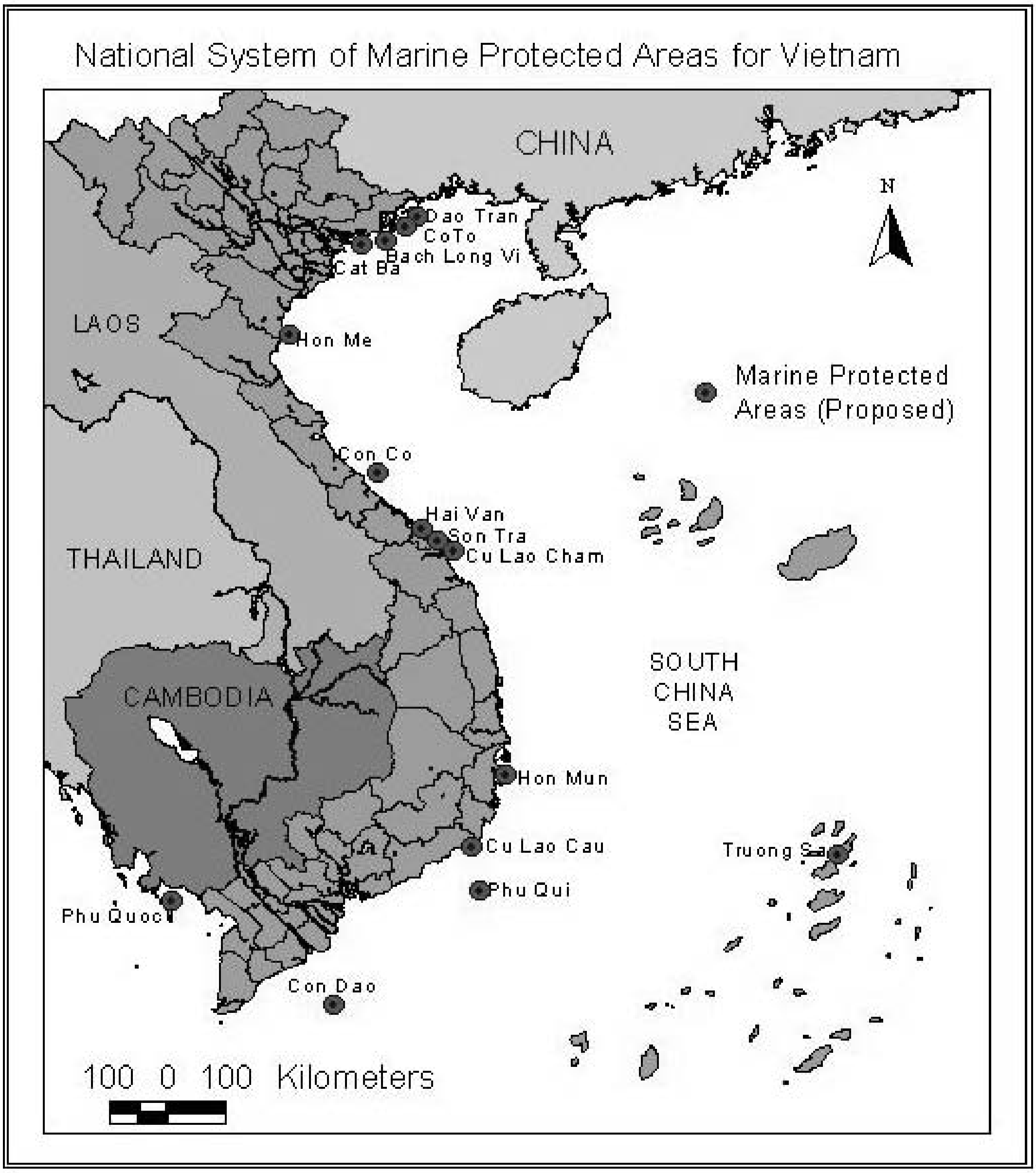


Fig. 2. Proposed sites for Vietnam's National System of Marine Protected Areas.

Management responsibility for Marine Protected Areas

The sustainable management of Vietnam's marine resources faces a number of institutional constraints. One of the most significant of these relates to the formal administrative jurisdiction for marine areas. There is a clear system of jurisdiction and administration for the terrestrial areas of Vietnam, but no comparable system exits for the country's marine areas. Coastal provincial jurisdiction is in practice assumed to end at the limit of the coastline (low water) with no provincial boundaries extending into the marine areas. Where Provincial governments are involved in marine management, they do so on a purely sectoral basic (e.g. fisheries, marine transportation, etc.). The result is that the maritime areas of Vietnam are treated as a completely open-access areas with few of the government and administration controls that exist for terrestrial areas.

There are also no laws providing for the establishment of MPAs, even as extensions of terrestrial parks. Governance responsibilities are also somewhat ambiguous because different Ministries have been legally responsible for certain aspects of marine resource management, specifically MOSTE, MARD and the Ministry of Fisheries (MOFI) (Carew-Reid 2002; NEA/IUCN 2000). A summary of these responsibilities is provided below:

- **MOSTE/NEA** (National EnvironmentAgency) is responsible for the administration of the *Law on Environmental Protection (1993)*. This law is broad and provides strategic direction for environmental protection in Vietnam. In 1994, it was followed by Decree No. 175/CP, which provides guidance for its implementation. The Law contains a broad mandate for environmental impact assessment, and it establishes environmental quality standards specifying the provisional environmental quality criteria that are to be used for monitoring and inspections of projects and activities. Vietnam's Biodiversity Action Plan (1995) indicates that a system of protected areas of representative marine ecosystems may fall under MOSTE responsibility (MOSTE 1995). MOSTE responsibility also includes endangered or migratory species and Ramsar wetlands. (At the 2002 National Assembly Meeting, MOSTE was restructured into two new Government Ministries: the Ministry of Resources and Environment; and the Ministry of Science and Technology.)
- **MARD** (Ministry of Agriculture and Rural Development) is in charge of the National System of Protected Areas, but as its jurisdiction ends at the shoreline the basis for its role in MPA establishment and management is unclear. MARD responsibilities also include wildlife protection and the management of CITIES listed species.
- **MOFI** (Ministry of Fisheries) has recently been assigned responsibility for establishing and managing the national system of marine protected areas, although this mandate has yet to be supported by a Government Decree (Nguyen Chu Hoi 2000). The Fisheries Resources Protection Department (FRPD), under MOFI, is responsible, through the Provincial-level FRPDs, for the implementation of the *Ordinance on Conservation and Management of Living Aquatic Resources (Decree 18, 1986)*, which provides opportunities to establish fisheries protection zones to achieve conservation goals (ADB 1999; Nguyen Chu Hoi 2000). Provincial FRPDs are under the administrative supervision of Vietnam's Provincial governments, and receive technical, planning and management supervision from the national FRPD.

In the intermediate term, management of MPAs has been placed under the direct supervision of the Fisheries and Environmental Conservation Department of the MOFI in cooperation with related Ministries, sectors and local authorities (MOFI 2001). The Ministry of Fisheries is responsible for the management of MPAs, including

- the development of legal documents and policies related to the establishment of MPAs (official approval is obtained through the National Assembly);
- the development of programs and project proposals to expand the system of MPAs;
- the establishment of management boards for MPAs under the direct management of MOFI; and
- publicity, training and professional development for marine conservation.

It is likely that MOFI will have overall management responsibility for the national system of marine protected areas, but that sites with a terrestrial as well as a marine component will be managed by MARD together with MOFI. This situation is likely to continue for the foreseeable future. Management regulations for marine protected areas are also under discussion, but these are likely to be broad and flexible in order to allow management regulations to be tailored to specific geographical areas and management requirements.

The management of terrestrial National Parks is being reallocated to the provincial level of government (Carew-Reid 2002) unless an area is of 'national interest' or crosses provincial boundaries. It follows that management of some MPAs may also be allocated to provincial levels. The MOFI (2001) has stated that where such management is allocated to the provincial level, the responsibilities of the Provincial Peoples Committee (PPC) are to

- manage MPAs under the guidance of MOFI and related Ministries;
- develop management programsand policies for the MPA;
- organise and resource management agencies and enforcement programs within provincial boundaries; and
- work with MOFI to solve inter-sectoral and inter-provincial issues.

At the provincial level, work is to be undertaken through the Fisheries Resources Protection Branches of the Provincial Department of Fisheries or Department of Agriculture and Regional Development (provincial level agencies are under the supervision of the PPCs) – 37 such agencies exist with a combined staff of more than 1000 and 90 fisheries inspection boats (MOFI 2001).

Marine Protected Area Legislation

The first step towards developing more specific legislation for marine protected areas has been taken by MOFI through the development of *Draft Regulation on Marine Conservation Area Management (2000/QD-TTg).* This stipulates the general provisions and regulation for MPA management and includes a short description of the proposed categories for marine conservation areas. It also provides for the formation of a MPA Management Board and identifies a mechanism for state management and implementation. The regulation provides the basis for establishing a broader national legal framework for MPAs.

MOFI (2001) has identified three categories of MPAs for use in Vietnam. There are no specific criteria for these categories although there are recommendations for a management structure for each type. The proposed categories are

- Marine National Park (IUCN Category II);
- Marine Wildlife Sanctuary (IUCN Category IV); and
- Marine Natural Resources Protection Reserve (IUCN Category VI).

Marine National Parks
Under the instructions of MOFI, a management board will be established for each Marine National Park. Management boards are under the direct supervision of the MOFI and have a Director and Deputy Director appointed by the Ministry. Management Boards are corporate entities and possess their own financial accounts and corporate stamp, and have the legal right to take initiatives to implement assigned management duties.

Marine Wildlife Sanctuary/Marine Natural Resources Protection Reserves
Management Boards will be formulated under the direct supervision of the MOFI, the Provincial Department of Fisheries, or the Provincial Department of Agriculture and Rural Development (depending on the responsible agency). Management Boards are established by approval of MOFI after review of proposals from the FRPD of MOFI or the PPC as appropriate. The transfer of management authority for these areas to other socio-economic organisations, including fishermen, may be considered by the Minister for Fisheries.

None of the proposed candidate MPA sites have yet been gazetted (Nguyen Chu Hoi 2000 ; MOFI 2001). However, it is expected that they will be allocated as follows:

- Marine National Parks (3): Cat Ba, Con Dao, Hon Mun;
- Marine Nature Species/Habitat Conservation Areas (5): Co To, Con Co, Hai Van, Hon Cau, Truong Sa, (Son Tra); and
- Marine Nature Reserves (6): Dao Tran, Bach Long Vi, Hon Me, Cu-Lao-Cham, Phu Quy, Phu Quoc.

FACILITATING MPA DEVELOPMENT

The establishment of a national system of MPAs in Vietnam has been progressed as a collaboration between the Vietnamese Government, international organisations such as the World Bank and UNDP, non-government organisations and the international donor community. There are several important donor-supported projects, listed below, under implementation in the marine/coastal zone; in particular, the system of Marine Protected Areas currently under development has received substantial donor interest.

- UNDP/GEF/Danida/WWF is developing a project proposal for the Con Dao MPA; this project proposal focuses on coastal biodiversity conservation and sustainable uses in Con Dao Island, a national park offshore of Ba Ria-Vung Tau Province.
- The Danish International Development Agency (Danida) has recently approved funding for supporting the Marine Protected Areas Network through this development project, the 'sustainable management of marine and coastal natural resources in a national protected areas system'. The project will be implemented at the National Government and Provincial levels, in cooperation with MOFI and the Quang Nam PPC through two distinct but integrated sub-projects. The national-level sub-project will assist the Government of Vietnam to (*i*) develop a legal and policy framework for national MPAs system and (*ii*) establish a coordinating mechanism for the development of a multi-sectoral approach to marine management issues. The provincial-level sub-project will be the implementation of a marine protected area in the Cu Lao Cham Archipelago (Quang Nam province).
- The National Environment Agency Integrated Coastal Zone Management Project is funded by the Government of the Netherlands: this project is working with Vietnamese counterparts to establishment an ICZM 2000 + Programme in Vietnam. The project aims to

facilitate approaches to integrated planning and development in Vietnam's coastal zone and is focused on Thua Thien Hue, Nam Ha and Ba Ria Vung Tau Provinces.

- NOAA and the United States East Asia Pacific Environment Initiative is funding preliminary work on integrated management of the North Tonkin Archipelago, including the Ha Long Bay World Heritage Site. This work is expected to result in the production of a GEF Block B International Waters/Biodiversity funding application.
- The UNEP/GEF Coastal and Marine Environmental Management in South China Sea Project is regional in scope and comprises seven major components: mangroves, coral reefs, seagrasses, wetland, pollution issues, legal and institutional management, and over-exploitation of marine biodiversity.
- The Hon Mun Pilot MPA Project for Vietnam is supporting the development of the Hon Mun MPA, which is the first protected area in Vietnam to focus primarily on the conservation of marine biodiversity. This project is implemented by the Ministry of Fisheries, Khanh Hoa PPC and IUCN – The World Conservation Union, and funded by the Global Environment Facility/World Bank (GEF/WB), Danida, the Government of the Socialist Republic of Vietnam, and IUCN-The World Conservation Union. The project is seeking to ensure that Hon Mun MPA is managed as a regional example of 'best practice' and as a resource for the development of approaches to the management of marine protected areas in Vietnam.

HON MUN MARINE PROTECTED AREA PILOT PROJECT

The Hon Mun Marine Protected Area, about 10 km off the coast of Nha Trang City in Khan Hoa Province, is one of the 15 areas that have been approved by the Government for inclusion in the National System of Marine Protected Areas and will be the first MPA to be implemented under that program.

The Hon Mun Pilot Marine Protected Area Project was developed under a partnership between IUCN's World Commission on Protected Areas – Marine Program, the World Bank and the Great Barrier Reef Marine Park Authority to promote the development of a Global Representative System of Marine Protected Areas. This partnership has produced a global assessment of marine protected areas as well as a series of three GEF-supported pilot marine protected area projects in Samoa, Tanzania and Vietnam. IUCN–The World Conservation Union, through its network of regional offices, is the implementing agency for these pilot MPA projects.

The national focal point for this 4-year project is MOFI, with on-ground work being conducted by MOFI, the Khanh Hoa PPC and the IUCN Vietnam Office under a joint Memorandum of Understanding. A set-up phase of 18 months began in 2001, to be followed by an implementation phase of 30 months.

The Hon Mun MPA covers about 12,000 ha and includes 8 islands. There is an in-park population of 5000 people in 7 villages on the islands, of whom 95% gain their income from fishing. The Hon Mun area receives about 100,000 tourists annually (60% Vietnamese and 40% international tourists); major activities include diving and snorkelling, water sports, boat trips and beach visits. The area contains a regional freight and fishing port; approximately 66 tourist boats are based at the Cau Da Port. Fishing in the area includes subsistence fisheries, commercial seine netting, line fishing and trawling, the aquarium trade, and aquaculture for lobster and fish species. Swiftlet nests are collected from offshore islands (within the protected area) for birds' nest soup (IUCN 1999).

The marine environment of the area is dominated by fringing reefs and seagrasses on a sandy bottom. Known biodiversity of the area, based on surveys by the Nha Trang Institute of Oceanography, includes 193 species of coral and 176 species of fish (which is the highest diversity of coral or fish recorded for mainland Vietnam), as well as 112 crustacean species, 27 species of echinoderms, 112 mollusc species, and 104 species of marine algae (WWF 1993; Vo Si Tuan 1996; IUCN 1999).

Major threats to conservation of the area include the over-harvesting of resources, illegal fishing (in particular the use of explosives and cyanide), visitor damage from anchoring and tourism wastes, poorly planned coastal developments and pollution from the land.

Through the development of a zoned, multiple-use MPA that protects important examples of coral reef, mangrove and seagrass ecosystems, the project aims to enable local island communities to improve their livelihoods and, in partnership with other stakeholders, effectively protect and sustainably manage the internationally significant and threatened marine biodiversity at Hon Mun as a model for collaborative MPA management in Vietnam (IUCN 1999). The project places a strong emphasis on building partnerships among stakeholders, establishing a financially self-sufficient management system and providing long-term socio-economic benefits to the local communities that rely on the resources of the Hon Mun area.

The Hon Mun MPA project will develop an effective Provincial MPA Authority and a system for co-management with local resource users. Implementation will involve four components:

1. Participatory planning and management by stakeholders.
2. Development of alternative income-generating (AIG) activities to as an alternative to activities associated with excessive resource use.
3. Capacity building through management training and public education.
4. Monitoring and evaluation of program success.

Participatory planning and management by stakeholders

Zoning of the marine park will be developed, in participation with local communities, to allow for a range of management controls in each zone. These zones will be based upon biodiversity assessments and socio-economic assessments to allow for managed use of the area to be continued by local people.

The proposed zones are

- *Biodiversity zone* – to provide for a no-fishing zone covering 10–20% of the total area of the MPA; education and research activities will be allowed in addition to nature-based tourism such as diving and snorkelling.
- *Buffer zone* – to allow for traditional fishing and permitted tourist activities including boating, diving and swimming. Trawling will be prohibited.
- *Aquaculture zone* – to allow for planned aquaculture (in conjunction with AIG activities).

Local communities will be given a key role in enforcement in partnership with the MPA Authority. This approach has proved effective in recent enforcement of bans on dynamite fishing.

Development of alternative income sources

The project includes a substantial alternative income generation (AIG) component (US$ 225,000), which aims to provide alternative employment opportunities to assist with the removal of fishing pressures and to replace economic benefits that may be displaced by the protected area. AIG investment under the project will focus on long-term participatory approaches including a micro-credit scheme under the management of the Women's Union. The project will develop trial models for up to 20 activities including tourism, commercialisation of traditional crafts and cage aquaculture. A particular focus will be placed on women and poorer households.

Capacity building through management training and public education

The MPA Project will develop a national training centre for MPA management. Key partners in the development of the training centre are the IUCN, MOFI and its Research Institute for Marine Products (Hai Phong) and the Research Institute for Aquaculture (Nha Trang), the Nha Trang Fisheries University and the Nha Trang Institute for Oceanography. An environmental-awareness program is also being developed for local communities.

A financially self-sufficient management system

The project has initial funding for four years, after which it is required to become financially self-sufficient (unless core Government funding is provided). Surveys of tourists conducted by the project team have indicated support for a fee on tourists – this is expected to be in the order of US$1/tourist/day (lower for domestic tourists). This funding model has been used in the Ha Long Bay World Heritage Area where visitor fees constitute approximately 50% of the Ha Long Bay Management Department's funding appropriation. A proportion of funding derived from tourist user fees (approximately 10%) will be allocated to the AIG program.

Monitoring and evaluation of program success

The project will develop a monitoring system to evaluate success of MPA management programs over the long term. The monitoring system, which will include a community-based element, will review both environmental and socio-economic data against the pre-project baseline. It is anticipated that the evaluation process will not only contribute to the adaptive management of the protected area, but will also contribute to the design and development of other MPAs in Vietnam.

CONCLUSION

Vietnam places a growing emphasis on sustainable development and is promoting stronger focus on integrating environmental issues into development. This trend, which is strongly supported by the international donor community, is demonstrated through an increasing series of national strategies and action plans for sustainable development – e.g. the preparation of Vietnam's National Environmental Action Plan and Biodiversity Action Plan (UNDP/MPI 1999).

The national system of MPAs is becoming established significantly more slowly than the system of terrestrial protected areas. This is in part due to jurisdictional overlap and a lack of legislation, as well as to a stronger national focus on the management of terrestrial and agricultural systems and a tradition of donor interest in agriculture and forestry development projects.

There is an emerging Government acknowledgment of the need to manage marine environments and the potential for MPAs to act as a management tool. In establishing functioning MPAs in Vietnam, there is a need to integrate and consolidate the institutional arrangements for marine biodiversity conservation from the central through to local government levels. Provincial and local governments of the coastal provinces are directly responsible for the day-to-day management of MPAs and will need to be adequately resourced accordingly. Central government ministries such as MOSTE and MOFI are responsible for the formulation of management policies/regulations and the provision of technical support and advice to the MPA system, and capacity-building work is being undertaken in this regard.

Significant progress on key issues of jurisdiction and funding allocation has been made through the active supervision of the Ministry of Planning and Investment, a pilot MPA project has commenced, and additional funding is being made available to strengthen the institutional framework for development of a national system of MPAs. However, a number of key challenges remain in order to develop an effective system of MPAs. These include the need to:

- establish and manage priority pilot MPA sites;
- provide sustainable financing – raise Government funding for recurrent and capital expenditure on protected areas;
- improve the efficiency of domestic funding mechanisms and ensure the effective use of expenditure funds;
- resolve ambiguity over institution mandates and responsibilities, as well as legislative frameworks.
- increase government and public awareness of marine conservation issues;
- encourage ecosystem based approaches to the management of fisheries;
- extend marine conservation beyond coral reefs into other marine environments; and
- develop multiple-use MPAs as nodes for sustainable development and poverty alleviation programs, in order to provide direct community benefits.

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Assessing the Importance of Coastal Habitats for Fisheries, Biodiversity and Marine Reserves: A New Approach Taking into Account "Habitat Mosaics"

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Abstract

Effective management of estuarine and coastal fisheries resources requires detailed information on the relationships between the habitats being protected and the fisheries dependent on them. Past research of nekton has focused on comparisons of abundance and species composition between single habitats (e.g. mangroves versus seagrass or vegetated versus unvegetated habitats). These studies have provided valuable insights into the role of coastal habitats for sustaining fisheries and biodiversity but have not considered the importance of adjacent habitats to the overall value of an area. For example, fish are only able to occupy mangrove forests for a restricted amount of any high tide period. The nature of the habitats lower down the shore may be crucial to the overall value of any patch of mangrove for supporting fisheries. We are taking a new approach to assessing the value of estuarine habitats for fisheries and biodiversity that considers the spatial arrangement of different habitats within an area – or the "mosaic" of habitats within the area. The scale of the area for study is defined by the life history and biology of the species of interest. In addition to estimating the abundance, biomass and community structure of nekton (e.g. fish, crustaceans, molluscs), the functioning of mosaics will be studied by estimating growth rates and describing the food webs in different mosaics and the characteristics of the mosaics will be measured. This approach has the potential to be extended to allow much better criteria to be developed for the selection of marine reserves by managers.

Keywords: estuarine habitats, mosaics, spatial arrangement, fisheries, biodiversity, habitat complexity

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Importance of Estuarine Habitats

Estuarine systems comprise a large number of different types of shallow-water habitats, including seagrasses, mangroves, saltmarshes, sand and mudflats and rubble banks, that support diverse communities of plants and animals (e.g. Hatcher *et al.* 1989). The importance of these nearshore estuarine habitats for the survival and maintenance of biodiversity (Hockey and Branch 1997; Brailovskaya 1998), fisheries resources (e.g. Roberts 1995; Kaufman and Dayton 1997; Castilla and Fernandez 1998; Hastings and Botsford 1999) and ecosystem services (e.g. Costanza *et al.* 1997; Peterson and Lubchenco 1997) has led to an increasing focus on the need to design and establish marine reserves and aquatic protected areas as a tool for conservation and resource management (e.g. Allison *et al.* 2002).

From a fisheries perspective, most research has concentrated on evaluating the relative importance of vegetated habitats such as mangroves (e.g. Bell *et al.* 1984; Hatcher *et al.* 1989; Robertson and Blaber 1992; Laegdsgaard and Johnson 1995, 2001), seagrasses (Orth *et al.* 1984; Bell and Pollard 1989; Heck and Crowder 1991; Edgar and Shaw 1995) and saltmarsh (Odum *et al.* 1988; Orth and van Montfrans 1990; Heck and Crowder 1991; Minello and Zimmerman 1992; Thomas and Connolly 2001). Other habitats dominated by structural and topographical relief, including woody debris (Harmon *et al.* 1986; Robertson *et al.* 1991; Everett and Ruiz 1993), rock and oyster reef (Lenihan and Peterson 1998; Harding and Mann 1999; Micheli and Peterson 1999; Lenihan *et al.* 2001) and rubble (Dumbauld *et al.* 1993; Eggleston and Armstrong 1995; Feldman *et al.* 1997; Gotceitas *et al.* 1997) are also known to play an important role in the recruitment and survival of commercially important species. Unvegetated habitats, although receiving less attention from a conservation and management perspective (Hoss and Thayer 1993), also support diverse assemblages of finfish and decapod crustaceans

(Lasiak 1986; Brown and McLachlan 1990; Kailola *et al.* 1993; Morrison *et al.* 2002).

The characteristics of vegetated habitats that are thought to contribute to their value in supporting and maintaining fisheries stocks include the provision of enhanced food supply (often associated with large levels of primary production), enhanced survival due to the provision of refuges from predation and/or enhanced food supply, and reduced physical harshness and less turbulence than in other habitats. These issues have all been well reviewed elsewhere (e.g. Orth *et al.* 1984; Bell and Pollard 1989; Heck and Crowder 1991; Butler and Jernakoff 1999; Jackson *et al.* 2001) and will not be examined in detail here. Our focus is to draw attention to the need for a shift in focus in estuarine fisheries research from an approach that concentrates on the fauna of individual habitat types and makes comparisons between single habitats to one that considers the habitat as part of a mosaic of interconnected patches within a broader landscape (or seascape) made up of many different types of habitat. At present, there is almost no information about the importance of the particular arrangement of the different patches of habitat within land/seascapes on the abundance and diversity of finfish and crustacean communities.

We review the reasons for such a paradigmatic shift and propose an approach that takes into account the potential interactions that occur between different patches of habitat and their use by biota. In reviewing the extensive literature that has examined issues of the relationships between fisheries and estuarine habitats, we focus mostly on those studies that provide a mechanistic understanding of these linkages, rather than those that are primarily descriptive. It is these mechanistic studies that provide key insights into the reasons why finfish and decapod crustaceans use key estuarine habitats and therefore how they might be affected by changes in the spatial arrangement of the patches within a mosaic. These studies also provide a basis for determining the variables that might be considered as measures of differing levels of habitat quality for different mosaics.

MOVEMENT AND MIGRATION AMONG DIFFERENT PATCHES OF HABITAT

Many of the species using estuarine habitats are highly mobile and move readily between multiple habitat types regularly over a tidal cycle or during the course of their life cycle; however, surprisingly few studies have attempted to quantify the specific patterns of movements among the different patches (Beck *et al.* 2001; Morrison *et al.* 2002). Access to intertidal estuarine habitats, such as mangroves, saltmarsh and seagrass, by nekton is a function of the geomorphological and tidal characteristics at each site (Kneib 1997b) and only occurs during a portion of each tidal cycle: many species move into intertidal areas during the flood tide, but retreat to the shallow subtidal during the ebb flow (Rozas and Odum 1987; Hettler 1989; Kneib and Wagner 1994; Lin and Shao 1999; Thomas and Connolly 2001; but see Kneib 1977a). For example, juvenile prawns (*Penaeus merguiensis*) move into mangrove forests on high tide, but use the adjacent banks downshore during the low-tide period (Robertson 1988; Vance *et al.* 1996, 2002). Over longer time periods, some species are found in different parts of an estuary at different ontogenetic stages (e.g. Chubb *et al.* 1981; Middleton *et al.* 1992; Worthington *et al*. 1992; Gillanders 1997), potentially exposing the animals to a variety of types of mosaics during their lifetime if the distribution of habitat types varies along estuarine gradients (e.g. Hutchings and Saenger 1987).

Movement between different habitat types on a daily basis, or during the course of its life cycle, provides an opportunity for an animal to use different resources, such as food or shelter, found in different parts of the mosaic (e.g. Weisberg *et al.* 1981; Minello and Zimmerman 1983; Boesch and Turner 1984; Hansson *et al.* 1995), but it also potentially exposes it to different predators and other threats (Saunders *et al*. 1991). It is likely that the value of an intertidal habitat to a species will be at least partially a function of the nature of the subtidal habitat into which it must retreat during low tide. A mosaic comprising an intertidal area adjacent to a subtidal habitat that provides a high-quality refuge (e.g. Rozas and Odum 1987; Sogard and Able 1991; Everett and Ruiz 1993) may be of greater overall value than a mosaic where animals leaving the intertidal with the falling tide are forced to enter an area that offers no protection from predators, such as an unvegetated mudflat. In an elegant study, Irlandi and Crawford (1997) showed that the common pinfish, *Lagodon rhomboides*, was found in greater numbers and grew faster in intertidal saltmarsh adjacent to subtidal seagrass than in saltmarsh adjacent to unvegetated mudflat. The value of the saltmarsh habitat was therefore enhanced by the location of the subtidal high-quality seagrass. Micheli and Peterson (1999) found that the proximity of saltmarsh and oyster reefs affected the survival of benthic clams on the reefs; survival of benthic clams was lower on reefs closer to saltmarsh because of the greater abundance of the predatory blue crabs (*Callinectes sapidus*) that are found in saltmarsh habitats. In both cases, the survival of prey organisms within a mosaic was affected by the spatial arrangement of the patches of habitat.

The generality of such responses needs to be investigated, given the mobility of many groups using estuarine habitats and the potential for them to interact with a broad range of habitat types varying greatly in their relative quality and value.

USE OF DIFFERENT HABITATS BY FINFISH AND DECAPOD CRUSTACEANS

A major focus of past research has been on comparisons of different types of estuarine habitats in terms of their relative importance to finfish and decapod crustaceans. These studies generally fall into two broad categories: contrasts between vegetated and unvegetated habitats (e.g. mangroves *v.* mudflats) and contrasts between different types of vegetated habitats (e.g. seagrass *v.* mangroves, or seagrass beds of different species).

Numerous descriptive and experimental studies have demonstrated that vegetated habitats support a greater diversity and abundance of nekton (fish and decapod crustaceans – *sensu* Kneib 1997b), and this has been the basis for the focus on protection and conservation of such areas within estuaries. This general pattern is usually explained by reference to the importance of structural complexity in mediating predator–prey interactions. As the structural complexity of the habitat increases, the intensity and success of predation generally declines (e.g. saltmarsh – Vince *et al.* 1976; Minello and Zimmerman 1983; seagrass – Coen *et al.* 1981; Heck and Thoman 1981; Stoner 1982; Summerson and Peterson 1984; Leber 1985; Kenyon *et al.* 1995). Some studies have not supported this general paradigm though, suggesting that more detailed understanding of the specific links between the habitats and the biota is needed. For example, Thomas and Connolly (2001) found no clear difference in the assemblage of fish using patches of saltmarsh and adjacent unvegetated sediments, and Edgar and Shaw (1995) found that for many commercial species, seagrass beds were not more important nursery areas than nearby unvegetated areas. Importantly, there are some clear indications that the use of one habitat is affected by the proximity to another. Ferrell and Bell (1991) and Jenkins and Hamer (2001) found that the number of fish that occurred in unvegetated areas was tightly linked to the proximity of those sites to nearby patches of seagrass, suggesting that factors affecting one part of a mosaic would also influence the dynamics in the other patches (see also Heck and Thoman 1984; Shaw and Jenkins 1992).

Fewer studies have specifically contrasted different types of vegetated habitat (reviewed by Jackson *et al.* 2001). Robertson and Duke (1987) and Laegdsgaard and Johnson (1995) compared the abundance of nekton in mangrove and seagrass habitats, and in general found that the mangroves supported greater densities of fish than seagrass. Similarly, Sogard and Able (1991) compared the abundance of nekton in seagrass and saltmarsh creeks and found similar results. Irrespective of whether such patterns are true across a broader range of geographic areas and times, an important unaddressed question relates to how use of intertidal mangrove (or saltmarsh) areas is affected by the nature of the adjacent habitats into which nekton must migrate at low tide (Laegdsgaard and Johnson 1995). Areas of high-quality mangrove, available for only a small proportion of any tidal cycle, may vary in their value as a nursery (*sensu* Beck *et al.* 2001) depending on the nature of the subtidal habitats in which the animals spend the majority of their time (Irlandi and Crawford 1997; Jenkins *et al.* 1997).

Within the broad category of studies contrasting different types of vegetated habitats, important information on the features that determine the relative value of an estuarine habitat has also been obtained through comparisons of seagrass beds composed of species with different morphological characteristics. Factors such as leaf length (canopy height above the substratum), blade width and blade density have all been shown to influence the composition of the nekton community that uses seagrass beds (e.g. Stoner and Lewis 1985; Bell and Westoby 1986a, 1986b, 1986c; Middleton *et al.* 1984; Worthington *et al.* 1992; Kenyon *et al.* 1995; Gotceitas *et al.* 1997; Loneragan *et al.* 1998, 2001). Features providing structural complexity within mangroves, such as the density of pneumatophores and prop roots, have also been linked with differences in community composition of nekton (e.g. Thayer *et al.* 1987; Blaber *et al.* 1995; Laegdsgaard and Johnson 2001). Again, these patterns have mostly been explained in relation to the role of structural complexity and the effects on predator–prey interactions (see references above), although there is some debate as to whether the role of predation is a proximal or indirect control on abundance (see Bell and Westoby 1986a). Given that different seagrass beds consist of a mosaic of patches of different sizes and shapes, interspersed with unvegetated corridors (Irlandi 1994, 1996), variation in these structural characteristics of the seagrass would suggest that the overall quality of a habitat mosaic that included mangroves and seagrass could vary considerably at different spatial scales. Experimental studies, manipulating levels of structural complexity with associated effects on other measures of habitat quality, have confirmed that these factors strongly influence the value of a patch for supporting nekton communities but these studies have all focussed on within-habitat type comparisons. No studies in marine or estuarine environments have examined the interactions between habitats or how the composition and spatial arrangement of different types of patch affect the way mosaics are used by organisms. The evidence suggests strongly that the presence of different types of patch in an estuarine mosaic will change the overall value of that mosaic because of the different resources that are provided.

SPATIAL ARRANGEMENT OF PATCHES IN A MOSAIC

The size and spatial arrangement of a patch of habitat may also influence its value to the animals that are using it. Irlandi *et al.* (1995) showed that survival of juvenile bay scallops (*Argopecten irradians*) declined in beds of seagrass that were very patchy (22% vegetation) compared with patchy (70% cover) or continuous (97%) cover, and that these effects were not due to variation in characteristics of the vegetation such as density, blade length or biomass. They attributed these results to greater access of predators to prey in very patchy areas because of increased edge-to-interior ratios compared with the more continuous beds. The unvegetated areas within the seagrass bed essentially act as corridors for movement of predators, enhancing their effectiveness at locating and acquiring prey (see also Micheli and Peterson 1999). Similarly, growth and survival of another commercial bivalve, *Mercenaria mercenaria*, was also significantly affected by the size of seagrass patch (Irlandi 1996, 1997). Bowden *et al.* (2001) found that patch size significantly affected the composition of infaunal assemblages in seagrass, although spatial variation at the regional level was relatively more important in determining the differences among seagrass beds. These novel approaches need to be applied in studies on more mobile fauna, such as the nekton that use estuarine mosaics.

A NEW APPROACH – EVALUATING HABITAT MOSAICS FOR FISHERIES AND DIVERSITY

Stage 1: Large-scale GIS mapping of mosaics

The consequences to fisheries from the large-scale loss of and damage to estuarine habitats (e.g. Naylor *et al.* 2000; Jackson *et al.* 2001) is now well recognised and has focused attention on the need for the establishment of marine protected areas and reserves (Margules and Nichols 1988; McNeill 1994; Kelleher *et al.* 1995). In many cases, specific types of habitats (e.g. mangroves) are protected from development and/or loss (Valiela *et al.* 2001) but this does not take into account deterioration of adjacent patches of habitat that may not receive the same level of protection. The ecological significance of the spatial arrangement of the different patches within a mosaic and the interactions across boundaries between patches has been well explored in terrestrial environments (e.g. Wiens *et al.* 1985; Hansson and de Castri 1992) but is only now being investigated for marine and estuarine systems (Irlandi 1994, 1996; Robbins and Bell 1994; Irlandi and Crawford 1997; Brooks and Bell 2001).

The basis of our approach here is to incorporate information on the spatial arrangement, structure and condition of the patches of different habitat within a mosaic, rather than focusing just on individual types of habitats. This allows us to address the issue of whether deteriorating quality of any particular patch of habitat affects the value to fisheries of adjacent elements within the mosaic. Using this approach, we are able to ask whether the loss of or damage to a subtidal seagrass bed may have consequences for the value of a patch of intertidal mangrove, even when the latter is protected within a reserve and/or is relatively undisturbed. Answers to such questions will allow a more focused approach to deciding which combinations of habitat types are best protected within a region, given that the total area to be included within a reserve system will be limited.

Our approach is to measure and quantify the spatial extent and arrangement of the different habitats within an estuarine area, drawing on techniques and methods developed for terrestrial landscape ecology (e.g. Forman and Godron 1986; Turner 1989; Turner and Gardner 1991). Spatial-pattern metrics are used to describe the characteristics of the patches of different habitat based on their extent and configuration within the mosaics. The metrics being used include area metrics (e.g. total area of habitat patch), edge metrics (e.g. patch perimeter) and connectivity metrics (e.g. nearest neighbour, proximity and fragmentation). Data on wetland distribution in south-east Queensland are being obtained from a variety of sources. Detailed methodology on the analysis and interpretation of the data can found in Manson *et al.* (2003).

An important component of the analysis of the spatial mapping information is the change-detection analysis on the distribution and arrangement of different mosaics through time. These analyses provide us with a measure of how much the distribution of a particular type of mosaic has changed through time and, more importantly, which mosaics have been interchanged in any area. This then provides a basis for considering the implications of any differences in the relative value to fisheries and biodiversity of the different mosaics and also provides a means of evaluating the effects of large-scale habitat fragmentation and loss within estuarine systems.

Stage 2: Measures of structural complexity for habitat mosaics

Given the demonstrated importance of characteristics of habitats that provide structural complexity (see above), the differentiation and categorisation of different mosaics is based on the quantitative analysis of these measures for each of the patches within the mosaic. Detailed mapping and measurement of the physical characteristics of each of the patches or elements within each mosaic (Table 1) will be done to define whether each element could be considered as a high-, medium- or low-quality patch. Multivariate analysis of these physical data (e.g. nMDS – Clarke 1993 and Canonical Correspondence Analysis – ter Braak 1987) is used to differentiate between patches of differing quality. The core hypotheses being examined are about whether the use of these different patches is affected by the nature of the adjacent elements within the mosaic. Thus, the following three mosaics might be chosen for comparison: high quality for both mangroves and seagrass (i.e. multiple sites of high-quality mangroves with dense seagrass lower down the shore), high-quality mangroves and low-quality seagrass (multiple sites of high-quality mangroves with sparse seagrass lower down the shore) and low-quality mangroves and high-quality seagrass (multiple sites with low-quality mangroves and dense seagrass lower on the shore). A range of potentially suitable sites will be selected from the GIS database, followed by detailed ground-truthing of the physical characteristics of the patches within the mosaics.

Stage 3: Sampling of fish and decapods

Continuing the above example, sampling the nekton in the mangrove component of the mosaic would examine whether use of this habitat type varies as a function of the nature of the downshore habitat (high- or low-quality seagrass). Thus, multiple sites containing mangroves of similar quality would be sampled and compared on the basis of the nature of the adjacent habitats. It is important to note that this approach avoids the problem of trying to make direct comparisons of abundance and community composition between different habitat types (e.g. mangroves *v.* seagrass) when the methods required to sample within those habitats usually vary (e.g. Robertson and Duke 1987; Laegdsgaard and Johnson 1995). The specific comparisons are all, initially at least, based on an examination of whether use varies within a particular patch-type – each patch of mangroves is sampled using the same methods and experimental design. Conversely, using the same data set, we are also able to examine whether use of the seagrass habitat varies as a function of the quality of the upshore mangroves. Choosing a range of mosaics that include patches of habitat along a gradient of relative quality enhances our capacity to determine whether the composition and spatial arrangement of the elements affects use of the mosaic by the nekton.

This approach allows the specific methods and experimental design for sampling the nekton to be optimised for each of the habitat types within the mosaic and, where necessary, multiple methods to be employed in order to obtain the best estimates of community composition using the elements of the mosaic. A combination of methods has been chosen to sample the different elements of the mosaic, including: fyke nets (e.g. Lin and Shao 1999), stake nets (e.g. Vance *et al.* 1996, 2002) and pop nets (e.g. Connolly 1994; Thomas and Connolly 2001) for within the mangroves; two different sizes of seine nets (e.g. Hindell *et al.* 2000) and pop nets for intertidal unvegetated and seagrass areas; and seine nets and a small otter trawl (e.g. Peterson and Skilleter 1994) for subtidal habitats.

The design for the sampling program incorporates multiple spatial scales including comparisons between two regions in Moreton Bay (western – heavily urbanised, eastern – relatively pristine), between two separate coastal embayments in south-east Queensland (Moreton Bay and Hervey Bay) and different proximity to the shoreline (mosaics along the edge of the estuary *v.* those existing as isolated banks and islands within the embayments). Sampling will be done in spring/summer and winter of two successive years to test whether the different mosaics are used in the same way by different species and different ontogenetic stages of the same species. During different times of the year, depending on when particular species are recruiting, the nekton communities in some mosaics are likely to be dominated by new recruits, whereas at other times of the year the fauna will be dominated by larger individuals, possibly from several different year-classes, or different species (Connolly *et al.* 1999).

FUNCTIONAL VALUE OF DIFFERENT MOSAICS

In response to the challenges posed by Beck *et al.* (2001), we recognise that measures of abundance alone are not a good indication of the relative value of an estuarine habitat, or of patches of habitat within a mosaic, and we have therefore explicitly included in this study measures of the ecological function (O'Neill *et al.* 1992;

Fairweather 1999) provided by different mosaics. Trophic structure and predator–prey interactions represent important attributes of the functional aspects provided by habitats, and these may vary in response to changes in the spatial arrangement and structural complexity of the mosaics and the fauna that are using them. The growth and survival of abundant species in different mosaics also provide an indication of overall habitat quality and function, and hence will be measured to provide other indices of the ecological functioning of a mosaic.

CONCLUSIONS

Previous detailed studies have provided much information on the value of single aquatic habitats and their use by a variety of nekton species. However, in general these studies have not taken into account the location of the habitat and the spatial arrangement of adjacent habitats in the area. The approach we shall be taking involves the following: broad-scale mapping of estuarine and shallow marine coastal habitats; identifying the different broad categories of mosaics within the region and measuring the characteristics of the mosaic; and sampling the fauna within different mosaics to investigate whether the abundance and distribution of fauna varies between mosaics and investigate whether ecological function varies between mosaics. This approach builds on previous studies with a "single" habitat focus and hence provides more comprehensive information to managers for improving the design of protected areas in estuarine and shallow coastal waters.

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ROLE OF HABITAT MAPPING IN MARINE PROTECTED AREA PLANNING – A CASE STUDY IN THE BRUNY BIOREGION, TASMANIA

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Abstract

Mapping of coastal seabed habitats throughout the Bruny bioregion has proved to be an essential tool needed to determine Marine Protected Area (MPA) options in a ecologically complex bioregion, to select appropriate boundaries and to ensure that protected areas are comprehensive, representative and adequate. This has been achieved through the accurate definition of habitat boundaries and description of the dominant macroalgae and seagrass assemblages. Suggestions for potential MPA locations can be objectively derived from the mapping results in a process aimed at maximising the habitat diversity for each location. However, substantial additional biological information is also required if the protection of small-scale unique features or species distribution is to be an important component of the MPA planning process. Much of this information already exists but requires analysis within an MPA framework and incorporation into a comprehensive Geographic Information System. A habitat management strategy, together with MPAs and appropriate fisheries and land use management, would provide the flexibility required to conserve coastal and estuarine biodiversity within the Bruny bioregion, Tasmania, in the long term.

Keywords: habitat mapping, seagrass, rocky reef, biodiversity

INTRODUCTION

Mapping of marine and estuarine seabed habitats is increasingly being recognised as an important component of the overall research required to identify the nature, scale and distribution of Australia's marine ecosystems and biodiversity for conservation planning and multiple-use ecosystem management (e.g. Fern and Hough 1999; Barrett *et al.* 2001; Breen and Avery 2001). Although much of this research has been concentrated on the coastal zone, there has been a strong focus recently on mapping seabed habitat distributions and examining the associations between habitats and fish community structure and composition on the continental shelf (Bax and Williams 2001; Kloser *et al.* 2001; Williams and Bax 2001). Such research is also becoming more achievable through recent advancement and application of technologies such as multi-frequency acoustics, differential Global Position System (GPS), powerful Geographic Information System (GIS) software, real-time digital underwater videorecording and other remote-sensing techniques.

One of the tools progressed by Australia and its States and Northern Territory since the 1990s for protecting biodiversity and ecosystem processes is the establishment of a National Representative System of Marine Protected Areas (NRSMPA) (ANZECC 1998a). In order to provide a national framework for the NRSMPA a hierarchical classification of the marine environment has been developed based primarily on the distribution of biota and other physical attributes (ANZECC 1998b). The Interim Marine and Coastal Regionalisation (or IMCRA) has defined Australia's inshore ecosystems down to the 'bioregion' or mesoscale level (1000 km^2) (ANZECC 1998b).

The island State of Tasmania has jurisdiction over State Coastal Waters and Internal Waters that have a combined area of around 23,000 km^2, and, at around 5000 km, it has the longest coastline per unit of land of any State of Australia. Based on the analysis of the distribution of Tasmanian marine biota, including rocky-reef biota, beach-washed shells, and beach-seine collections of coastal and estuarine fishes, nine bioregions have been identified (Edgar *et al.* 1995). The detailed regionalisation of Tasmanian waters is primarily due to the complex oceanography of this region, interacting with substantial gradients in exposure to waves and oceanic swells. Tasmania has declared five marine reserves that represent around 3.4% of State waters.

In Tasmania, a Marine Protected Area (MPA) strategy has been developed to structure the process of further MPA development (Marine and Marine Industries Council 2000). This process recognised that although the mesoscale classification is sufficient to assess the comprehensiveness of a NRSMPA at the large marine ecosystem scale, it is not fine-scale enough to assess the adequacy or representativeness of the NRSMPA, or to assist with the identification of specific MPAs. The lack of information on the distribution of marine habitats in Tasmania also precluded a systematic and informed process of MPA development. As a result, the mapping of seabed habitats was identified as essential for developing a comprehensive, representative and adequate system of MPAs in Tasmania and this led to the detailed mapping of coastal habitats (depth <40 m) throughout the Bruny bioregion (Barrett *et al.* 2001).

Much of the south-east corner of Tasmania is contained within the Bruny bioregion (Fig. 1), which is characterised by a high degree of endemism of marine species restricted to this region. The sheltered embayments are the southernmost refuges available in Australia for a number of cold-adapted species (Edgar *et al*.1995; ANZECC 1998b). A further distinctive feature is the presence of large "forests" of the giant string kelp *Macrocystis pyrifera*, a species restricted to the cool temperate waters of southern Tasmania (Edgar 2001).

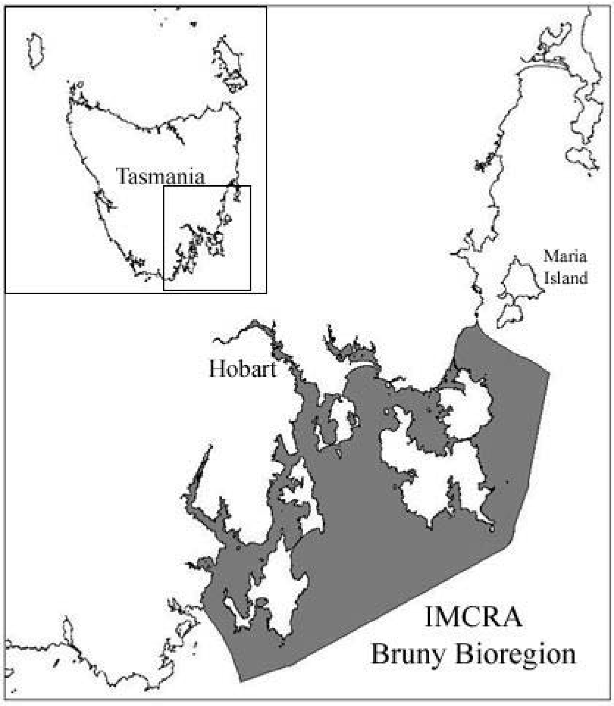

Fig. 1. Distribution of the Bruny bioregion, south-eastern Tasmania, Australia.

There was limited information available on the spatial distribution of marine habitats within the Bruny bioregion. Nevertheless, a number of studies have examined site-specific diversity and patterns in community structure throughout Tasmanian coastal waters,. including fishes, macroalgae and large invertebrates on rocky reef (Edgar *et al.* 1995), beach-seined inshore and estuarine fishes and invertebrates (Last 1983; Edgar *et al.* 1999), fishes associated with coastal unvegetated areas and seagrasses (Jordan *et al.* 1998), and invertebrates (Moverley and Jordan 1996). In addition, the distribution of communities with respect to major physical characteristics is relatively well known for Tasmanian waters (e.g. Edgar 1984; Edgar *et al*.1995; Last 1989; Edgar 2001

All these studies have provided some capacity to predict communities within broad habitat types such as rocky reef, unvegetated sediments and seagrass. A good representation of habitats should lead to an equally good representation of marine species diversity (Ward *et al*.1999). Additionally, for the planning of individual MPAs, it is important that locations are chosen that can incorporate a range of representative habitats, and are of sufficient size with suitable boundaries to minimise the loss of protected species to adjacent areas (Barrett 1995; Kramer and Chapman 1999).

Previous studies examining habitat distributions in Tasmania have assessed specific areas for marine farm development, sought potential MPA locations (Barrett and Wilcox 2001), recorded the distribution of selected seagrass beds (Rees 1993), or been at a very coarse scale (Edyvane *et al.* 2000). In order to assist with the identification of individual MPAs and assess the representativeness or adequacy of any proposals, the distribution of the principle marine habitats to the 40 m depth contour throughout the Bruny bioregion were comprehensively mapped (Barrett *et al.* 2001). This paper aims to examine the role of seabed habitat mapping in the planning and development of MPAs by broadly describing the distribution of habitats throughout the Bruny bioregion, detailing the factors influencing community structure at a range of spatial scales and describing how this information can be incorporated into an MPA framework.

METHODS

The production of maps involved the digitising of habitat boundaries in shallow water from available aerial photographs, and extensive field surveys from the low-tide mark to the 40 m depth contour from small vessels equipped with colour echosounders, differential GPS, digital videorecorders and sediment grabs. Habitat boundaries and attributes in water generally >10 m depth were determined with an echosounder and video surveys. Information included depth, substratum type and differentially corrected Global Positioning System (DGPS) data. Information on the dominant seagrass, macroalgae and invertebrates was continuously logged into a laptop computer using a purpose-built software package, *Seabed Mapper 2.4*.

Habitats were broadly categorised into three main groups: rocky reefs, and unconsolidated substrata that were unvegetated or vegetated (i.e. seagrass and *Caulerpa* sp.). Each of these broad categories was broken down into sub-categories based on relief for reefs, dominant sediment type for unvegetated habitats, and blade density and patchiness for seagrasses (Table 1).

As detailed studies of all biotic communities are particularly difficult and time consuming in the marine environment, and also require very fine-scale mapping in areas with any depth transitions, this study has used "indicator" physical characteristics for the identification of marine habitats. In order to identify the dominant marine communities present, regular video surveying was conducted. The main physical characters used to identify key habitats were depth, substratum type and exposure to wave action. Biotic factors were included for soft-sediment areas where the presence of seagrass or *Caulerpa* beds on the sediment surface provided a distinctly identifiable habitat.

Table 1. Substratum and habitat categories used in the mapping of habitats in the Bruny bioregion.

Rocky Reef
High relief reef Depth variation greater than 4–10 m over short distances.
Medium relief reef Depth variation greater than 1–4 m over short distances.
Low relief reef Hard bottom type with <1 m change in the relief.
Patchy reef Reef elements, including boulders and rocks, intermittently outcropping from unconsolidated sediments, principally sand.
Unvegetated unconsolidated substratum
Sand Sand represented the coarser end of a scale of sediments from silt to sand and was generally characterised by a distinct second echo on the sounder trace.
Silty sand Silty sand broadly incorporated any sediment with a significant proportion of coarse "sand" particles and fine "silt" particles and characterised by a less-distinct second echo on the sounder trace.
Silt Represented the finest unconsolidated substratum and characterised on the sounder by a lack of a second echo and often little scatter in the trace tail.
Hard sand Includes large grain size, shell matter (either whole shells or shell grit) or biological material. Refers to unconsolidated substrata containing elements that confound the sounder output causing the signal to appear either harder or rougher than would be expected from that substratum.
Vegetated unconsolidated substrate
Seagrass Represents seagrass with little patchiness and relatively high percentage cover. The dominant seagrass was *Heterozostera tasmanica* with *Halophila australis* often occurring in the same beds.
Patchy seagrass Represents areas where patch size varied from <1 m up to 20 m in linear extent. The patches generally consisted of dense seagrass.
Sparse seagrass Category usually applied to the density of the shoots of the seagrass (primarily *Heterozostera tasmanica*), where the substratum beneath the seagrass was easily visible, often consisting of more than 50% of the field of view in the camera frame.
Caulerpa Category applies to distinct beds of *Caulerpa* (principally *C. trifaria)*, a green algal species that can have extensive rhizoidal networks in the sediment. This species can form extensive beds similar to those formed by seagrass; these are often found on the seaward extent of seagrass beds.

Rock type has been found to have a minor influence on community structure along Australia's Victorian coast compared with exposure, depth, substratum and coastal region (Edmunds *et al.* 1998). For this reason, rock type (geology) was not included in the habitat characterisation of the Bruny bioregion. However, reef structure and complexity influences the availability of refuges and therefore species diversity, and in the Bruny bioregion this component was mapped as reef profile. Some rock types have characteristic weathering patterns, and sedimentary rocks form marine cave systems more readily than other rock types; however, it is important not to generalise more broadly. All rock types produce a range of structural complexity, and it is this range that determines the structure of the biotic community, not the type of rock itself.

Detailed bathymetry based on tidally corrected depth data was generated in order to analyse habitat variations by depth. Coastal exposure was estimated for the Bruny bioregion by use of a wave-exposure index based on aspect, extent of fetch and possible exposure to oceanic swells. Maps produced at 1:100,000 and 1:25,000 (see Figs 2 and 3) using the GIS software (*ArcView 3.2)* allowed detailed analysis of habitat distribution by depth and exposure. Suggestions for potential MPA locations were objectively derived from the mapping results in a process aimed at maximising the habitat diversity for each location. More information on aerial photograph geo-referencing and analysis, field data collection, depth contouring, wave-exposure index calculation and mapping error estimation is available in Barrett *et al.* (2001).

RESULTS AND DISCUSSION

Mapping of coastal seabed habitats throughout the Bruny bioregion provided detailed information on the spatial distribution of habitats at a fine scale. Given the lack of information on the distribution of habitats and overall diversity of the bioregion, such mapping has proven essential to the process of identifying potential MPAs that are comprehensive, adequate and representative. This has been achieved through the accurate definition of habitat boundaries and description of the dominant macroalgal and seagrass communities.

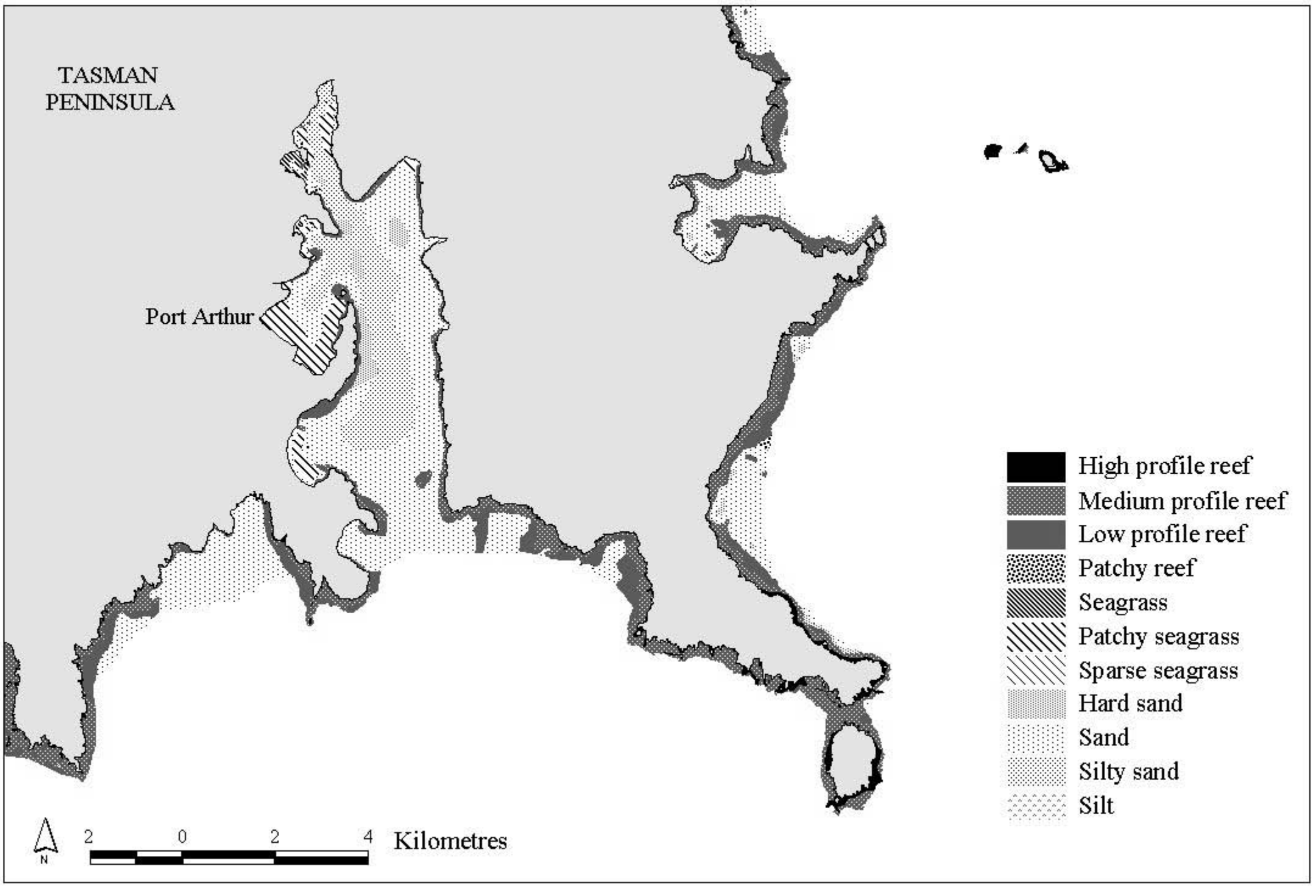

Fig. 2. 1:100,000 scale habitat map of the Tasman Peninsula region of the Bruny bioregion.

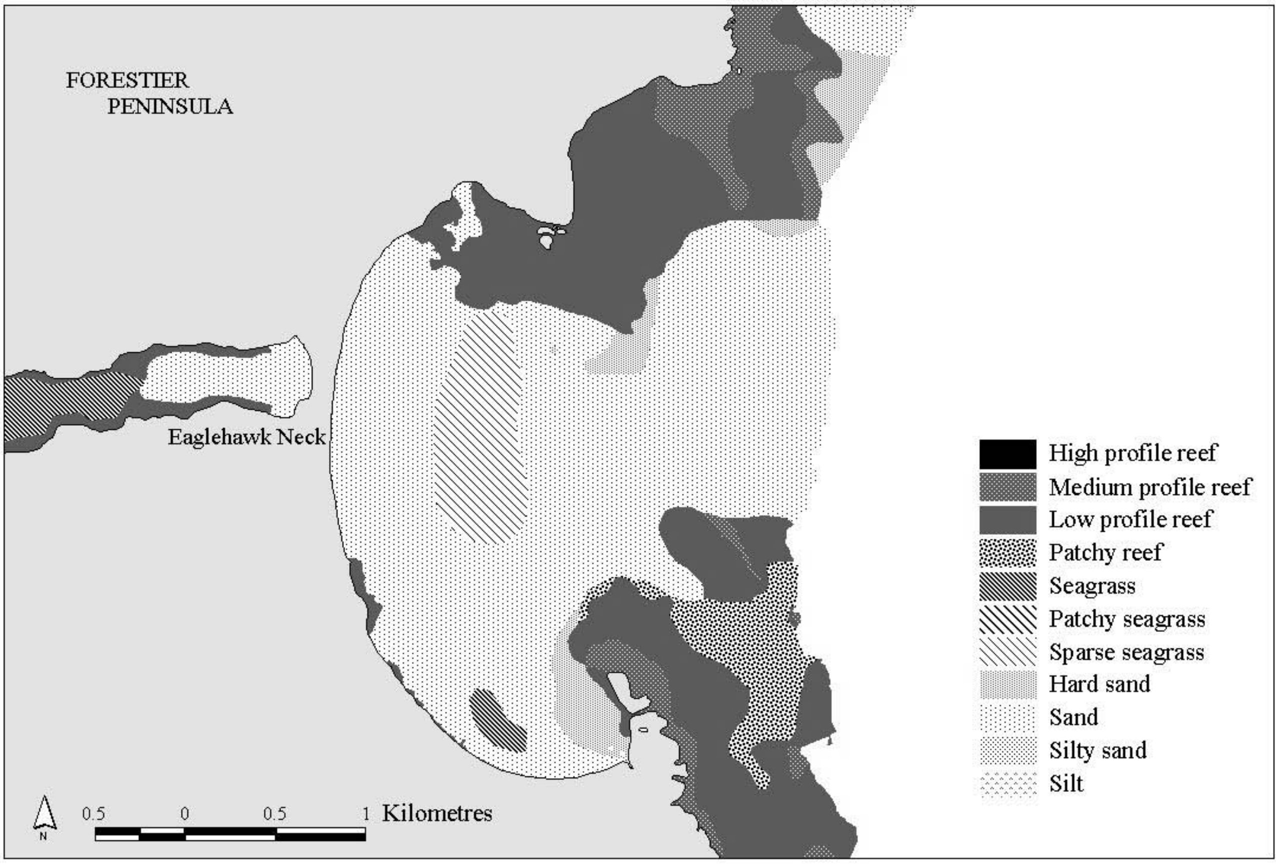

Fig. 3. 1:25,000 scale habitat map of the Forestier Peninsula region of the Bruny bioregion.

The bioregion contains a particularly complex and convoluted coastline with numerous islands, peninsulas and estuaries. On the outer coast, south-facing shores are exposed to persistent and often extremely large swells originating from southern ocean gales, while east-facing shores are exposed to less frequent but occasionally large swells derived from easterly weather patterns in the Tasman Sea. Despite this exposure, the convoluted nature of this coastline also provides a substantial component of sheltered waters and associated habitats. Much of the southern and eastern coastline comprises a steep coastline which extends underwater, with the 40 m depth contour usually being less than 1 km from the shoreline. The more sheltered coasts of the embayments and channels have more gradual slopes, and the waters are relatively shallow, except for areas where ancient drowned river valleys result in depths in excess of 40 m.

A significant output of the field mapping is the generation of the detailed bathymetry that is essential in the analysis of habitat/depth relationships. The bathymetry of much of the Tasmanian coast is still poorly defined and is often not suitable for analysis at a fine scale (i.e. <1:10,000). This is particularly the case with depth data used in the development of 3-D hydrodynamic models that may be used to model such things as egg and larval advection patterns and point-source pollution inputs in relation to MPA locations.

The biota of the Bruny bioregion is influenced at a local scale by a wide range of physical characteristics including exposure, aspect, depth, rock type and complexity, sediment type, and regional oceanography that determines nutrient levels, turbidity, current speeds, salinity, temperature and seasonal and interannual variation. However, there are several features that broadly characterise the bioregion, including a significant component of sheltered waterways and embayments, a substantial influence of nutrient-rich sub-Antarctic water and a narrow continental shelf (Edgar *et al.* 1995; ANZECC 1998a).

Representative features of this system therefore include a range of sheltered habitats and cool-temperate species assemblages. At an intermediate level, freshwater discharge from the region's two large river systems, the Derwent and Huon, substantially structures the biota such that assemblages can be characterised by the influence of one, both, or neither of these. Clear oceanic water influences much of the outer coast, resulting in macroalgae extending to at least 30 m depth. While the habitat categories used can adequately describe seabed distribution at one hierarchical level, local-scale differences can influence patterns of diversity and community structure. The following sections describe the distribution of the primary habitat categories of rocky reef, unvegetated unconsolidated substratum and vegetated unconsolidated substratum.

Rock reef

Rocky reefs represent an important and diverse habitat within the Bruny bioregion, which supports major fisheries for abalone, rock lobster and reef-associated scalefish species. The habitat occurs primarily adjacent to rocky headlands, although continuous reef is present along much of the coast of the Tasman Peninsula (Fig. 2), where the shoreline is either steep or composed of cliffs, and underwater gradients are usually steep until depths of greater than 10 m. At most exposures and depths, reefs make up around 8% of the seabed habitats as these are mostly restricted to a coastal fringe (Table 2).

Table 2. Areal estimates (hectares) of rocky reef, unvegetated unconsolidated substratum and vegetated unconsolidated substrate within the Bruny bioregion from 0 to 40 m depth.

Habitat type	Area (ha)	%
Rocky reef	12,665	8.5
Unvegetated unconsolidated substratum	127,802	85.2
Vegetated unconsolidated substrata		
- seagrass	6,472	4.3
- *Caulerpa* sp.	3,022	2.0
Total	**149,961**	

In general, substantially more reef occurs on exposed coast than on sheltered coast at all depth ranges, reflecting the erosional and depositional nature of the differing exposures. At the more exposed end of the scale, the proportion of reef habitat increases with depth, with the majority of reef being found at depths greater than 10 m. In the most sheltered locations reefs are primarily restricted to depths of less than 10 m.

Broad-scale mapping of coastal habitats generally requires restriction of the examination of biological communities to the major cover-forming species of macroalgae and large invertebrates such as sponges and seawhips. The majority of species are widespread throughout the bioregion, but their local-scale distribution and abundance is primarily determined by depth and exposure. For example, *Durvillaea* is found on the most wave-exposed rocky coast, followed by *Phyllospora*, *Ecklonia*, red algae and then sponges with increasing depth. With decreasing wave action there is a corresponding reduction in the depth to which these communities occur. *Durvillaea* and *Phyllospora* are replaced by brown algae of the order Fucales (Fucoides), including *Xiphophora*, *Acrocarpia*, *Cystophora* and *Caulocystis* species. Giant string kelp *M. pyrifera* often forms extensive beds in areas of moderate exposure, but can also be relatively common seasonally at almost all exposures. Beneath the canopy of these kelps there is often a large number of foliose, filamentous and encrusting red algal species. The distribution of these is generally patchy at a small spatial scale, a pattern common in understorey species (Steinberg and Kendrick 1999).

Rocky reefs dominated by sponge communities are most abundant in areas of high current flow. In sheltered waters these communities can occur at depths less than 10 m if fast currents and high turbidity have restricted the abundance of macroalgae. For example, in the D'Entrecasteaux Channel, sponges, seawhips and gorgonians can be found in relatively shallow depths, sometimes on beds of shells. More generally, similar communities are found in waters greater than 33 m, the lower limit at which brown algae dominate reefs in the region.

Detailed video surveys also revealed that in some areas, despite similarities in depth and exposure, differences in the dominant macroalgal species do occur. For example, *Phyllospora commosa*, a very common alga in temperate Australian waters, was absent from substantial sections of the coast, with *Lessonia corrugata*, *Carpoglossum confluens*, *Ecklonia radiata* and *Pyura* spp. (ascidians) occupying the niche that this normally occurs.

The accurate mapping of reef habitat boundaries also provides the ability to define MPA boundaries that maximise the benefits to reef-associated macrofaunal species. For example, the Ninepin Point Marine Reserve currently includes habitats in this area ranging from low exposure to sheltered reef. However, because of the small size of this reserve, many resident reef fishes are currently lost across the boundary to adjacent fished areas, limiting the effectiveness of the area to fulfil the conservation role intended of a representative MPA (Edgar and Barrett 2000).

Unvegetated unconsolidated substrata

Unvegetated unconsolidated substrata are the dominant habitat type within the <40 m depth range of the Bruny bioregion, representing around 85% of all seabed habitats (Table 2). The substratum type is strongly related to exposure and depth, with shallow exposed locations being dominated by sand, and the most sheltered locations such as the deep parts of embayments being dominated by silt (Fig. 2). The general trend of increasing silt concentrations with depth indicates that it is a lower-energy environment where fine suspended sediment is being deposited. A notable feature of the bioregion is also the area of hard sand in the D'Entrecasteaux Channel, reflecting the extensive and high-density cover of dead scallop and the introduced New Zealand screw shell (*Maoricolpus roseus)*, particularly in areas of high current flow.

These gradients in sediment type, in combination with depth, result in considerable differences in the macrofaunal composition. For example, the dominant fish species in shallow (<10 m) unvegetated habitats in the region include atherinids (Family: Atherinidae), flounders (Family: Pleuronectidae), leatherjackets (Family: Monocanthidae), mullets (Family: Mugilidae) and eastern Australian salmon (*Arripis trutta*) (Jordan *et al.* 1998), while leatherjackets, gurnards (Family: Triglidae), skates (Family: Rajidae) and stingarees (Family: Urolophidae) dominate the deep (10–40 m) unvegetated habitats (Jordan 1997). Similar changes across these depths and sediment type are likely to occur also in invertebrate communities (Edgar *et al.* 1999). This suggests that analysis within an MPA framework for these habitats requires a sound understanding of macrofaunal community structure.

Another factor influencing the diversity in such habitats is the extent of anthropogenic impacts. Increased siltation in heavily cleared catchments in Tasmania has modified invertebrate species composition, resulting in local-scale differences in community composition (Edgar *et al.* 1999) and therefore the capacity for the affected area to be 'representative'. In addition to siltation, the Derwent Estuary (located within the Bruny bioregion) also has extremely high levels of heavy metal contamination (Coughanowr 1997), which is likely to significantly modify the macrofaunal composition. There has also been a reduction in the abundance of benthic invertebrate species in the lower Derwent Estuary as a result of the presence of introduced marine pests, particularly the seastars *Asterias amurensis* and *Patiriella regularis*, gastropod *Maoricolpus roseus*, chiton *Amaurochiton glaucus*, ascidian *Ascidiella aspersa* and crab *Cancer novaezelandiae* (Morrice 1995). Such areas of heavily impacted unvegetated habitats are likely to be poor candidates for MPAs.

Vegetated unconsolidated substrata

This broad category primarily refers to seagrass beds, which are distributed throughout the Bruny bioregion representing around 4% of the overall seabed habitat (Table 2). However, the vast majority of this is restricted to one large embayment (Norfolk Bay), where the majority of shoreline has beds of seagrass down to at least 8 m deep. The estimate of 4% coverage by seagrasses does not include estuaries, where extensive seagrass beds are known to exist (Rees 1993; Jordan *et al*. 2001). The dominant species is *Heterozostera tasmanica* which forms extensive subtidal beds, generally in sheltered locations (Fig. 2). *Halophila australis* is found subtidally in smaller quantities, while *Zostera muelleri* occurs in intertidal areas but is generally only present in small quantities and rarely forms distinct beds.

The factors influencing the distribution of seagrasses include depth, temperature, wave action or exposure, season and, most importantly, the availability of light. These factors generally affect seagrass species differently. In *Heterozostera tasmanica* they generally restrict growth to depths <10 m in areas of sheltered water, although some sparse beds down to around 18 m do occur in the bioregion (Fig. 3). Seasonal variations in seagrass biomass are also likely to occur, as has been shown in *H. tasmanica* beds elsewhere in temperate Australia (Bulthuis and Woelkerling 1983). All these factors act to generate the seagrass categories mapped; however, without small-scale (~1:20,000) and recent good-quality aerial photographs it is often difficult to accurately field map the boundaries of the various categories, because variations in *H. tasmanica* can occur at the fine scale (Jordan *et al*. 2002).

Broadly, however, there are differences in fish species composition between the beds of varying density and patchiness. For example, two of the three most abundant fish species, and six of thirty-two fish species overall in *H. tasmanica* beds in Norfolk Bay showed a distinct preference for beds with the highest seagrass density (Jordan *et al*. 1998). Preference for high-density beds is not the case with all species on all occasions, however (Bell and Westoby 1986). There is also evidence that, regardless of the density and size of seagrass beds, the position of the bed in an estuary is often important in determining fish abundance and diversity (Jenkins *et al.* 1997). While small-scale differences are important, bay-wide scale differences in fish communities in *H. tasmanica* beds in eastern Tasmania are also significant, with each bay having a 'unique' community (Jordan *et al.* 1998). It is clear that habitat mapping and community studies at a range of scales are important in making decisions on whether specific seagrass areas are comprehensive, representative and adequate, particular as the seagrass/sand interface is very important for foraging and in determining boundary definition.

Species of the green algal genus *Caulerpa* (principally *C. trifaria*) occurred in large beds on the deeper boundaries of seagrass beds in Norfolk Bay. Throughout the Bruny bioregion, *Caulerpa* species occupy an area almost half the size of that occupied by seagrass. This alga can form a habitat for fish and invertebrate communities similar to that of seagrass beds (Edgar 1997).

Marine Protected Area planning

A consequence of the complexity of the Bruny bioregion is that, unlike some other Tasmanian bioregions, it is difficult to define exactly what areas (in addition to the existing MPAs at Tinderbox and Ninepin Point) should be included in a MPA network to ensure that it is fully compliant with the comprehensive, adequate and representative (CAR) definitions of ANZECC (1998b).

In the absence of detailed biological studies of all assemblages at all locations and depths within the bioregion, initial MPA planning for this complex bioregion requires the assumption that categories mapped can act as surrogates for biological diversity. Biological descriptions for mapping units give a broad overview of the dominant macroalgal community and an indication of the influence of exposure and depth at that scale, but this is inadequate to identify small-scale differences. Planning at this broad habitat scale will certainly lead to omission of some species assemblages, including threatened species and those with unique features. However, given a good understanding of the processes that structure communities, and given the selection of areas of suitable size across the broad range of habitats and systems identified, MPA locations could be chosen to provide a high degree of representation (Ward *et al.* 1999).

To try to represent every possible habitat combination in a legislated MPA network would lead to an overly complex and potentially unmanageable outcome. The most logical approach will be to develop a coastal habitat management strategy for this bioregion, based on the mapping presented here and in related studies (Rees 1993; Jordan *et al.* 2001) and the results of biological surveys (e.g. Edgar *et al.* 1995; Moverley and Jordan 1996; Jordan *et al.* 1998; Edgar *et al.* 1999; Murphy and Lyle 1999). This strategy, in combination with an MPA network, would ensure that habitats and unusual or unique areas gain a high level of protection. Such a strategy could also be used to protect critical habitats for some of the species that are known to be endemic to this bioregion (Edgar *et al.* 1995).

Ideally, areas suggested as being suitable for MPA locations would include representatives from all the broad habitat categories, including the range of characteristic features for that habitat type. They would also include areas identified as being unique or characteristic of an area. The most suitable locations are those that include a wide range of habitats within a relatively small geographical area. Habitat mapping has generated a GIS capability to calculate these areas at the appropriate stage of the planning process. The 1:100,000 scale maps can be used to gain an indication of the relative components included, and at the 1:25,000 scale give finer detail. Habitat mapping can also act as a GIS framework for more detailed community descriptions to be developed in the future as resources become available to conduct finer-scale biological inventories.

Ideally, if MPA locations are to be nominated they should fulfil a number of criteria recognised by the Tasmanian Marine Protected Areas Strategy (2001). These include representativeness, size and complexity. Nominated areas should be sufficiently large, with appropriately chosen boundaries to adequately protect populations of the species they are intended to represent. For many resident reef fishes this may include the entire home reef (because many resident species rarely leave that area (Barrett 1995)) or, where the reef is large and continuous, sufficient coast for the proportion of the population moving across MPA boundaries not to be significant. Small-scale maps provide the capacity to identify locations that offer the possibility of protecting a range of distinct habitats and associated species assemblages within each MPA, increasing the number of habitats and species represented (local biodiversity) while minimising the total number of MPAs needed to afford protection. Additional selection criteria should include community acceptance and public access, if the maximum conservation benefits are to be obtained from these areas.

In conclusion, habitat mapping is an essential tool needed to determine MPA options in an ecologically complex bioregion, to select appropriate boundaries and to ensure that protected areas are comprehensive, adequate and representative. Suggestions for potential MPA locations can be objectively derived from the mapping results in a process aimed at maximising the habitat diversity for each location. It is recognised that numerous alternative compromises exist and habitat maps can be used to facilitate discussion of all possibilities.

The broad nature of these surveys often means that, other than dominant macroalgae and seagrasses, unique features at the species, population or community level can not be readily detected. Substantial additional biological information is needed, however, if the protection of small-scale unique features or species distribution is to be an important component of the MPA planning process. Much of this information often already exists but requires analysis within an MPA framework. As more biological information becomes available within the Bruny bioregion, important and unique features requiring some level of protection may be identified that are not included in the present MPAs. Sufficient habitat information also now exists for the development of a habitat management strategy, and this should be developed concurrently with the MPA planning process. Such a strategy, together with MPAs and appropriate fisheries and land use management, would provide the flexibility required to conserve marine and estuarine biodiversity within the Bruny bioregion in the long term.

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USE OF A TEMPERATE REEF-FISH COMMUNITY TO IDENTIFY PRIORITIES IN THE ESTABLISHMENT OF A MARINE PROTECTED AREA

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Abstract

Few studies have dealt with biodiversity, composition and dynamics of temperate reef fish. The present study area is a 33 km stretch of coastline (53 km^2) on the west Portuguese coast that has recently been assigned as a Marine Park (Marine Park of the Arrábida Nature Park), for which basic information on composition of the marine communities is very scarce. From a biogeographical perspective, mainland Portugal is in a transitional zone where many species of cold- and warm-water fish reach their southern and northern limits of distribution respectively. This situation contributes strongly to a high level of biodiversity in the Lusitanian province, and also makes it very sensitive to climatic oscillations such as those predicted as part of global warming.

This study analysed the fish community composition in the marine park and ascribed a hierarchical importance for the coastal sectors and the different habitats present. The results reflect the heterogeneous nature of the substrata present and their significant differences in biodiversity values and in the occurrence of rare species. For each species, dispersion and abundance indexes were calculated and species that require particular attention are noted. Appropriate management measures are suggested. Procedures for the implementation of these measures must be suited to a situation where basic biological information is scarce. This research is included in a broader project aimed at building a long-term database of the fish communities in this area, assessing the main factors influencing their structure and distribution patterns, and monitoring reserve effects in the long term.

Keywords: fish communities, dispersion index, abundance index, marine reserve design, management plan

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INTRODUCTION

The advent of SCUBA diving opened a new era in ecological studies of fish communities in hard substrata (Harmelin-Vivien and Harmelin 1975), especially in coral reef ecology where a large number of visual census techniques were developed (e.g. Ehrlich 1975; Colton and Alevizon 1981; Doherty and Williams 1988; Sale 1988, 1991a; Greenfield and Johnson 1990). These techniques contributed decisively to a profound change of perspective in ecological studies of coral-reef fish communities. Basic descriptive research, in which assemblage composition, biomass assessment and food-web characterization had priority, has evolved into a new stage in which a number of important theoretical issues are guiding the scientific research (e.g. Sale 1978, 1988, 1991b; Barlow 1981; Doherty 1991; Ebeling and Hixon 1991; Hixon 1991; Leis 1991; Williams 1991). The development of accurate quantitative census techniques applied to long-term studies opened the door to studies of stability, resilience and the impact of disturbances in reef fish communities. Regular monitoring of recruitment processes, combined with that of adults, is helping to assess the importance of stochastic and deterministic control mechanisms in different habitats and geographical locations. It is also helping to detect which life stages are most susceptible to the controlling factors limiting the populations of each species (e.g. Williams and Sale 1981; Brothers *et al.* 1983; Victor 1986; Sale 1988; Thresher 1991; Victor 1991). Quantitative data on fish populations and new forms of habitat characterization, often combined with experimental manipulations, are helping to clarify the relationship between biodiversity and habitat complexity. Finally, these new monitoring methods are allowing the study of the impact of human activities, both fishing and habitat degradation, and are essential in the building of predictive models of drastic environmental changes, such as those likely to be caused by global warming, as well as in the planning, design and management of marine protected areas.

In comparison with studies on coral reefs, those on temperate rocky shores have progressed at a much slower pace, largely owing to the harsh environment of these habitats (e.g. Stephens and Zerba 1981; Stephens *et al.* 1984; Jansson *et al.* 1985; Diamant *et al.* 1986; Harmelin 1987). Information on subtidal fish communities is available for only a limited number of sites (Harmelin-Vivien and Harmelin 1975; Stephens *et al.* 1984; Jansson *et al.* 1985; Bodkin 1986; Harmelin 1987, 1990; Clavijo *et al.* 1989; Illich and Kotrschal 1990; Bortone *et al.* 1991; Falcón *et al.* 1993; Henriques *et al.* 1999), despite the fact that almost all important questions raised by coral-reef fish ecology are fully applicable to temperate habitats. On European shores, many of these studies have been carried out in the Mediterranean (Harmelin 1987, 1990; Zander 1992; Ody and Harmelin 1994; Jouvenel 1997) and more studies at higher latitudes are clearly needed (Jansson *et al.* 1985; Henderson 1989; Minchin 1987). This applies especially to long-term studies with adequately standardized procedures. Such information is fundamental for comparative analysis between geographical locations and to distinguish between inter-annual fluctuations and long-term trends, such as those predicted by global warming. This applies in particular to habitats that are changing rapidly, because of either climatic change (in its broad sense) or habitat degradation.

The role of marine protected areas in preserving intact marine habitats, where these comparative studies can be performed in a meaningful way, is fundamental for evaluation and comparison with other areas where human activities are severely affecting the marine communities (Fishelson 1980; Santos *et al.* 1995; Rakitin and Kramer 1996; Kramer and Chapman 1999; Roberts and Hawkins 2000; Côté *et al.* 2001).

From a biogeographical perspective, mainland Portugal is in a transitional zone where many species of cold- and warm-water fish reach their southern and northern limits of distribution, respectively. This situation contributes strongly to a high level of biodiversity in the Lusitanian province (of which mainland Portugal makes a very substantial proportion (Ekman 1953; Briggs 1974)), and also makes it very sensitive to climatic oscillations such as those predicted as part of global warming.

This study was conducted at Arrábida Marine Park (AMP), Portugal, with the aim of characterising the rocky-habitat fish fauna of this region, thereby providing a reference database against which future studies can be compared and a framework to the design and management plan of this MP. One main objective was to characterise the situation before the creation of the MP; this serves as a baseline study against which future surveys can be compared after the implementation of the protective measures; it also suggests a number of management measures for this Area. The results are discussed in terms of the biogeographic importance of the study area and its relevance for conservation.

METHODS

Study area

Our study area is a 33 km stretch of coastline (53 km²) and comprises the rocky shore and adjacent mixed sandy substrata between Cape Espichel (38º27'N, 9º12'W) and Portinho da Arrábida (38º29'N, 8º57'W), on the west coast of Portugal (Fig. 1).

Most of the study area faces south, being protected from the prevailing north and north-west winds by the adjacent mountain chain of Arrábida. The shore is very steep and the intertidal zone includes mainly rocky cliffs, small beaches and several areas covered by boulders. The subtidal zone begins with a narrow stretch of rocky substratum that extends offshore for some tens of metres, and to depths of less than 20 m (except at the Espichel Cape area where it reaches more than 40 m). Many stones and boulders, from a few centimetres to several metres in size, resulting from the erosion of the nearby calcareous cliffs increase habitat complexity. In some places, sandy beaches interrupt this stretch. Beyond the rocky substratum, sandy bottoms that usually present gentle slopes are found. Between Sesimbra and Setúbal there is a terrestrial Nature Park created in 1976 (Arrábida Nature Park) that since 1998 has included a marine area – the AMP.

Until a few years ago, the study area harboured extensive eelgrass beds, which are now almost absent as a result of unrestrained clam harvesting. During late spring and summer, dense algal beds are present in many places, ranging from dense tufts of *Asparagopsis armata* Harvey (a probable invasive species from Australia), to some brown algae such as *Cystoseira usneoides* (L.) and, in some sites, *Saccorhiza polyschides* (Lightfoot). Crustose red algae have formed reefs in some areas. The filter-feeding invertebrate fauna is particularly developed and abundant in the MP. A detailed description of the algal and invertebrate communities can be found in Saldanha (1974).

This area is near the northern limit of the main north-east Atlantic upwelling events (Wooster *et al.* 1976). This means that, during the summer, water temperature nearshore is frequently lower than that of the offshore waters at the same latitude. Water temperature can vary from around 13ºC in January to 21ºC in September (Almada *et al*. 1990, based on data from a nearby meteorological station).

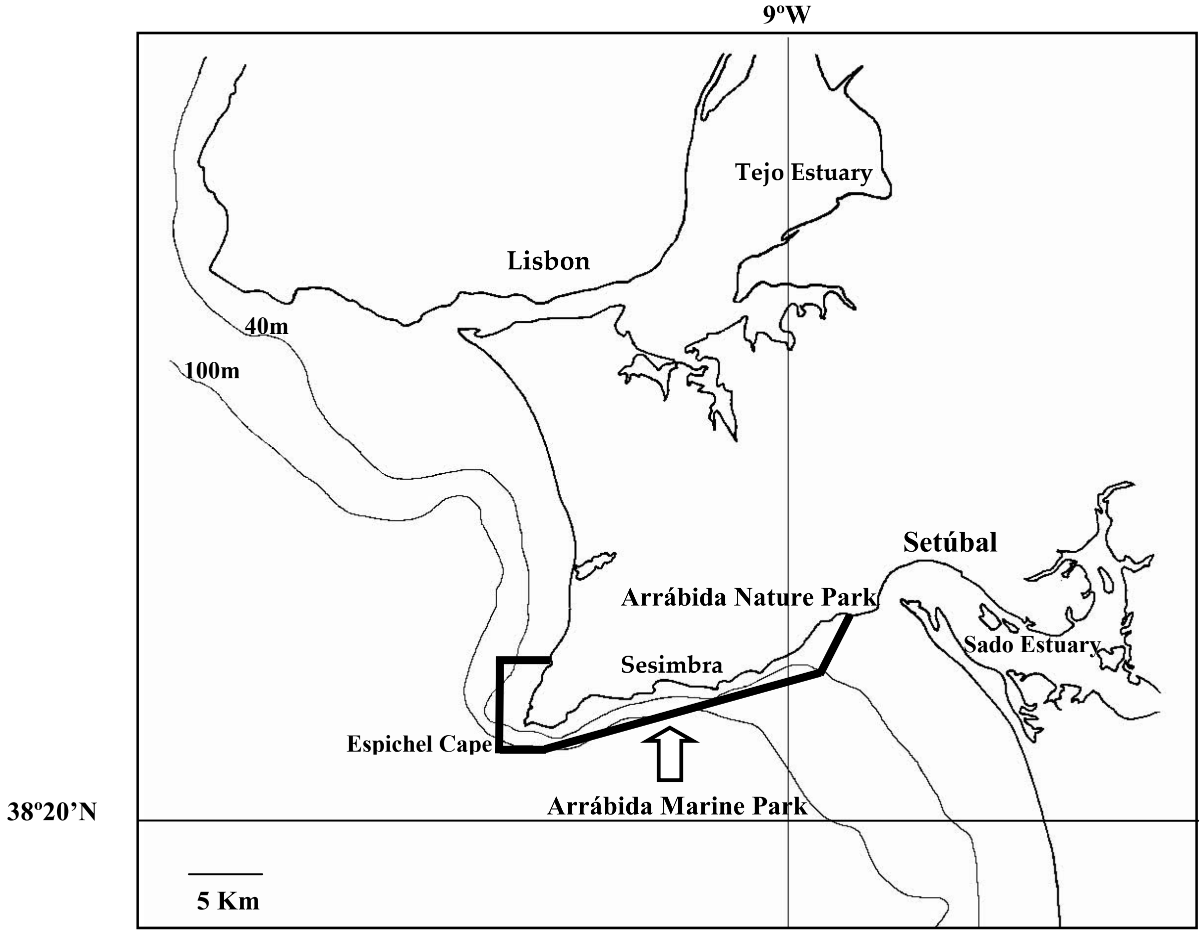

Fig. 1. Location of the study area in the west coast of Portugal.

Data collection

Data were collected from May 1996 to February 2000 during SCUBA surveys. Dives were made every month, with a few exceptions in the winter due to rough seas. Each dive lasted on average one hour and the habitats and microhabitats prospected ranged from the surface down to the limit of the rocky substratum (8 to 30 m). About ten metres of adjacent sandy bottoms were also inspected. In each station (Fig. 2), the sampling procedure started in the adjacent sandy bottom and followed a line perpendicular to the coast as far as the intertidal zone.

In each dive, a cumulative list of the observed species was updated and the new occurrences were added for each habitat. For each species, a qualitative scale of abundance (Harmelin-Vivien and Harmelin 1975) was determined for each station as follows: 1, Single observation (one individual); 2, Rare (2–10); 3, Common (11–100); 4, Abundant (>100). Apart from this qualitative database, the habitats, microhabitats, depths, fish size, patterns of aggregation, and the occurrences of juveniles and their sizes, were also noted. Based on this information, the following indexes were calculated:

$$\text{Dispersion Index} = \frac{\text{Number of stations where the species occurred}}{\text{Total number of stations}}$$

$$\text{Abundance Index} = \frac{\sum \text{ of abundances in the stations where the species occurred}}{\text{Total number of stations where the species occurred}}$$

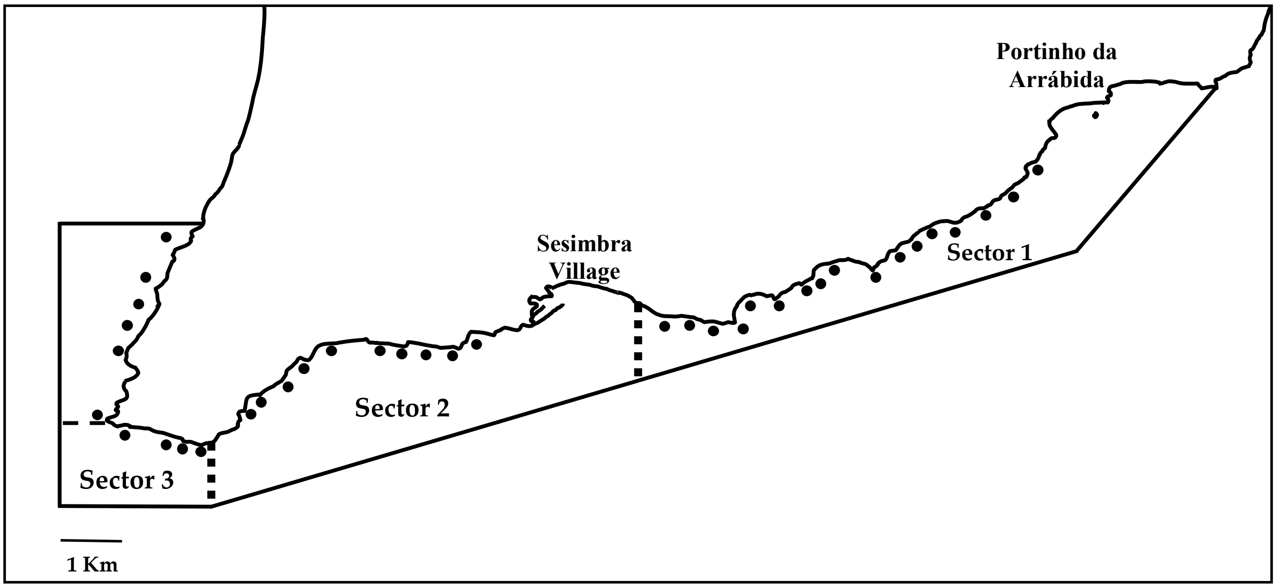

Fig. 2. Sampling stations (≈ 500m apart) and sectors, Arrábida Marine Park, Portugal.

For the Dispersion Index the occurrence of each species in the sampled stations was rated as follows: 4, Regular (> 50% of stations); 3, Scattered (>25% and ≤50%); 2, Localized (>10% and ≤25%); and 1, Very Localized (≤10%). For the Abundance Index, the following groups were defined: 4, Abundant (>3 to 4); 3, Common (>2 and ≤3); 2, Rare (>1 and ≤2); 1, Very Rare (=1).

RESULTS

The main habitats and microhabitats present in the study area were evaluated and mapped for each station. This method allowed the division of the study area in three main sectors (Fig. 2).

Sector 1 is characterised by an extensive vertical coastline with very high (to 400 m) and steep calcareous cliffs. The disintegration of these cliffs forms a very heterogeneous subtidal habitat characterised by the presence of random-sized blocks of rock (from a few centimetres to several metres) and of boulder fields (in shallow water). The rocky habitat occurs in a narrow band and is relatively shallow in this sector (to 20 m depth). Some small and protected bays with intertidal boulder fields are also present. The rocky substratum is bordered by sandy bottoms that can reach 100 m around the limits of the MP. In the eastern part of the sector shallow sandy bottoms and sand banks are present, as well as small and very protected bays with intertidal boulder fields.

In *Sector 2* the main substratum type is "bedrock", which constitutes a natural continuation of the rocky formations found inland in this area. They are much less diversified than the random-sized blocks of rock and present fewer microhabitats.

Sector 3 has two sections. In the eastern section, the rocky vertical substratum reaches medium depths (10—15 m) and the bedrock bottoms can occur at 30–40 m. In the western (most exposed) section the bedrock habitat is very homogeneous and extends to several hundreds of metres from shore with a gentle slope to around 30 m.

The 96 fish species recorded (Table 1) constitute a high level of fish biodiversity for this biogeographic region (Henriques *et al.* 1999). At present, we have recorded more than 110 species of coastal fish in the MP (including more recent data from other studies), using visual identification techniques. Some species typical of sandy bottoms adjacent to seagrass beds were not recorded since these beds are now severely reduced. The three identified sectors were compared for the occurrence of each species. Only species that were recorded in more than 25% of the sampled stations were considered, to reduce the error introduced by rare species (Table 2). Species diversity was significantly different between sectors (ANOVA: F = 5.45, d.f. 2, p<0.01) with a significant difference between extremes (Spjotvoll/Stoline test: Sector 1 *v*. Sector 3 = p<0.05) and with Sector 1 presenting a higher mean number of species (Fig. 3). The percentage of occurrence of rare species (<25% of the sampled stations) is presented in Fig. 4. There is a decrease in the occurrence of rare species from Sector 1 to Sector 3.

Table 1. Fish species recorded in the Arrábida Marine Park, Portugal.
Classification follows Whitehead *et al.* (1984/86) for genus and species and Nelson (1994) for families. *Gobius xantocephalus* (Heymer and Zander 1992) replaces *G. auratus* cited in Whitehead *et al.* (1984/86). *Sphoeroides marmoratus* (Shipp 1990) replaces *S. spengleri* cited in Whitehead *et al.* (1984/86).

Family	Species	Family	Species
Carcharhinidae	*Carcharhinus plumbeus* (Nardo, 1827)	Labridae	*Centrolabrus exoletus* (Linnaeus, 1758)
Muraenidae	*Muraena helena* Linnaeus, 1758		*Coris julis* (Linnaeus, 1758)
Congridae	*Conger conger* (Linnaeus, 1758)		*Ctenolabrus rupestris* (Linnaeus, 1758)
Clupeidae	*Sardina pilchardus* (Walbaum, 1792)		LABRUS BERGYLTA Ascanius, 1767
Phycidae	*Ciliata mustela* (Linnaeus, 1758)		*L. bimaculatus* Linnaeus, 1758
	Gaidropsarus mediterraneus (Linnaeus, 1758)		*Symphodus bailloni* (Valenciennes, 1889)
	G. vulgaris (Cloquet, 1824)		*S. cinereus* (Bonnaterre, 1788)
	Phycis phycis (Linnaeus, 1766)		*S. melops* (Linnaeus, 1758)
Gadidae	*Pollachius pollachius* (Linnaeus, 1758)		*S. roissali* (Risso, 1810)
	Trisopterus luscus (Linnaeus, 1758)		*S. rostratus* (Bloch, 1797)
Batrachoididae	*Halobatrachus didactylus* (Schneider, 1801)	Ammodytidae	*Ammodytidae n.id.*
Mugilidae	*Chelon labrosus* (Risso, 1826)		*Hyperoplus lanceolatus* (Le Sauvage, 1824)
	Liza aurata (Risso, 1810)	Trachinidae	*Echiichthys vipera* (Cuvier, 1829)
	L. ramada (Risso, 1826)	Tripterygiidae	*Tripterygion delaisi* Canedat & Blache, 1971
Atherinidae	*Atherina presbyter* Cuvier, 1829	Blenniidae	*Coryphoblennius galerita* (Linnaeus, 1758)
Belonidae	*Belone belone* (Linnaeus, 1761)		*Lipophrys canevai* (Vinciguerra, 1880)
Zeidae	*Zeus faber* Linnaeus, 1758		*L. pholis* (Linnaeus, 1758)
Syngnathidae	*Entelurus aequoreus* (Linnaeus, 1758)		*L. trigloides* (Valenciennes, 1836)
	Hippocampus hippocampus (Linnaeus, 1758)		*Parablennius gattorugine* (Brunnich, 1768)
	Syngnathus acus Linnaeus, 1758		*P. incognitus* (Bath, 1968)
Macroramphosidae	*Macroramphosus scolopax* (Linnaeus, 1758)		*P. pilicornis* (Cuvier, 1829)
Scorpaenidae	*Scorpaena notata* Rafinesque, 1810		*P. rouxi* (Cocco,1883)
	S. porcus Linnaeus, 1758		*P. sanguinolentus* (Pallas, 1811)
Triglidae	*Trigloporus lastoviza* (Brünnich, 1768)	Gobiesocidae	*Apletodon dentatus* (Fascciola, 1887)
Cottidae	*Taurulus bubalis* (Euphrasen, 1786)		*Diplecogaster bimaculata* (Bonnaterre, 1788)
Moronidae	*Dicentrarchus labrax* (Linnaeus, 1758)		*Lepadogaster candollei* Risso, 1810
Serranidae	*Serranus atricauda* Günther, 1874		*L. lepadogaster* (Bonnaterre, 1788)
	S. cabrilla (Linnaeus, 1758)	Callionymidae	*Callionymus lyra* Linnaeus, 1758
	S. hepatus (Linnaeus, 1758)		*C. reticulatus* Valenciennes, 1837

Family	Species	Family	Species
Carangidae	*Trachurus trachurus* (Linnaeus, 1758)	Gobiidae	*Gobius cobitis* Pallas, 1811
Sparidae	*Boops boops* (Linnaeus, 1758)		*G. cruentatus* Gmelin, 1789
	Diplodus annularis (Linnaeus, 1758)		*G. gasteveni* Miller, 1974
	D. bellottii (Steindachner, 1882)		*G. niger* Linnaeus, 1758
	D. cervinus (Lowe, 1838)		*G. paganellus* Linnaeus, 1758
	D. puntazzo (Cetti, 1777)		*G. xantocephalus* Heymer & Zander, 1992
	D. sargus (Linnaeus, 1758)		*Gobiusculus flavescens* (Fabricius, 1779)
	D. vulgaris (E.G. Saint-Hilaire, 1817)		*Pomatoschistus marmoratus* (Risso, 1810)
	Oblada melanura (Linnaeus, 1758)		*P. pictus* (Malm, 1865)
	Pagellus acarne (Risso, 1826)		*Thorogobius ephippiatus* (Lowe, 1839)
	Pagrus auriga (Valenciennes, 1843)	Scombridae	*Scomber japonicus* Houttutyn, 1782
	P. pagrus (Linnaeus, 1758)	Bothidae	*Arnoglossus thori* Kyle, 1913
	Sarpa salpa (Linnaeus, 1758)		*Bothus podas* (Delaroche, 1809)
	Sparus aurata (Linnaeus, 1758)	Scophthalmidae	*Phrynorhombus regius* (Bonnaterre, 1788)
	Spondyliosoma cantharus (Linnaeus, 1758)		*Zeugopterus punctatus* (Bloch, 1787)
Centracanthidae	*Spicara n.id.*	Soleidae	*Solea senegalensis* Kaup, 1858
Mullidae	*Mullus surmuletus* Linnaeus, 1758		*Synaptura lusitanica* (Capello, 1868)
Pomacentridae	*Chromis chromis* (Linnaeus, 1758)	Balistidae	*Balistes carolinensis* Gmelin, 1789
		Tetraodontidae	*Sphoeroides marmoratus* (Lowe, 1839)
		Molidae	*Mola mola* (Linnaeus, 1758)

The most representative families were the Labridae (wrasses) with 9 species, the Sparidae (sea breams) with 6 species, the Gobiidae (gobies) with 5 species and the Blenniidae (blennies) with 3 species (Table 2). Eleven species occurred in more than 80% of the sampled stations (Table 2): six wrasses two sea breams, two blennioids and a grouper. These are species that have a consistent presence throughout the MP.

For each species, the abundance and dispersion indexes were determined. These indexes allow the identification of ubiquitous species and also abundant but localised species. For a clearer presentation of this information, the dispersion values of abundant and common species and the abundance values of regular and scattered species were determined (Table 3). Four species were very abundant but localised or very localised in the study area; these species were associated with a specific habitat that occurs in only a few places in the study area, or they appeared at a specific time of the year (e.g. reproductive schools of *L. ramada* that migrate from the nearby estuary in November). On the other hand, 17 species were rare but occurred regularly in the sampled stations (Table 3).

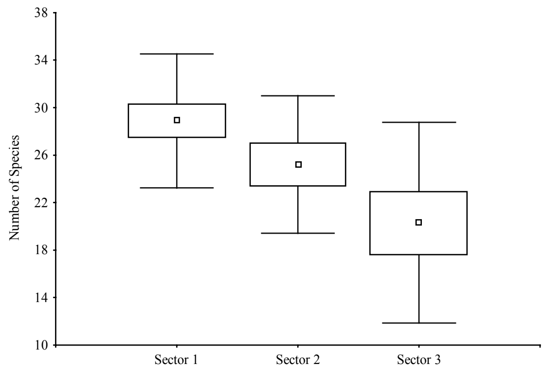

Fig. 3. Number of fish species in each sector of the Arrábida Marine Park. Mean (squares), standard error (boxes) and standard deviation (whiskers). Sector 1, N = 17, Sector 2, N = 10, Sector 3, N = 10.

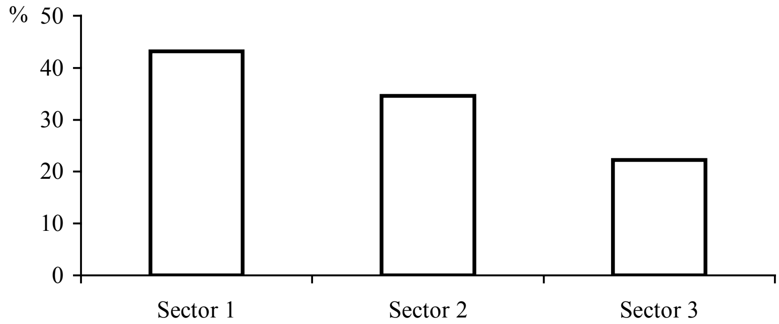

Fig. 4. Occurrence (%) of rare fish species (recorded in <25% of the sampled stations) per Sector, Arrábida Marine Park.

Table 2. Distribution of fish species in the Arrábida Marine Park; percentage and number (in parenthesis) of stations where each species occurred in each sector and for all stations sampled (for species with >25% of occurrences in the sampled stations).

Species	Family	Sector 1 % (*N*)	Sector 2 % (*N*)	Sector 3 % (*N*)	Total (%)
Coris julis	Labridae	100% (17)	100% (10)	100% (10)	100%
Ctenolabrus rupestris	Labridae	100% (17)	100% (10)	100% (10)	100%
Diplodus vulgaris	Sparidae	100% (17)	90% (9)	100% (10)	97%
Parablennius pilicornis	Blenniidae	100% (17)	100% (10)	80% (8)	95%
Symphodus melops	Labridae	94% (16)	100% (10)	90% (9)	95%
Labrus bergylta	Labridae	94% (16)	90% (9)	90% (9)	92%
Symphodus bailloni	Labridae	88% (16)	100% (10)	90% (9)	92%
Centrolabrus exoletus	Labridae	94% (16)	90% (9)	80% (8)	89%
Diplodus sargus	Sparidae	88% (15)	100% (10)	70% (7)	86%
Serranus cabrilla	Serranidae	82% (14)	90% (9)	80% (8)	84%
Tripterygion delaisi	Tripterygiidae	100% (17)	90% (9)	50% (5)	84%
Gobiusculus flavescens	Gobiidae	59% (10)	90% (9)	80% (8)	73%
Symphodus roissali	Labridae	94% (16)	60% (6)	40% (4)	70%
Gobius xantocephalus	Gobiidae	88% (15)	80% (8)	20% (2)	68%
Parablennius gattorugine	Blenniidae	76% (13)	60% (6)	60% (6)	68%
Boops boops	Sparidae	53% (9)	50% (5)	70% (7)	57%
Callionymus reticulatus	Callionymidae	59% (10)	80% (8)	30% (3)	57%
Sarpa salpa	Sparidae	65% (11)	80% (8)	20% (2)	57%
Diplodus cervinus	Sparidae	41% (7)	80% (8)	50% (5)	54%
Scorpaena notata	Scorpaenidae	76% (13)	20% (2)	40% (4)	51%
Lepadogaster candollei	Gobiesocidae	53% (9)	50% (5)	40% (4)	49%
Atherina presbyter	Atherinidae	76% (13)	40% (4)	0% (0)	46%
Symphodus cinereus	Labridae	53% (9)	60% (6)	10% (1)	43%
Gobius paganellus	Gobiidae	47% (8)	30% (3)	40% (4)	41%
Scorpaena porcus	Scorpaenidae	53% (9)	10% (1)	50% (5)	41%
Apletodon dentatus	Gobiesocidae	24% (4)	50% (5)	50% (5)	38%
Gobius cruentatus	Gobiidae	59% (10)	30% (3)	10% (1)	38%
Mugilidae n.id.	Mugilidae	65% (11)	20% (2)	0% (0)	35%
Mullus surmuletus	Mullidae	35% (6)	20% (2)	50% (5)	35%
Labrus bimaculatus	Labridae	47% (8)	20% (2)	10% (1)	30%
Spondyliosoma cantharus	Sparidae	29% (5)	30% (3)	30% (3)	30%
Thorogobius ephippiatus	Gobiidae	47% (8)	10% (1)	20% (2)	30%
Trisopterus luscus	Gadidae	29% (5)	20% (2)	40% (4)	30%
Coryphoblennius galerita	Blenniidae	47% (8)	20% (2)	0% (0)	27%
Serranus atricauda	Serranidae	29% (5)	20% (2)	30% (3)	27%

Table 3. Dispersion values of abundant and common species and abundance values of regular and scattered species in the Marine Park.

Dispersion Index: 4, Regular (>50% of stations); 3, Scattered (>25% and ≤50%0; 2, Localized (>10% and ≤25%); and 1, Very Localized (≤10%). Abundance Index: 4, Abundant (>3 to 4); 3, Common (>2 and ≤3); 2, Rare (>1 and ≤2); 1, Very Rare (=1).

Dispersion of Abundant and Common Species

Species	**Dispersion**
Coris julis	4
Diplodus vulgaris	4
Callionymus reticulatus	4
Centrolabrus exoletus	4
Ctenolabrus rupestris	4
Gobius xantocephalus	4
Gobiusculus flavescens	4
Labrus bergylta	4
Parablennius pilicornis	4
Sarpa salpa	4
Serranus cabrilla	4
Symphodus bailloni	4
Symphodus melops	4
Symphodus roissali	4
Tripterygion delaisi	4
Apletodon dentatus	3
Atherina presbyter	3
Coryphoblennius galerita	3
Lepadogaster lepadogaster	2
Pomatoschistus marmoratus	2
Pomatoschistus pictus	2
Liza ramada	1

Locally Abundant

Abundance of Regular and Scattered Species

Species	**Abundance**
Coris julis	4
Diplodus vulgaris	4
Callionymus reticulatus	3
Centrolabrus exoletus	3
Ctenolabrus rupestris	3
Gobius xantocephalus	3
Gobiusculus flavescens	3
Labrus bergylta	3
Parablennius pilicornis	3
Sarpa salpa	3
Serranus cabrilla	3
Symphodus bailloni	3
Symphodus melops	3
Symphodus roissali	3
Tripterygion delaisi	3
Apletodon dentatus	3
Atherina presbyter	3
Coryphoblennius galerita	3
Boops boops	2
Diplodus cervinus	2
Diplodus sargus	2
Parablennius gattorugine	2
Scorpaena notata	2
Gobius cruentatus	2
Gobius paganellus	2
Labrus bimaculatus	2
Lepadogaster candollei	2
Mugilidae n.id.	2
Mullus surmuletus	2
Scorpaena porcus	2
Serranus atricauda	2
Spondyliosoma cantharus	2
Symphodus cinereus	2
Thorogobius ephippiatus	2
Trisopterus luscus	2

Rare but Regular

DISCUSSION

In the past decade or so, we have recorded an increase in sea-water temperature and the complete disappearance of laminarian beds from the AMP. This is a transitional zone, with many fish species finding their northern or southern limits of distribution within the area (Henriques *et al.* 1999). Similar situations have been described for other regions. Temperature change in the eastern Pacific has caused the disappearance and subsequent reappearance of the kelp beds 'typical' of temperate regions in southern California (Eber 1981). This is a transitional zone between the cold and the warm temperate north-eastern Pacific regions, and a gradient of northern and southern faunal elements is present (Stephens *et al.* 1984). In warm years, there is an increase in warm-water species and a decrease in cold-water species.

Major changes in weather trends (e.g. the NAO – North Atlantic Oscillation and the ENSO – El Niño/Southern Oscillation) or sea currents are also potential disturbance factors that must be taken in consideration when trying to explain present-day distribution patterns on the basis of short-term studies. These factors, among many others, are likely to influence the abundance and diversity of reef-fish communities and must be taken into consideration in the design of marine reserves.

The biodiversity of the coastal fish community in our study area is quite high compared with values published for similar latitudes in the north-eastern Atlantic and the Mediterranean (see Henriques *et al.* 1999), as well as for Californian coastal reef fishes (e.g. Palos Verdes 73 species – Stephens *et al* 1984; King Harbour 105 species – Stephens and Zerba 1981).

This is also true for the few other taxa that have been investigated in the study area, with more than 1100 species of marine macroorganisms having already been recorded in a few studies (e.g. Palminha 1958; Saldanha 1974; Calado and Urgorri 1999; Henriques *et al.* 1999). This emphasises the extreme importance of the AMP, since it provides representative habitats for a high proportion of the total shallow-water fish and invertebrate fauna of the Portuguese mainland shores (Henriques *et al.* 1999).

The management plan for the park is presently under a public discussion process, in which the results of this study allowed the suggestion of several management and design measures. The main objective of this MP is the conservation of coastal biodiversity and rocky-shore habitats. Although one of the objectives is also to contribute to the sustainability of local fisheries (local fishermen from Sesimbra village are greatly dependent upon the coastal marine resources), there is a great need for reference sites with minimum anthropogenic disturbance in this biogeographic region, and this MP is a key area in this respect.

The present work used the fish communities as a guide so that conservation efforts can be aimed at priority areas while some human activities are permitted in areas that are less important from a conservation point of view. Since the basic ecological information was very scarce, the management and proposed design plan has to be evaluated and continually monitored, and at the same time other key marine taxa must be studied.

The main disturbances present in the AMP are the high level of exploitation of natural resources and the uncontrolled proliferation of leisure activities. These leisure activities include sport fishing (angling and spearfishing), nautical sports, SCUBA diving and several beach activities (most intense in the summer months). The main guidelines for the conservation measures to be implemented in the area based on the results of the present study, point to a severe reduction or prohibition of the most destructive human activities. Additionally, the most important area from a conservation point of view (Sector 1) should include a fully protected area with a partially protected area in each side. The fully protected area should constitute a reference site for monitoring the impacts of the protective measures to be adopted, and should also function as a study area for evaluation of changes in the composition of marine communities induced by natural factors. Only scientific research and monitoring procedures should be allowed in this area. In the partially protected areas, only non-extractive human activities that can be unequivocally compatible with the conservation objectives should be considered. In the area of the Cape Espichel, where the greatest depths of the rocky habitat and the conditions of rough seas allow the development of a distinct community of marine organisms, a partially protected area is also proposed. This means that an area of approximately 50% of the Marine Park would be fully protected from extractive activities. The economic dependence of the Sesimbra village fishing community requires the maintenance of some traditional fishery activities in the remaining areas of Sector 2 and Sector 3, and does not allow the extension of the fully protected area to the whole Marine Park. However, the restriction of those activities to more-selective fishing gear (angling and traps) is proposed. These transition zones will also serve to monitor the impact of conservation measures to be adopted in the marine park and to compare the development of marine communities with those in the other areas where extractive human activities are forbidden.

The results and proposals presented in this study are a starting point to the identification of the most relevant areas from a conservation point of view in a situation where the basic biological and ecological information is lacking. This simple approach allowed us to gather sufficient data for inclusion in a management plan proposal for the MP. Political time schedules are incompatible with the time needed to gather enough scientific information to create a more or less "bullet-proof" plan, but an appropriate set of measures can be proposed on the basis of data collected for a few key elements of the community in a short to medium-term study (four years in the present case). There is, however, a need to implement adequate monitoring procedures to evaluate the proposed measures and modify the plan accordingly.

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