THE GLOBAL NETWORK OF MARINE PROTECTED AREAS: DEVELOPING BASELINES AND IDENTIFYING PRIORITIES

by

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Abstract

Recently adopted global marine protection targets aim to protect 10-30% of marine habitats within the next 3 to 5 years. However, these targets were adopted without prior assessment of their attainability. Moreover, our ability to monitor progress towards such targets has been constrained by a lack of robust data on marine protected areas (MPAs). In this thesis I present the results of the first explicitly marine-focused, global assessment of MPAs in relation to three global marine protection targets. Approximately 2.35 million km^2, equivalent to 0.65% of the world's oceans, are currently protected, and only 12% of that is 'no-take'. Over the last two decades, the marine area protected globally has grown at ~5% per year. At this rate, even the most modest target is unlikely to be met for at least several decades.

The utility of large-scale conservation targets has been repeatedly questioned, although mainly on ecological grounds. However, if, as is suggested here, their primary role is to motivate behavioural change, then a more serious problem is that they seem to be failing in this regard, too. I explore possible reasons for this and suggest two main problems: firstly, an as yet unmet need to develop a hierarchical system of targets that reflects the multi-scale and pluralistic nature of ecological and political systems; and secondly, feedback mechanisms between political will, perceived attainability, and target formulation which may impede implementation of the targets.

Since the adoption of the global targets, no implementation strategy has been developed, which may also impede target attainment. In order to fill this gap, I applied a rarity-complementarity heuristic place prioritisation algorithm (PPA) to a dataset consisting of 1038 global species distributions with 0.5° latitude/longitude resolution, under ten scenarios devised to reflect the

global targets. This is the first time that species distribution ranges of marine species have been used in a globally synthetic way, and is by far the largest application of a PPA to date. Global priority areas for protection are identified for each scenario, which may be used to identify where regional-scale protected areas network design efforts might be focused.

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Co-authorship Statement

Apart from Chapters 1 and 7, all chapters have been prepared as stand-alone manuscripts for submission to a peer-reviewed journal. They are currently all either published, accepted, or in review, with the exception of Chapter 6, which is in preparation for submission. I am the senior author on all papers, and I assumed primary responsibility for the design, implementation, analysis, and writing of co-authored papers. I am the sole author on Chapters 3 and 4, although Daniel Pauly provided guidance on both. The contributions of co-authors to Chapters 2, 5 and 6 are summarized below.

Chapter 2 is co-authored by Lucy Fish, Josh Laughren, and Daniel Pauly. Lucy Fish provided the original World Database on Protected Areas MPA data that were used to build the MPA database, and feedback on the manuscript. Josh Laughren provided feedback on the design of the research program and the manuscript. Daniel Pauly provided guidance on all stages of the work.

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1. Introduction

Rationale

The negative impacts of human activities, both exploitative and non-exploitative, on the world's coastal and marine ecosystems are now widely documented. Marine habitats have undergone substantial declines over the last few decades – and, indeed, centuries (Jackson, 2001) – among the most well documented being coral reefs (Roberts *et al.*, 2002; Wilkinson, 2002; Bellwood *et al.*, 2004; Côté *et al.*, 2005; Graham *et al.*, 2006), seagrasses (Fortes, 1988; Duarte, 2002), and mangroves (Valiela *et al.*, 2001; Alongi, 2002). Concerns are now also growing for offshore habitats, particularly seamounts and cold-water coral reefs (Koslow *et al.*, 2000; Clark and O'Driscoll, 2001; Koslow *et al.*, 2001; CBD, 2004b; Roberts and Hirshfield, 2004). Despite the difficulties in measuring marine extinction rates, there is growing evidence of rapid and profound declines and/or local extinctions for various marine species (Carlton, 1993; Gray, 1997; Casey and Myers, 1998; Roberts and Hawkins, 1999; Dulvy *et al.*, 2003; Hutchings and Reynolds, 2004). Indeed, there are now 416 marine species listed as Vulnerable, Endangered, or Critically Endangered on the 2007 IUCN Red List of Threatened Species, including various new additions such as Galápagos coral and seaweed species (IUCN, 2007a). These numbers are likely to increase as more species are assessed under the Global Marine Species Assessment (currently, only 1372 of the 40177 species assessed in the IUCN Red List are considered marine species) (IUCN, 2007b).

Exploitation and habitat loss and/or degradation are considered to be responsible for the vast majority of observed declines and local marine extinctions, much of which is attributable to fishing, the biggest and most ubiquitous threat to marine ecosystems globally (Jennings and Kaiser, 1998; Sadovy, 2001; Dulvy *et al.*, 2003; Myers and Ottensmeyer, 2005; Myers and Worm, 2005; Preikshot and Pauly, 2005). Worldwide, FAO reports that around 77% of the world's marine fisheries are either fully exploited, with no room for expansion, or overexploited (FAO, 2006, p29). Fisheries collapses have been observed at varying scales and from many parts of the world's oceans, most notably in the North Atlantic, where extensive data are available (Baum *et al.*, 2003; Christensen *et al.*, 2003; Myers and Worm, 2003). Although 95% of fisheries catches have historically been made on the continental shelf (Pauly and Christensen, 1995), fisheries have also been shifting to deeper waters over the last 50 years (Morato *et al.*, 2006), raising concerns for deep-sea fishes and habitats, which are generally vulnerable to, and slow to recover from, (over)exploitation and physical perturbation, respectively (Koslow *et al.*, 2000; Devine *et al.*, 2006; Davies *et al.*, in press).

These declines have all been observed within the context of existing fisheries management and governance mechanisms, which have, as a consequence, been criticised for failing to prevent the collapses (Botsford *et al.*, 1997; Browman and Stergiou, 2004). Common criticisms of traditional fisheries management mechanisms include the 'roving bandit' problem (Berkes *et al.*, 2006), which has masked the decline of individual stocks by their sequential replacement with new ones. This apparently sustained high level of catches reinforced the impression of inexhaustible stocks that had been expressed by the likes of Lamarck and Huxley in the 19th century (Roberts and Hawkins, 1999), leading to substantial overinvestment in the structural capacity of fleets during the 1970s, resulting in overcapacity, currently estimated as being double or more of the optimal level (Pauly *et al.*, 2002; Mace, 2004). Fisheries management has for the last 50 years been embedded in the neoclassical economic view, which considered the fish stock as the primary unit of management concern, and the fishing industry as the primary user of marine resources (Norse and Crowder, 2005). The development of natural resource economics and environmental economics did little to correct for the fundamental flaw in neoclassical economic theory that the economy is a closed system of production and consumption that is not dependent on the state and/or quality of the broader environment (Minteer and Manning, 2003). This presiding view (along with increased disciplinary segregation of science at the time) led to the development of single species stock assessment models, which gave little consideration to the ecosystem from which the fish were taken, including the habitats and other species that may have been impacted directly or indirectly through the process of fishing. Nevertheless, these models were used to estimate maximum sustainable yield (MSY), which formed the basis of many management recommendations, such as quotas. However, the MSY concept has been challenged for the last two decades on multiple grounds: it doesn't account for spatial variability; it is a high risk strategy (overshoot can have very serious consequences for stock persistence); it only considers the target species, and it fails to consider the costs of fishing (Botsford *et al.*, 1997).

In response to these criticisms, a popular current trend in the fisheries management literature is to move away from single-species management approaches towards an ecosystem approach to fisheries (Browman and Stergiou, 2004; Pikitch *et al.*, 2004). The underlying premise is that an ecosystem approach is spatially explicit, adaptive, considers uncertainties and external influences, and strives to balance societal objectives. Marine protected areas (MPAs) form an integral part of ecosystem-based management (Sumaila *et al.*, 2000) because they have the potential to: maintain and restore ecosystems, biodiversity and ecological processes; manage conflicting uses of ocean space; buffer against natural and anthropogenic uncertainty; promote integrated management of marine resources; augment fisheries through spillover and larval replenishment; and maintain aesthetic and traditional values (Jones, 1994; Alder, 1996; Jones, 2002; Gerber *et al.*, 2003; Lubchenco *et al.*, 2003). This, along with a growing sense of urgency about the need to mitigate and reverse the global declines in marine fisheries and habitats, has led to the adoption of various global marine protection targets in recent years.

The 2002 Plan of Implementation of the World Summit on Sustainable Development (WSSD) committed to establishing a representative global network of MPAs by 2012 (United Nations, 2002b, Section IV, paragraph 32(c)). At the Vth World Parks Congress (WPC) in 2003, the recommendation was made to "[g]reatly increase the marine and coastal area managed in marine protected areas by 2012; these networks should include strictly protected areas that amount to at least 20-30% of each habitat". Most recently, at the Eighth Ordinary Conference of the Parties (COP8) to the Convention on Biological Diversity (CBD) in 2006, a target that "at least 10% of each of the world's ecological regions [including marine and coastal be] effectively conserved [by 2010]" was adopted (CBD, 2006a).

The adoption of these targets automatically confers an explicit need to monitor progress towards their attainment. However, the only existing global database on protected areas, the World Database on Protected Areas (WDPA), maintained by the United Nations Environment Programme World Conservation Monitoring Centre (UNEP-WCMC), contained limited information for MPAs, enabling only very broad analyses on the total number and area of MPAs (Chape *et al.*, 2005). Prior to the work presented here, the level of protection in the world's oceans was estimated at around 0.5%, although these assessments suffered from data limitations (Kelleher *et al.*, 1995a; Chape *et al.*, 2005). The portion of the global marine area that was no-take, or strictly protected, was largely unknown, although guesstimates on the order of 0.01% have been published (Pauly *et al.*, 2002; Roberts, 2003). Consequently, formal calls for better information on MPAs have been issued (CBD, 2004a). Unfortunately, systematic collection of baseline data such as this is now widely perceived as a 'filling-in' activity rather than an innovative branch of marine science (Longhurst, 1998). Yet, it is the critical foundation to our understanding of the extent, distribution, and level of regulation of human activities in the world's oceans, and our ability to monitor progress towards targets such as those under discussion here. The global targets are also ambitious: based on previously available information, they imply an 8- to 3000-fold increase in area protected, in less than a decade.

The process of increasing the global marine area protected is fundamentally a decision-making problem. Decision-making occurs at and amongst multiple, nested levels, ranging from individual to group to organisation to society. Over time, decision-making at any, or all, of these levels has become increasingly complex and multidimensional, due to increases in the range and availability of goods and services, and increasingly variable extent and magnitude of impacts of any one alternative on other stakeholders and/or use/non-use options for a given resource (Kleindorfer *et al.*, 1993). Early decision theory described problems with a unique, determinate solution as 'benign', and problems with no determinate solution – instead, having only a negotiated and temporarily stable balance that will always be open to renegotiation and adjustment as context and power relations change in society – as 'wicked' (Minteer and Manning, 2003). The decision over what places to protect in the world's oceans appears to be inherently wicked: measurement of marine ecosystems is practically difficult, making spatial representation of species, habitats, and ecological processes, as well as the location, intensity and impacts of human activities, a major challenge. Variability in the marine environment through space and time further confounds our ability to understand and manage the system effectively (Longhurst, 1998; Norse and Crowder, 2005). There are also many socio-political objectives for use of marine systems, including fisheries extraction, biodiversity conservation, oil and gas exploration and extraction, substrate (e.g., sand and gravel) extraction, shipping, mining, tourism and recreation, and cultural uses (Duxbury and Duxbury, 1989; Norse and Crowder, 2005). The development of a global network of MPAs imposes opportunity costs on these other objectives, to varying degrees, and as such is a multicriteria decision problem (Moffett and Sarkar, 2006). Multicriteria decision-making (MCDM) provides a means to evaluate the trade-offs between different alternative solutions to a decision-making problem, on the basis of often conflicting criteria, and without necessarily requiring that all parameters under consideration be placed on the same scale (as is the case with cost-benefit analysis) (Carver, 1991; Faith and Walker, 1996; Malczewski, 1999).

In light of the multicriteria nature of the problem of creating a global network of MPAs, as well as the limited resources available to implement it (Wilson *et al.*, 2006), it is clear that not all areas of biological interest can be protected (Sarkar *et al.*, 2006). As a consequence, a science of selecting and designing protected area networks has been developing over the last 25 years (to a large extent from the theory of island biogeography), to identify, using place prioritisation algorithms, maximally spatially economical ways to achieve representation and persistence of biodiversity, within specified constraints (e.g., total area, surrogate representation targets) (Kirkpatrick, 1983; Kingsland, 2002; Sarkar *et al.*, 2006). This emerging field of research is now known as systematic conservation planning (Margules and Pressey, 2000; Sarkar *et al.*, 2006). Although these techniques take socio-political and economic considerations into account to some degree, for example, the cost criterion implemented in MARXAN (Stewart and Possingham, 2005), it is in a largely ad hoc manner, and so these methods do not represent formal MCDM techniques. Indeed, formal MCDM techniques have either never, or very rarely, been used in conservation planning (Moffett and Sarkar, 2006).

It was suggested in the early 1990s that the most appropriate strategy for systematic conservation planning is to first develop global priority areas, then, in a subsequent process, design regional or national-level networks (Vane-Wright *et al.*, 1991; Pressey *et al.*, 1993). However, at that time, computational capacity was a significant limiting factor to the maximum size of tractable problems, so that although global prioritisation analyses were possible using, for example, WORLDMAP, this was only achievable with relatively small data matrices, for example, spatial resolution of 10° latitude/longitude, i.e., 864 cells, and using only 43 species (Vane-Wright *et al.*, 1991; Williams, 1991). Similarly, appropriate data on the global spatial distributions of surrogates for marine diversity were largely unavailable. As a consequence, although a global marine prioritisation exercise has already been undertaken (Kelleher *et al.*, 1995a), it did not use systematic conservation planning methods, and did not attempt to ensure representation of known biodiversity. Rather, it identified a number of regional and national priority areas using expert knowledge on a range of criteria: biogeography; ecology; naturalness; economic importance; social importance; international or national significance; and practicality or feasibility.

Since the work of Kelleher *et al* (1995a), three new global marine protection targets have been adopted, and systematic conservation planning techniques and computational capacity have improved dramatically. In addition, global distribution data for marine species have been developed, for example, through the *Sea Around Us* Project (Close *et al.*, 2006; Kaschner *et al.*, 2006; Sea Around Us, 2007). However, until this thesis, they have not been used together to identify global marine protection priorities. In light of the ambitiousness and imminence of the target deadlines, therefore, this thesis provides very timely assessments of the current level of protection in the world's oceans and the attainability of the global marine protection targets. It also identifies global priority areas for protection in the world's oceans, based on the targets, which may be used as a strategic framework to guide subsequent stages of the planning process to develop global networks of MPAs.

Research Objectives

The research I present in this thesis is framed within the context of two main objectives. The first objective is to develop a robust baseline for the world's MPAs that enables progress towards the global targets to be monitored. The second is to identify priority areas for protection in the world's oceans, based on the quantitative requirements imposed by the global targets. To address these problems, I posed the following questions:

1. What is the current global extent and distribution of the world's marine protected areas?
2. What is the current growth rate of global marine area protected and is it sufficient for the global targets to be met on time?
3. Where in the world's oceans should be prioritised for protection, according to the spatial distribution of known biodiversity, and as specified by the global targets?

Thesis Outline

In total there are seven chapters in this thesis, consisting of five research chapters, opened with a general introductory chapter (Chapter 1). The next three chapters (Chapters 2, 3, and 4) relate to the first research objective, to develop a robust global baseline of MPAs. I approached this by extracting MPA data already available in the World Database on Protected Areas, restructuring it and creating a new database, called MPA Global. The database is available online (Wood, 2007) and allows for registered users to review and edit existing data. The database also has field-level referencing to enable greater transparency of source data. Updating of the data is an ongoing process, but to date, over 200,000 edits have been made using over 1100 sources, and the current list of MPAs has changed by around 75% from the original dataset. In Chapter 2, I provide a global analysis of MPAs based on the new MPA data, including an assessment of the attainability of the global targets. In Chapter 3, I explore some of the conceptual and philosophical challenges associated with global monitoring of a human construct such as MPAs, which became apparent during the process of updating the global database. In Chapter 4, I assess the extent to which the formulation of the global targets may affect their attainability. I investigated the second thesis objective, to identify global priority areas for marine protection in two different ways. I present these two approaches in Chapters 5 and 6. In Chapter 5, I use a species-richness based approach within multicriteria evaluation (MCE) to investigate the variability in priority areas identified depending on how two resource use objectives, biodiversity conservation and fisheries profit maximisation, are weighted. The study area for this analysis was the Pacific Canadian Exclusive Economic Zone (EEZ). In Chapter 6, I use a place prioritisation algorithm to identify priority areas in the global oceans, based on distribution range maps for 923 fish and invertebrate species, developed by the *Sea Around Us* Project (Close *et al.*, 2006) and 115 marine mammals (Kaschner *et al.*, 2006). Five scenarios that reflect the various global marine protection targets were investigated, and each of these was investigated using (a) all species distributions, and (b) fish and invertebrate species distributions only, in order to assess the influence of the wide-ranging annual movements of many marine mammal (Kaschner *et al.*, 2006) species on the size and spatial configuration of the solutions. Finally, in Chapter 7, I provide a synthesis of the findings presented in this thesis, their limitations, as well as some recommendations for future work.

References

Alder, J. (1996) Have tropical marine protected areas worked? An initial anaysis of their success. *Coastal Management*, **24**(2), 97-114.

Alongi, D.M. (2002) Present state and future of the world's mangrove forests. *Environmental Conservation*, **29**, 331-349.

Baum, J.K., Myers, R.A., Kehler, D.G., Worm, B., Harley, S.J. & Doherty, P. (2003) Collapse and conservation of shark populations in the northwest Atlantic. *Science*, **299**, 389-392.

Bellwood, D.R., Hughes, T.R., Folke, C. & Nystrom, M. (2004) Confronting the coral reef crisis. *Nature*, **429**, 827-833.

Berkes, F., Hughes, T.P., Steneck, R.S., Wilson, J.A., Bellwood, D.R., Crona, B., Folke, C., Gunderson, L.H., Leslie, H.M., Norberg, J., Nyström, M., Olsson, P., Österblom, H., Scheffer, M. & Worm, B. (2006) Globalization, roving bandits, and marine resources. *Science*, **311**, 1557-1558.

Botsford, L.W., Castilla, J.C. & Peterson, C.H. (1997) The management of fisheries and marine ecosystems. *Science*, **277**, 509-515.

Browman, H.I. & Stergiou, K.I. (2004) Perspectives on ecosystem-based approaches to the management of marine resources. *Marine Ecology Progress Series*, **274**, 269-303.

Carlton, J.T. (1993) Neoextinctions of marine invertebrates. *American Zoologist*, **33**, 499-509.

Carver, S. (1991) Integrating multi-criteria evaluation with geographical information systems. *International Journal of Geographical Information Systems*, **5**(3), 321-339.

Casey, J.M. & Myers, R.A. (1998) Near extinction of a large, widely distributed fish. *Science*, **281**, 690-692.

CBD (2004a) *Decisions adopted by the Conference of the Parties to the Convention on Biological Diversity at its seventh meeting (Decision VII/30)*. Convention on Biological Diversity, Montreal, Canada.

CBD (2004b) *Management of risks to the biodiversity of seamounts and cold water coral communities beyond national jutisdiction. Note by the Executive Secretary*. Convention on Biological Diversity, Kuala Lumpur, Malaysia.

CBD (2006) *Decisions adopted by the Conference of the Parties to the Convention on Biological Diversity at its eighth meeting (Decision VIII/15, Annex IV)*. Convention on Biological Diversity, Curitiba, Brazil.

Chape, S., Harrison, J., Spalding, M. & Lysenko, I. (2005) Measuring the extent and effectiveness of protected areas as an indicator for meeting global biodiversity targets.

Philosophical Transactions of the Royal Society of London. Series B: Biological Sciences, **360**, 443-455.

Christensen, V., Guénette, S., Heymans, J.J., Walters, C., Watson, R., Zeller, D. & Pauly, D. (2003) Hundred-year decline of North Atlantic predatory fisheries. *Fish and Fisheries*, **4**, 1-24.

Clark, M. & O'Driscoll, R. (2001) Deepwater fisheries and aspects of their impact on seamount habitat in New Zealand. *Journal of Northwest Atlantic Fishery Science.*, **31**, 441-458.

Close, C., Cheung, W.L., Hodgson, S., Lam, V., Watson, R. & Pauly, D. (2006) Distribution ranges of commercial fishes and invertebrates. In *Fishes in databases and ecosystems, Fisheries Centre Research Reports 14(4)* (eds M.L.D. Palomares, K.I. Stergiou & D. Pauly), pp. 27-37. University of British Columbia, Vancouver, Canada.

Côté, I.M., Gill, J.A., Gardner, T.A. & Watkinson, A.R. (2005) Measuring coral-reef decline through meta-analyses. *Philosophical Transactions of the Royal Society: Biological Sciences*, **360**(1454), 385-395.

Davies, A.J., Roberts, J.M. & Hall-Spencer, J. (in press) Preserving deep-sea natural heritage: Emerging issues in offshore conservation and management. *Biological Conservation*.

Devine, J.A., Baker, K.D. & Haedrich, R.L. (2006) Deep-sea fishes qualify as endangered. *Nature*, **439**(7072), 29.

Duarte, C.M. (2002) The future of seagrass meadows. *Environmental Conservation*, **29**, 192-206.

Dulvy, N.K., Sadovy, Y.J. & Reynolds, J.D. (2003) Extinction vulnerability in marine populations. *Fish and Fisheries*, **4**, 25-64.

Duxbury, A.C. & Duxbury, A.B. (1989) *An introduction to the word's oceans*. William C. Brown, Dubuque, Iowa, USA.

Faith, D.P. & Walker, P.A. (1996) Integrating conservation and development: effective trade-offs between biodiversity and cost in the selection of protected areas. *Biodiversity and Conservation*, **5**(4), 431-446.

FAO (2006) *The state of world fisheries and aquaculture*. Food and Agriculture Organisation of the United Nations, Rome, Italy.

Fortes, M.D. (1988) Mangrove and seagrass beds of east Asia: habitats under stress. *Ambio*, **17**(3), 207-213.

Gerber, L.R., Botsford, L.W., Hastings, A., Possingham, H.P., Gaines, S.D., Palumbi, S.R. & Andelman, S. (2003) Population models for marine reserve design: A retrospective and prospective synthesis. *Ecological Applications*, **13**(1), S47-S64.

Graham, N.A.J., Wilson, S.K., Jennings, S., Polunin, N.V.C., Bijoux, J.P. & Robinson, J. (2006) Dynamic fragility of coral reef ecosystems. *Proceedings of the National Academy of Sciences*, **103**(22), 8425-8429.

Gray, J.S. (1997) Marine biodiversity: patterns, threats and conservation needs. *Biodiversity and Conservation*, **6**, 153-175.

Hutchings, J.A. & Reynolds, J.D. (2004) Marine fish population collapses: consequences for recovery and extinction risk. *Bioscience*, **54**, 297-309.

IUCN (2007a) *2007 IUCN Red List of Threatened Species*, IUCN Species Survival Commission. http://www.redlist.org/ [accessed 4th October 2007].

IUCN (2007b) *Global marine species assessment*, IUCN. http://www.iucn.org/themes/ssc/biodiversity_assessments/indexgmsa.htm [accessed 5th October 2007].

Jackson, J.B.C. (2001) What was natural in the coastal oceans? *Proceedings of the National Academy of Sciences*, **98**(10), 5411-5418.

Jennings, S. & Kaiser, M.J. (1998) The effects of fishing on marine ecosystems. *Advances in Marine Biology*, **34**, 201-352.

Jones, P.J.S. (1994) A review and analysis of the objectives of marine nature reserves. *Ocean and Coastal Management*, **24**(3), 149-178.

Jones, P.J.S. (2002) Marine protected area strategies: issues, divergences and the search for middle ground. *Reviews in Fish Biology and Fisheries*, **11**(3), 197-216.

Kaschner, K., Watson, R., Trites, A.W. & Pauly, D. (2006) Mapping world-wide distributions of marine mammal species using a relative environmental suitability (RES) model. *Marine Ecology Progress Series*, **316**, 285-310.

Kelleher, G., Bleakey, C. & Wells, S., (eds.) (1995) *A global representative system of marine protected areas.* The Great Barrier Reef Marine Park Authority, The World Bank, The World Conservation Union, Washington DC, USA.

Kingsland, S.E. (2002) Creating a science of nature reserve design: perspectives from history. *Environmental Modeling and Assessment*, **7**(2), 61-69.

Kirkpatrick, J.B. (1983) An iterative model for establishing priorities for the selection of nature reserves: an example from Tasmania. *Biological Conservation*, **25**(2), 127-134.

Kleindorfer, P.R., Kunreuther, H.C. & Schoemaker, P.J.H. (1993) *Decision sciences: an integrative perspective*. Cambridge University Press, Cambridge, UK.

Koslow, J.A., Boehlert, G.W., Gordon, J.D., Haedrich, R.L., Lorance, P. & Parin, N. (2000) Continental slope and deep-sea fisheries: implications for a fragile ecosystem. *ICES Journal of Marine Science*, **57**(3), 548-557.

Koslow, J.A., Gowlett-Holmes, K., Lowry, J.K., O'Hara, T., Poore, G.C.B. & Williams, A. (2001) Seamount benthic macrofauna off southern Tasmania: community structure and impacts of trawling. *Marine Ecology Progress Series*, **213**, 111-125.

Longhurst, A. (1998) *Ecological geography of the sea*. Oval Press, London, UK.

Lubchenco, J., Palumbi, S.R., Gaines, S.D. & Andelman, S.J. (2003) Plugging a hole in the ocean: the emerging science of marine reserves. *Ecological Applications*, **13**(1), S3-S7.

Mace, G.M. (2004) In defence of fisheries scientists, single-species models and other scapegoats: confronting the real problems. *Marine Ecology Progress Series*, **274**, 285-291.

Malczewski, J. (1999) *GIS and Multicriteria Decision Analysis*. John Wiley & Sons, Inc., New York, USA.

Margules, C.R. & Pressey, R.L. (2000) Systematic conservation planning. *Nature*, **405**, 243-253.

Minteer, B.A. & Manning, R.E., (eds.) (2003) *Reconstructing conservation : finding common ground*. Island Press, Washington DC, USA.

Moffett, A. & Sarkar, S. (2006) Incorporating multiple criteria into the design of conservation area networks: a minireview with recommendations. *Diversity & Distributions*, **12**(2), 125-137.

Morato, T., Watson, R., Pitcher, T. & Pauly, D. (2006) Fishing down the deep. *Fish and Fisheries*, **7**, 24-34.

Myers, R.A. & Ottensmeyer, C.A. (2005) Extinction risk in marine species. In *Marine Conservation Biology* (eds E.A. Norse & L.B. Crowder), pp. 58-79. Island Press, Washington DC, USA.

Myers, R.A. & Worm, B. (2003) Rapid worldwide depletion of predatory fish communities. *Nature*, **423**(6937), 280-283.

Myers, R.A. & Worm, B. (2005) Extinction, survival or recovery of large predatory fishes. *Philosophical Transactions of the Royal Society of London. Series B: Biological Sciences*, **360**(1453), 13-20.

Norse, E.A. & Crowder, L.B., (eds.) (2005) *Marine Conservation Biology*. Island Press, Washington DC, USA.

Pauly, D. & Christensen, V. (1995) Primary production required to sustain global fisheries. *Nature*, **374**, 255-257.

Pauly, D., Christensen, V., Guénette, S., Pitcher, T., Rashid Sumaila, U., Walters, C., Watson, R. & Zeller, D. (2002) Towards sustainability in world fisheries. *Nature*, **418**, 689-695.

Pikitch, E.K., Santora, C., Babcock, E.A., Bakun, A., Bonfil, R., Conover, D.O., Dayton, P., Doukakis, P., Fluharty, D., Heneman, B., Houde, E.D., Link, J., Livingston, P.A., Mangel, M., McAllister, M.K., Pope, J. & Sainsbury, K.J. (2004) Ecosystem-based fishery management. *Science*, **305**, 346-347.

Preikshot, D. & Pauly, D. (2005) Global fisheries and marine conservation: is coexistence possible? In *Marine conservation biology: the science of maintaining the sea's biodiversity* (eds E.A. Norse & L.B. Crowder), pp. 185-197. Island Press, Washington DC, USA.

Pressey, R.L., Humphries, C.J., Margules, C.R., Vane-Wright, R.I. & Williams, P.H. (1993) Beyond opportunism: key principles for systematic reserve selection. *Trends in Ecology and Evolution*, **8**(4), 124-128.

Roberts, C.M. (2003) Our shifting perspectives on the oceans. *Oryx*, **37**(2), 166-177.

Roberts, C.M. & Hawkins, J.P. (1999) Extinction risk in the sea. *Trends in Ecology and Evolution*, **14**(6), 241-246.

Roberts, C.M., McClean, C.J., Veron, J.E.N., Hawkins, J.P., Allen, G.R., McAllister, D.E., Mittermeier, C.G., Schueler, F.W., Spalding, M., Wells, F., Vynne, C. & Werner, T.B. (2002) Marine biodiversity hotspots and conservation priorities for tropical reefs. *Science*, **295**(5558), 1280-1284.

Roberts, S. & Hirshfield, M. (2004) Deep-sea corals: out of sight, but no longer out of mind. *Frontiers in Ecology and the Environment*, **2**(3), 123-130.

Sadovy, Y. (2001) The threat of fishing to highly fecund fishes. *Journal of Fish Biology*, **59**(Suppl. a), 90-108.

Sarkar, S., Pressey, R.L., Faith, D.P., Margules, C.R., Fuller, T., Stoms, D.M., Moffett, A., Wilson, K.A., Williams, K.J., Williams, P.H. & Andelman, S. (2006) Biodiversity conservation planning tools: Present status and challenges for the future. *Annual Review of Environment and Resources*, **31**, 123-159.

Sea Around Us (2007) *A global database on marine fisheries and ecosystems*, Fisheries Centre, University British Columbia, Vancouver (British Columbia, Canada). http://www.seaaroundus.org [accessed 5th September 2007].

Stewart, R. & Possingham, H. (2005) Efficiency, costs and trade-offs in marine reserve system design. *Environmental Modeling and Assessment*, **10**(3), 203-213.

Sumaila, U.R., Guénette, S., Alder, J. & Chuenpagdee, R. (2000) Addressing ecosystem effects of fishing using marine protected areas. *ICES Journal of Marine Science*, **57**(3), 752-760.

United Nations (2002) *Report of the World Summit on Sustainable Development, Johannesburg, South Africa, 26 August-4th September 2002*. United Nations, Johannesburg, South Africa.

Valiela, I., Bowen, J.L. & York, J.K. (2001) Mangrove forests: one of the world's threatened major tropical environments. *Bioscience*, **51**(10), 807-815.

Vane-Wright, R.I., Humphries, C.J. & Williams, P.H. (1991) What to protect? - Systematics and the agony of choice. *Biological Conservation*, **55**, 235-254.

Wilkinson, C., (ed.) (2002) *Status of coral reefs of the world: 2002*. Australian Institute of Marine Science, Townsville, Queensland, Australia.

Williams, P.H. (1991) The bumble bees of the Kashmir Himalaya (Hymenoptera: Apidae, Bombini). *Bulletin of the British Museum (Natural History, Entomology)*, **60**(1), 1-204.

Wilson, K.A., McBride, M.F., Bode, M. & Possingham, H.P. (2006) Prioritizing global conservation efforts. *Nature*, **440**(7082), 337-340.

Wood, L.J. (2007) *MPA Global: a database of the world's marine protected areas*, Sea Around Us Project, UNEP-WCMC & WWF. www.mpaglobal.org [accessed 5th July 2007].

2. Assessing progress towards global marine protection targets: shortfalls in information and action[1]

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Introduction

Marine protected areas (MPAs) are increasingly viewed as an important spatial management tool within a suite of policy alternatives to reduce, prevent and/or reverse, ongoing (and in some cases rapid) declines in marine biodiversity and fisheries (Agardy, 1994; Pauly *et al.*, 2002; Hoyt, 2005; Roberts *et al.*, 2005). This has led to their inclusion in several recent global marine protection targets. The 2002 Plan of Implementation of the World Summit on Sustainable Development (WSSD) committed to establishing a representative global network of MPAs by 2012 (United Nations, 2002b, Section IV, paragraph 32(c)). At the Vth World Parks Congress (WPC) in 2003, the recommendation was made to "[g]reatly increase the marine and coastal area managed in marine protected areas by 2012; these networks should include strictly protected areas that amount to at least 20-30% of each habitat". Most recently, at the Eighth Ordinary Conference of the Parties (COP8) to the Convention on Biological Diversity (CBD) in 2006, a target that "at least 10% of each of the world's ecological regions [including marine and coastal be] effectively conserved [by 2010]" was adopted (CBD, 2006a). However, these targets were adopted with very limited prior knowledge of the existing global MPA network (the most recent global assessment of MPAs is over 10 years old and had data limitations (Kelleher *et al.*, 1995a)). Similarly, the targets were adopted without any assessment of the feasibility of the targets.

The World Database on Protected Areas (WDPA), maintained by the United Nations Environment Programme - World Conservation Monitoring Centre (UNEP-WCMC) is a global source of MPA data that has been widely used for monitoring MPAs, such as the United Nations List of Protected Areas. However, its coverage of MPAs has significant limitations (CBD, 2003), permitting only relatively broad-scale analyses on the total number and area of

MPAs (see, e.g. Chape *et al.*, 2005). More complete information on individual MPAs has been largely unavailable. Consequently, formal calls for better information on MPAs have been issued (CBD, 2004a). In response, a collaboration was established between the *Sea Around Us* Project at the University of British Columbia, Canada, World Wildlife Fund, and UNEP-WCMC, to extensively revise and update the MPA data in the WDPA.

The objective of this study was to collect data to enable more effective monitoring of MPAs that relate to four stated requirements of the three global targets: i) their distribution and coverage; ii) their network characteristics, as defined by available information on larval dispersal distances, iii) the representativeness of the global MPA network, and iv) growth of the network over time. I present a global review of the current status of the world's MPAs, with explicit reference to the three global targets, as well as a preliminary quantitative assessment of the feasibility of meeting the targets. I discuss these results, their implications and their limitations, and the role of large-scale targets in advancing marine conservation.

Methods

Database

Spatial and descriptive data were extracted from the WDPA (version 6.2) for all sites that were listed as marine. This includes MPAs that have been designated using statutory and non-statutory mechanisms operating at a range of scales, including individual MPA agreements, customary or traditional mechanisms, state/provincial legislation, national legislation, and international conventions. It also includes MPAs of variable designation status including designated, proposed, and degazetted. These data were restructured and used to create a new database, called MPA Global. Some new fields were added, including marine area (portion of the total area which is below the mean high water mark), no-take area (portion of the marine area where extraction of resources – both living and, where information allowed, non-living – is prohibited), and regulatory information. The database is available online (Wood, 2007). Registered users can view, review, and submit edits to the database. Field-level referencing was built into the online editing process, to increase the transparency of the database as well as document discrepancies between source materials.

The criterion used for inclusion of an MPA in MPA Global is based on the IUCN definition:

"Any area of intertidal or subtidal terrain, together with its overlying water and associated flora, fauna, historical and cultural features, which has been reserved by law or other effective means to protect part or all of the enclosed environment" (IUCN, 1988).

This definition was applied by reviewing the legal boundary of the site. If it extended seaward of the mean high water mark, the site was left in, or added to, MPA Global. For sites designated under non-statutory mechanisms, or where the designating legislation did not specify the legal seaward boundary, eligibility for inclusion was assessed using multiple sources (see below). Protected areas in the Caspian Sea are not included on the grounds that they are saline but not marine. Sites whose only 'marine' area was lagoonal were included only if the lagoon has a permanent surface connection to the sea. A globally extensive (although not yet fully exhaustive), multi-pronged, site-level update and verification process was undertaken. MPAs whose boundaries appeared to fall completely inland, using the 1:3,000,000 countries coastline shapefile provided in the ESRI Data and Maps Media Kit 2003 (ESRI, 2003), were identified in a Geographic Information System (GIS) and individually assessed. Updates were made at regional, country, and sub-country levels, using multiple sources including: a range of existing MPA databases, legislation, websites, peer-reviewed and non-peer-reviewed literature, and direct communications with regional/in-country experts. Finally, stratified sampling was undertaken to verify the data for the largest sites.

To date, over 1100 sources have been used to perform over 200,000 edits, pertaining to all countries with MPAs. Almost 1000 non-qualifying MPAs have been removed, 1000 MPAs have been added from the WDPA but that were not previously listed in the WDPA as marine, and almost 900 new sites have been added. These updates represent a ~75% change to the original WDPA list of 'MPAs'. New spatial data (MPA boundary polygons) have been obtained for 1822 of 3061 MPAs with spatial boundary data.

MPA network coverage

Global MPA coverage was estimated for all MPAs designated up to 31st December 2006. Sites listed under international conventions (e.g., UNESCO World Heritage Convention 1972, RAMSAR Convention 1971) were excluded due to near-complete overlap with nationally designated sites. Sites whose status was not designated or informally designated, were excluded. It was considered more accurate to estimate global MPA coverage by summing marine area estimates obtained through the editing process rather than through spatial analysis, due to a lack of spatial boundary data for ~31% of MPAs, and knowledge that some of the boundary data are out of date and substantially under- or over-sized. Of the total MPA area estimate, 92% was obtained from verified sources, and 8% estimated. For MPAs with unknown marine area, their total area was prorated according to the median proportion of total area for MPAs with known marine area and matching broad habitat types (intertidal only; intertidal and subtidal; subtidal only). Where total area was also unknown, the area assigned to the MPA was the global median marine area (4.6 km^2). Double counting of area due to overlap between MPAs was eliminated by subtracting the area of sites identified throughout the verification process as overlapping. Some overlap may remain, but this is negligible relative to the total area.

Information on no-take status was collected for MPAs on two levels: a qualitative status (all/part/none of the MPA is no-take) and a quantitative areal estimate where available. No-take data are currently available for 65% of the total MPA area. Total global no-take area was estimated by summing the areas stored in the attribute data; no overlap is known to exist between sites for which no-take data are available.

MPA network characteristics

I assessed the 'connectedness' of MPAs globally in terms of recommendations for MPA size and inter-MPA spacing based on known marine larval dispersal distances. A size-frequency distribution was produced using marine area data to identify the number and combined area of the world's MPAs that are large enough to be self-seeding for short-dispersing species. Sizes assessed were: >3.14 km^2, 12.5-28.5 km^2 (Shanks *et al.*, 2003), and 10-100 km^2 (Halpern and Warner, 2003). Recommended inter-MPA distances were used to create buffer 'bands' around MPAs in a GIS. Distances used were: 10-20 km (Shanks *et al.*, 2003) and 20-150 km (adapted from Palumbi (2003) and Cowen *et al* (2006)). MPAs occurring within these bands were considered to be 'connected' to at least one other MPA. These two analyses were combined in order to identify MPAs that meet both size and spacing requirements.

Assessing global MPA network representativeness.

Four measures of MPA network representativeness were investigated. Firstly, the distance of the central point of each MPA from the coast was estimated in GIS (ESRI, 2003), enabling both the frequency and area to be plotted as a function of distance from shore. Secondly, the same procedure was used to measure distance of MPAs from the Equator. The highly variable size and shape of individual MPA boundaries relative to the land and the equator mean that these distances may be an overestimate in some cases and an underestimate in others, but it represents a standard measure for all MPAs. Thirdly, the proportion of the following individual habitat types (for which a global distribution map is available) that is protected was estimated in GIS: estuaries (Alder, 2003); mangroves (UNEP-WCMC data); seagrass (UNEP-WCMC data); coral reef (UNEP-WCMC data); seamounts (Kitchingman and Lai, 2004). Finally, the proportion of two large scale political and/or broad marine habitat classifications that is currently protected was estimated in GIS: Large Marine Ecosystem (LME) (Sherman, 1991) and Exclusive Economic Zone (EEZ).

Global MPA network growth and predicting feasibility of target attainment.

Designation dates were available for MPAs constituting 98% of the total global area protected. The remaining 2% of the area was distributed across all years, prorated according to the proportion of the total global MPA area (in sites of known marine area) designated in that year. Known chronological changes in the size of individual MPAs were incorporated into the cumulative growth data. Simple linear regression of the logged cumulative global MPA area was used to estimate annual growth rate with which to predict target attainment dates.

Results

The current global network of MPAs

Extent of the world's MPAs

By 31st December 2006, around 4435 MPAs had been statutorily or non-statutorily designated at national or more local levels, covering approximately 2.35 million km^2, and occurring entirely within EEZs. This represents 0.65% of the world's oceans, or 1.6% of the total global marine area within EEZs. Only 12.8% of those 2.35 million km^2, representing 0.08% of the world's oceans and 0.20% of the global marine area under national jurisdiction, is subject to no-take regulations (i.e, is 'strictly protected' in the wording of the World Parks Congress Recommendation (IUCN, 2003)) (Figures 2.1 and 2.2). This is the first estimate of global no-take area that has been based directly on no-take data, and improves upon previous estimates which, due to a lack of such information, used IUCN management category as a proxy (e.g., Agardy *et al* (2003), and Jones (2006)).

Network characteristics of the world's MPAs

The mean size of MPAs is approximately 544 km^2, with a median size of 4.6 km^2. The substantial difference between mean and median MPA size is largely attributable to ten very large MPAs, which constitute 68% of the global MPA area (Table 2.1). Following size range suggestions derived from larval dispersal distances (Halpern and Warner, 2003; Shanks *et al.*, 2003), 79% of MPAs, representing 98.6% of total marine area protected, appear to be either too small or too large (Figure 2.3 & Table 2.2), particularly the latter, due to the 10 largest MPAs. However, if the size recommendations are viewed as minima, between 35% and 60% of MPAs, representing over 99% of the total area protected, are large enough (Table 2.3).

A total of 2496 MPAs (56.3% of the world's MPAs), covering 1.28 million km^2 (54.5% of the world's marine protected area) are 'connected' within 10-20 km of at least one other MPA. The vast majority of these (85% by number and 98% by area) are connected to a maximum of ten MPAs (Figure 2.4a). Using the larger connectedness distance of 20-150 km, 3487 MPAs (78.6% of the world's MPAs), covering 1.88 million km^2 (80.3% of the world's marine protected area) are connected to at least one other MPA, and, as expected, are generally connected to more MPAs than under the previous scenario (Figure 2.4b).

Combining the minimum size and spacing requirements indicates that at best, 49% of MPAs (80% by area), and, at worst, only 18% of MPAs (54% by area) could be considered as part of a connected network (Table 2.4).

Representativeness of the global MPA network

The distribution of the world's MPAs is distinctly non-uniform, being heavily biased towards both coastal waters and the 10 largest MPAs referred to earlier (Figure 2.5, Table 2.1). The number of MPAs declines exponentially with distance from shore, as does the distribution of area protected with distance from shore, with the exception of some of the 10 largest MPAs. However, the boundaries of all of these very large MPAs do abut the coast. As such, the measured distance of their centroid from shore is high simply by virtue of their large size.

The majority of the global marine area protected (approximately 65%, representing 43% of all MPAs) is located within the tropical latitude belt, between 30°N and 30°S, suggesting that tropical coastal habitats may in fact be among the best protected of all marine habitat types, at least on paper (Figure 2.6a). However, most of the remaining global marine area protected

(31%, representing 26% of all MPAs and including 5 of the world's ten largest MPAs) is located in latitudes higher than 50°, two-thirds of which is located in the northern hemisphere. These northern MPAs protect by far the highest proportion of sea surface area by latitude (Figure 2.6b). However, this may be largely attributable to the relatively small surface area of sea north of 50°N. Intermediate latitudes (30°-50°), and particularly southern temperate and polar latitudes, appear to be the least well protected.

Laurel and Bradbury (2006) suggest that based on larval dispersal distances, MPA size should increase with latitude. Using a subset of MPA Global, they conclude that this trend is not observed in the global network of MPAs. Using the complete dataset, I found a similar result. However, I also found that mean and median MPA size at latitudes greater than 50° is larger than the global values, and increases through the high latitude range (see Figure 2.6b and Table 2.5).

Proportional representation of habitat types within the global MPA network is shown in Figure 2.7. These are the only habitats for which global distributional data are known to be available, and mirrors the paucity of global data found for terrestrial habitats (Balmford *et al.*, 2003). The accuracy of the proportions protected varies with habitat type, due to variable (and largely unknown) accuracy of the habitat distributions themselves, both in terms of their precision as well as the confounding problems of habitat loss and change through time.

Figures 2.8 and 2.9 show the proportions of large marine ecosystem (LME) and exclusive economic zone (EEZ) that are currently protected, respectively. LMEs are suggested by the CBD as an appropriate classification system for monitoring progress towards its target (CBD, 2005). However, this is problematic for Pacific Island countries and territories, none of which occur within a LME, but all of which (with the exception of overseas territories of the United States of America) are party to the Convention. Given this, and the largely national scale of implementation of the CBD target, I view the proportion of EEZ as the best current assessment of the representativeness of the existing global MPA network, despite the political basis of the boundaries. It indicates that the current global MPA network falls far short of target requirements. Over 87% of 226 coastal countries (including 69 overseas territories and the non-contiguous US states of Hawaii and Alaska, listed separately) have less than the global average of 1.6% of their EEZs protected (Appendix 1). Of the nine countries that currently have more than 10% of their EEZs protected, four have relatively small maritime territories, rather than a high absolute area under protection. The remaining five are overseas territories (including the non-contiguous US state of Hawaii), that include four of the ten largest MPAs in the world.

Feasibility of attaining global MPA targets

Growth of the global MPA network

The cumulative area of the world's MPAs has grown steadily since the mid 1970s, coincident with the coming into force of various international conservation conventions (e.g., UNESCO Man and the Biosphere Program 1970, Ramsar Convention 1971, UNESCO World Heritage Convention 1972), and with some irregularities due to the creation of a few large MPAs (Figure 2.10, Table 2.1). Growth of no-take area is less steady, and has been very slow until recently, when the rezoning of the Great Barrier Reef Marine Park (GBRMP) in 2004 (Great Barrier Reef Marine Park Authority, 2004) increased the global no-take area by over 50% and 100,000 km^2 (Figure 2.10). More recently still, on 15th July 2006, the Northwestern Hawaiian

Islands Coral Reef Ecosystem Reserve (341,362km^2, originally designated in 2000) was redesignated as a Marine National Monument. Although it is not yet completely no-take, various habitat-damaging activities and all fishing is required to have ceased within 5 years (Establishment of the Northwestern Hawaiian Islands Marine National Monument: a Proclamation by the President of the United States of America, 2006).

Simple linear regression of the log-transformed cumulative area of MPAs indicates a 4.6% annual increase between 1984 and 2006, $r^2 = 0.96$ (Figure 2.11). This timespan was selected as it represents a time of very steady growth and is representative of the recent political environment. As such it was considered an appropriate timeframe on which to base projections for target attainment. Subsequent to the designation of the majority of the GBRMP in 1984 (it was created through a series of extensions between 1978 and 1984 (GBRMPA, 2007)), seven of the 10 largest MPAs were designated, together covering 43% of the current global marine area protected and 67% of the combined area of the top 10 MPAs (Table 2.1). In spite of this substantial increase in area protected, the overall rate of global MPA growth has not shifted from what appears to be a very stable, but slow, trajectory.

<u>Projected attainment dates of MPA targets</u>

I extrapolated the 4.6% growth into the future to assess the attainability of the WPC and CBD targets. It was not possible to assess attainability of the WSSD target using this method as it does not state quantitative areal targets. Results indicate that even the most modest targets will not be met for at least several decades (Figure 2.11). Furthermore, the growth rates required to meet these targets on time are at least an order of magnitude greater than observed (Table 2.6). In other words, a marine area at least three times the combined size of the ten largest MPAs (i.e., ~4.5 million km^2) would have to be designated every year until and including 2010 for timely attainment of the CBD target. These projections do not impose any of the additional requirements stated in the targets, including 'strict protection', habitat representation, and management effectiveness.

Discussion

These results indicate that the current extent, distribution, sizing and spacing of MPAs globally is vastly inadequate, particularly for no-take MPAs, and especially in light of past, ongoing, and expected future impacts on the oceans. The coastal bias of existing MPAs may not be too serious a disadvantage, since the coastal shelves contribute most to the world's primary production, known marine biodiversity and fisheries productivity (Pauly *et al.*, 2002). However, other attributes of the existing MPA network may serve to reduce the 'effective' area and extent of the network. Between 20 and 46% of the global area protected occurs in small and isolated MPAs, which may thus not be effective at ensuring persistence of marine populations or form part of a coherent global network. At the other extreme, the majority of the total marine area protected globally is contained within a handful of extremely large MPAs. At least some very large MPAs are needed to protect highly migratory species such as large pelagic fish and marine mammals, as well as to offset the concentration of fishing effort outside them (Walters, 2000), particularly if (as is the current situation) fishing effort is high and not reduced in conjunction with the creation of MPAs (Pauly *et al.*, 2002; Worm *et al.*, 2003). However, the total marine area protected globally is currently so small that its concentration in a few MPAs means that much of the world's oceans are essentially unprotected. This configuration of the world's MPAs thus confers very low levels of representation of many marine habitats, as well as of various biophysical, geographical and political regions. All of these factors may limit the resilience of the global MPA network to many external threats, as well as anticipated spatial shifts in species, communities and hydrological features in response to climate change (Carr *et al.*, 2003; Perry *et al.*, 2005; Parmesan, 2006; Simmonds and Isaac, 2007).

In addition, the results presented here are also best case scenarios, representing only the areas of the world's oceans that are protected on paper. It should not be assumed that: i) the process that created these MPAs also provided mechanisms for regulating human activities in the marine area; ii) where regulatory mechanisms are in place, they are all being implemented; or that iii) they are implemented effectively. Indeed, in many, if not most, MPAs, the biodiversity and fisheries benefits that may accrue through protection are eroded or undermined by inadequate management resources (financial and human), poor compliance with regulations, and little- or un-managed external threats (Alder, 1996; McClanahan, 1999; Jameson *et al.*, 2002). The best available information on management effectiveness is currently from large-scale analyses that are either outdated or focused on a subset of the world's MPAs. These assessments indicate very low rates of effective management (Kelleher *et al.*, 1995a; Alder, 1996; Mora *et al.*, 2006).

These results imply almost certain failure, at the very least in terms of attainment of global marine protection targets. Despite the designation of the 184,700 km^2 Phoenix Islands Protected Area (PIPA) by the Government of Kiribati, a huge individual achievement, at least 76 more countries each need to create MPAs covering an area equivalent to PIPA before 2010 for the CBD target to be met on time. Unfortunately, I suspect that the negative connotations of these predictions may undermine the benefits and successes of positive results at smaller scales, such as that of Kiribati. These results do, however, demand that the question be asked (again): can large scale conservation targets do more harm than good?

The utility of broad scale conservation targets has been questioned on numerous occasions. Targets have historically been justified in terms of political expediency rather than ecological knowledge (Soulé and Sanjayan, 1998; Agardy *et al.*, 2003). Broad scale, uniform conservation targets may thus be inadequate for meeting biodiversity conservation objectives (Rodrigues *et al.*, 2004), and may ultimately weaken the political process to create protected areas if the expected benefits are not observed, particularly within the electoral timeframe. However, the terrestrial protected area network has developed over more than a century, with at least half of the area designated (Chape *et al.*, 2005) before quantitative global targets were first established in the early 1980s (Soulé and Sanjayan, 1998). Similarly, the first explicitly marine, quantitative global protection target was made in 2003 (IUCN, 2003), when over 95% of the current marine area protected had already been created. Therefore, the location and design of marine (and terrestrial) protected areas have, to date, been selected largely without explicit consideration of many of the recently formalised principles of ‘MPA (network) design theory’ (Lubchenco *et al.*, 2003; Roberts *et al.*, 2003a) or the application of systematic conservation planning tools which have developed over the last 25 years (Kirkpatrick, 1983; Kingsland, 2002). While it is important to understand the (in)adequacy of existing protected areas in meeting specific objectives to inform future conservation planning, it may be counterproductive (and perhaps irrelevant) politically to criticise the products of past processes in terms of current ones.

Perhaps a more pressing question is how to garner the political will required to motivate a rapid increase in marine protection, particularly in the face of wider policy concerns such as food security, human welfare, and health. In this regard, broad scale conservation targets can help mobilise support for, and schedule, conservation intervention in the face of limited resources, ongoing biodiversity losses, and inadequate protection (Margules and Pressey, 2000; Pressey *et al*., 2003). In particular, the CBD target demonstrates a commitment of the parties to the Convention (presently 188; Appendix 1) to translate their general obligations under the Convention into concrete action for conservation and sustainable use (Pauly and Watson, 2005). Nevertheless, given the mismatch between the resources available and the resources required to implement and monitor a global network of MPAs, it seems likely that the global MPA network developed by the time of the target deadlines will almost certainly be a compromise, between quantity (i.e., how closely the targets are met) and quality (i.e., how appropriately designed and effectively implemented the MPAs thus created are). Broad scale conservation targets are thus, perhaps, necessary but not sufficient for effective marine resource conservation and management.

The work presented here has substantially improved the global MPA baseline, and enhanced our ability to monitor various aspects of MPA targets. While the value of a 'list' of MPAs in terms of assessing the 'effective' level of protection has been questioned (Roff, 2005), it remains a fundamental prerequisite to any assessment of status or progress. This analysis has provided the first quantitative estimate of the rate of change needed for these targets to be met. While daunting, this new information arms decision-makers and conservation planners with a greater understanding of the magnitude of the task ahead, and the urgency with which they must tackle it.

Tables

Table 2.1 Total and marine areas of the ten largest MPAs globally.

Country	MPA name	Designation type	Year designated	Total area ('000 km^2)	Marine area ('000km^2)
Australia	Great Barrier Reef	Marine Park	1979	344.4	344.4
USA	Northwestern Hawaiian Islands	Coral Reef Ecosystem Reserve*	2000	341.4	341.4
Republic of Kiribati	Phoenix Islands	Protected Area	2006	184.7	184.7
Australia	Macquarie Island	Marine Park	1999	162.0	162.0
Ecuador	Galapagos	Marine Reserve	1996	133.0	133.0
Denmark	Greenland	National Park	1974	972.0	110.6
Colombia	Seaflower	Marine Protected Area	2005	65.1	65.0
Australia	Heard Island and McDonald Islands	Marine Reserve	2002	64.6	64.6
Russia	Komandorsky†	Zapovednik (Strictly Protected Nature Reserve)	1993	58.3	55.8
Russia	Wrangel Island†	Zapovednik (Strictly Protected Nature Reserve)	1976	54.7	46.7
Total:				2380.2	1,508.2

* This site was redesignated as a Marine National Monument in June 2006
† Total and marine areas for these sites include buffer zone areas.

Table 2.2 Proportion of the world's MPAs by number and area that are within the size recommendations made by a) Halpern and Warner (2003) and b) Shanks *et al* (2003).

Size recommendation (km^2)	% of MPAs	% of area
10 - 100[a]	21	1.4
12.5 - 28.5[b]	8	0.3

Table 2.3 Proportion of the world's MPAs by number and area that meet minimum size requirements made by a) Halpern and Warner (2003) and b) Shanks *et al* (2003).

Size recommendation (km^2)	% of MPAs	% of area
> 3.14 [b]	58	99.7
> 10 [a]	35	99.4
>12.5 [b]	33	99.4

Table 2.4 Summary of the percentage of the world's MPAs by number and area that meet both minimum size and inter-MPA distance recommendations made by a) Halpern and Warner (2003) and b) Shanks *et al* (2003), and c) Palumbi (2003).

Minimum size (km^2)	connected within 10-20km[b]		connected within 20-150km[c]	
	% of MPAs	% by area	% of MPAs	% by area
>3.14[b]	34.1	54.6	49.1	80.3
>10[a]	19.9	54.4	29.9	80.1
>12.5[b]	18.4	54.4	27.6	80.0

Table 2.5 Summary statistics for MPAs by number and area in high latitudes (>50° north and south)

Latitude	% of world ocean	% of world MPA area	Mean MPA size (km^2)	Median MPA size (km^2)	Number of MPAs	% of MPAs	# of top ten largest MPAs
World	100	100	544	5	4435	100	10
> 50°	33	31	699	4	1169	26	5
> 60°	21	17	1521	7	263	6	2
> 70°	11	14	7629	398	43	1	2

Table 2.6 Summary of the annual rates of increase in global MPA coverage required to meet various marine protection targets on time, both at the time the targets were made and also currently.

Target	Target start	Target deadline	MPA area (10^3 km^2)			Annual rate of increase in global MPA coverage required to meet target	
			At target start	End 2006	Target	At target start	As of end 2006
CBD 10% of EEZs	2006	2010	2,162	2,350	16,444	50.0	91.3
(CBD) 10% of world ocean*	2006	2010	2,162	2,350	36,106	75.6	148.6
WPC 20% of world ocean	2003	2012	2,086	2,350	72,212	48.3	98.4
WPC 30% of world ocean	2003	2012	2,086	2,350	108,318	55.1	115.1

* The CBD target does not explicitly include the high seas in its target, although states that the high seas should be urgently protected using international cooperation. The data presented here are based on an extension of the CBD 10% target to include the high seas as well, but should not be viewed as an official or adopted target of the CBD.

Figures

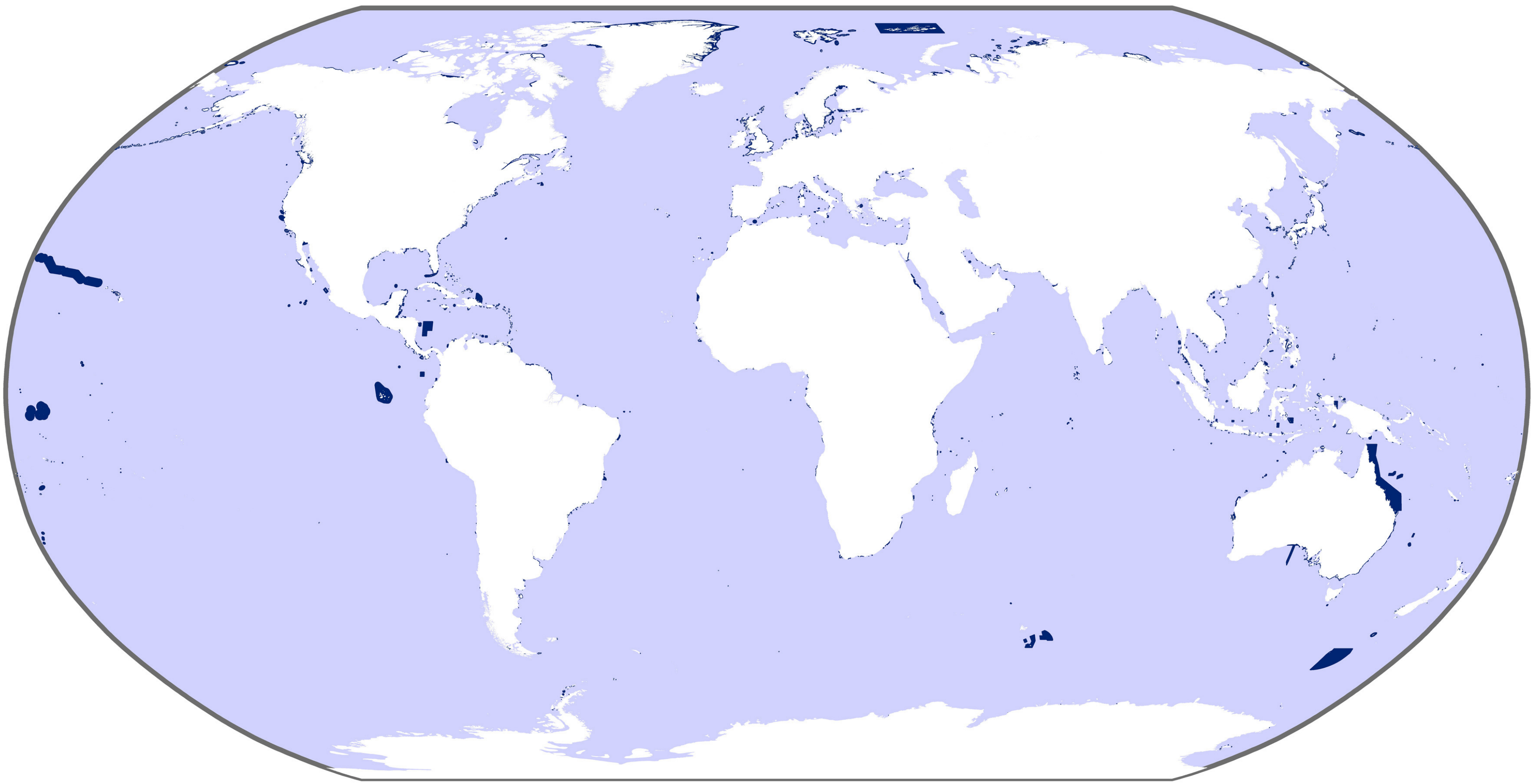

Figure 2.1 Global distribution of all MPAs designated by 31st December 2006.

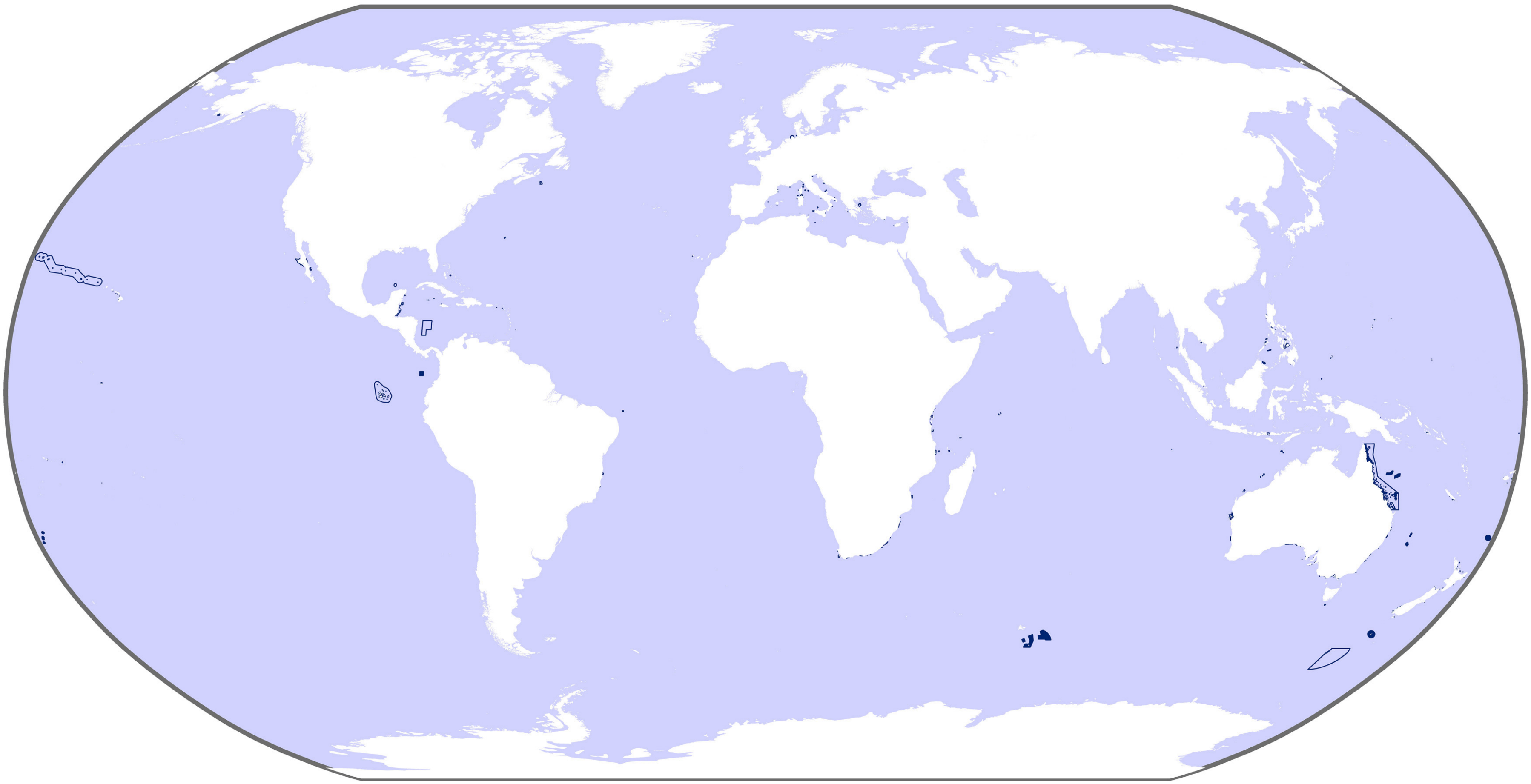

Figure 2.2 Global distribution of no-take MPAs designated by 31st December 2006. Solid areas are MPAs that are entirely no-take; open areas are MPAs which include one or more no-take zones.

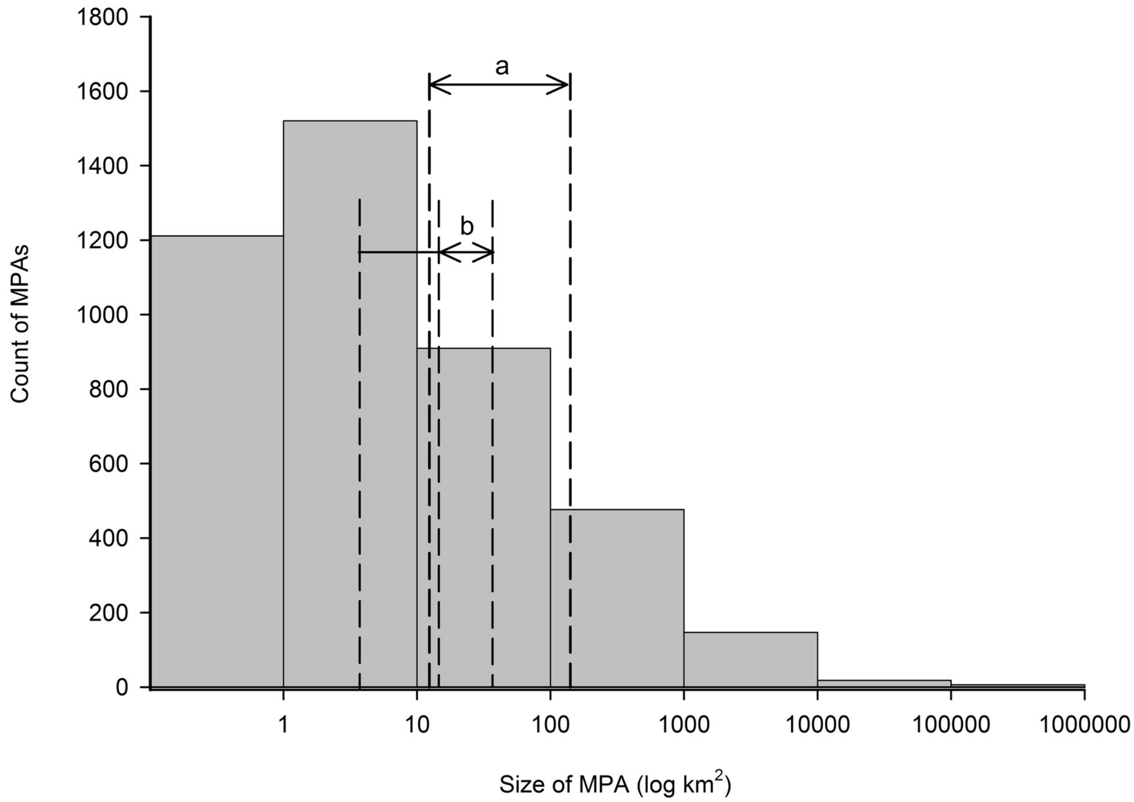

Figure 2.3 Area-frequency distribution of the world's MPAs, showing recommended MPA sizes using marine larval dispersal distances: a) 10-100km^2 (Halpern and Warner, 2003); b) minimum 3.14km^2, preferable 12.5-28.5km^2 (Shanks *et al.*, 2003).

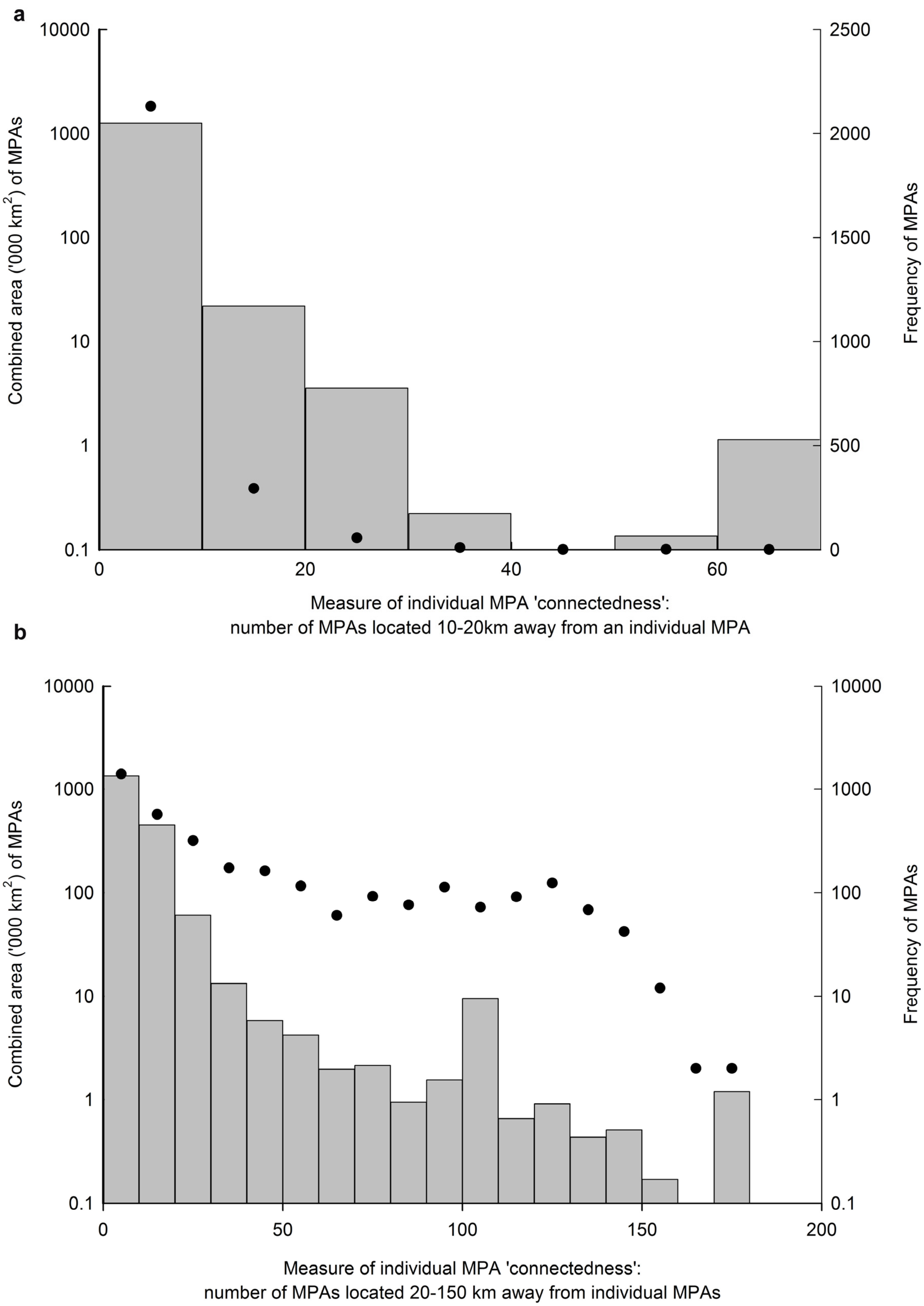

Figure 2.4 Frequency (black dots) and area (grey bars) of MPAs exhibiting variable individual levels of 'connectedness', as measured by the number of MPAs occurring a) 10-20km away from each MPA and b) 20-150km away from each MPA.

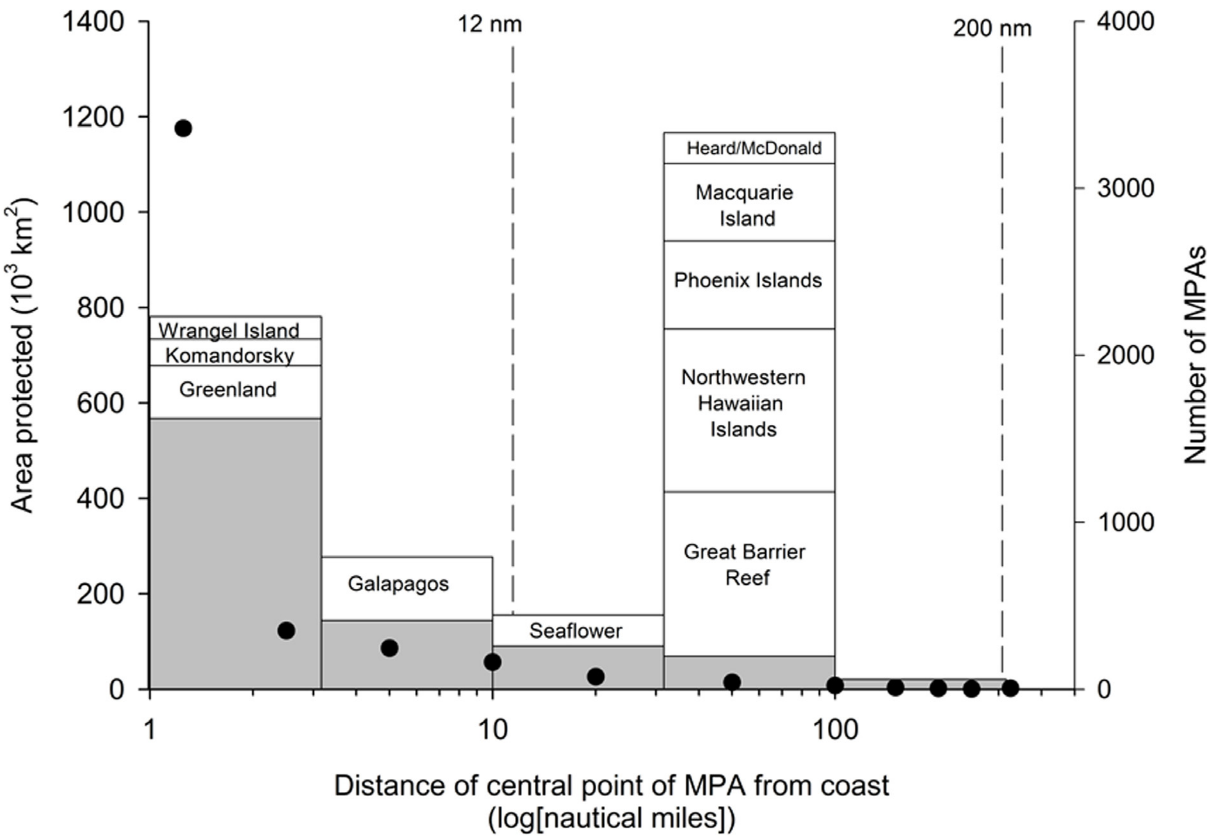

Figure 2.5 Marine area protected, as a function of distance from the coast. The world's ten largest MPAs are shown separately (see Table 2.1). The limits for territorial sea (12nm) and Exclusive Economic Zone (200nm) are indicated for clarity.

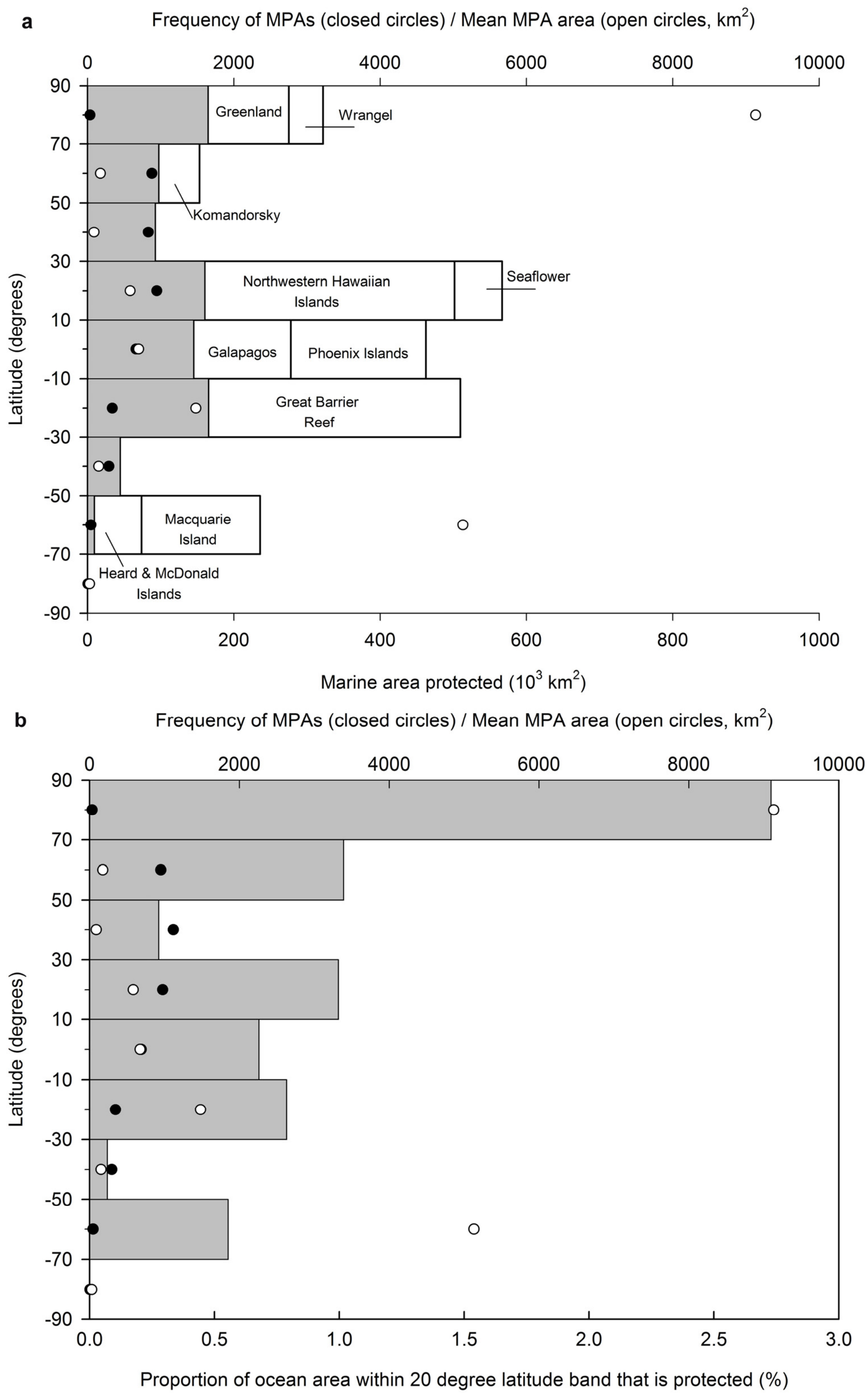

Figure 2.6 Distribution of MPAs (by number and area) as a function of distance from the Equator. Graph a) shows absolute area protected, with the world's ten largest MPAs shown separately (see Table 2.1). Graph b) shows the proportion of the sea area within 20 degree latitudinal bands that is protected.

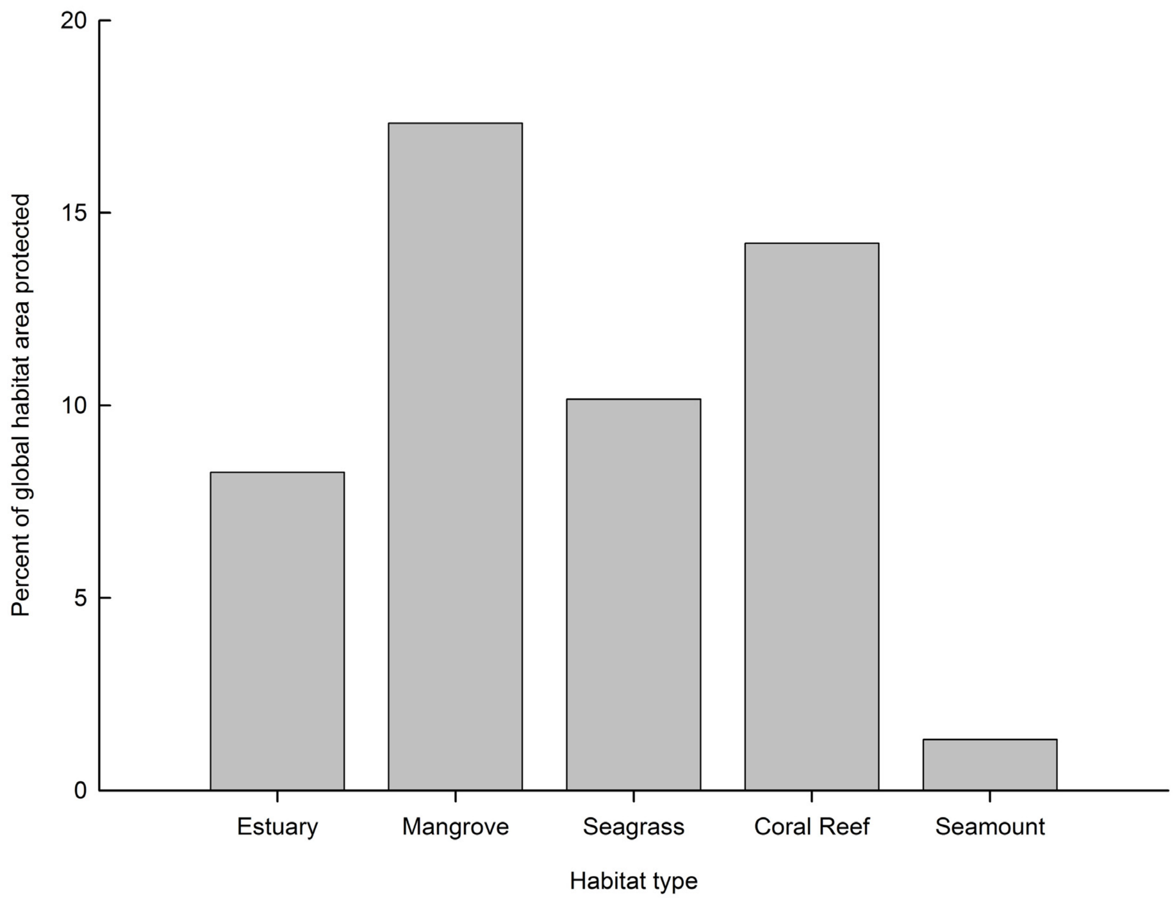

Figure 2.7 Estimated proportion of marine habitats protected within the current global MPA network, for habitat types where global distribution data are available.

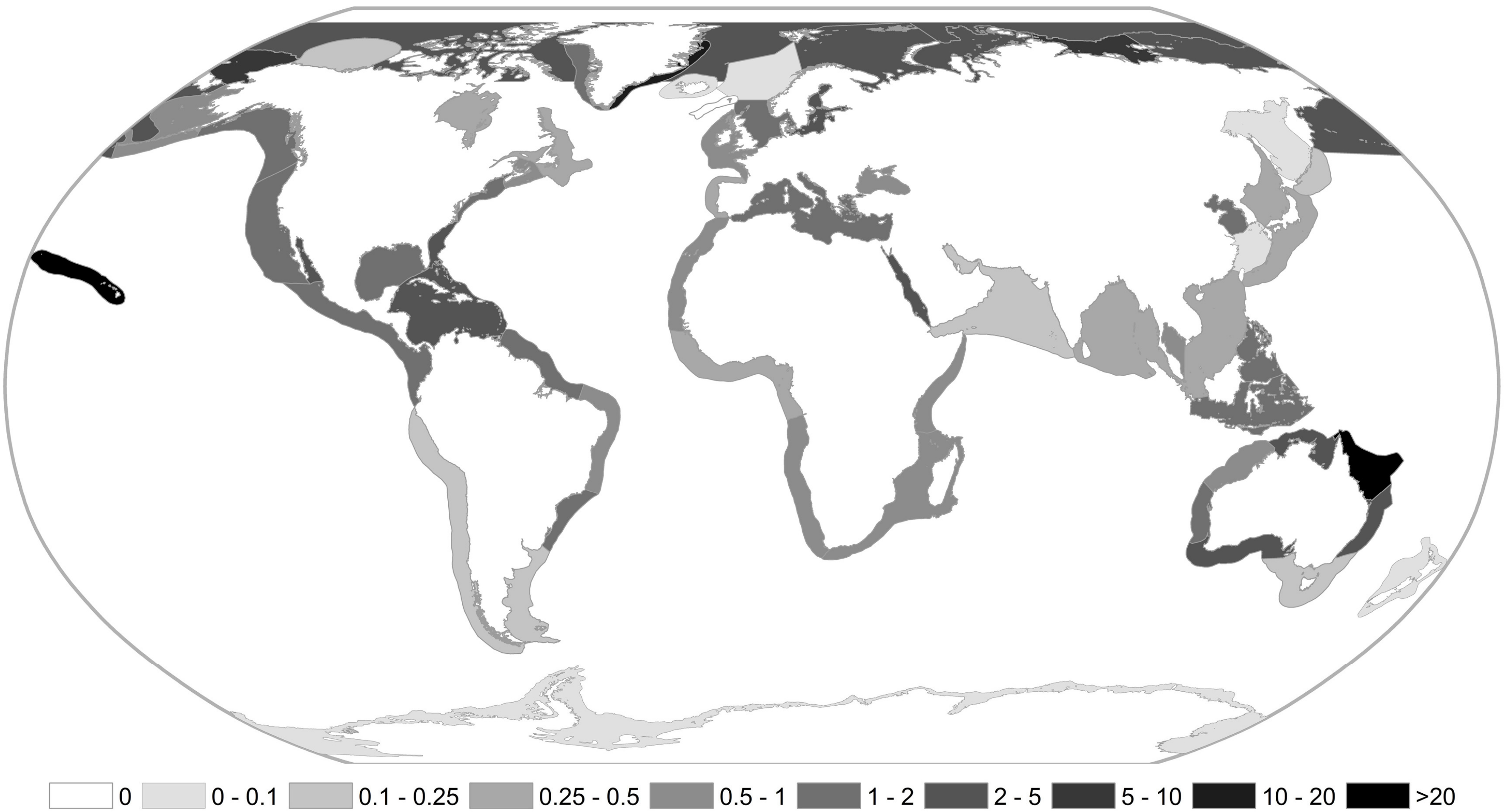

Figure 2.8 Percentage of Large Marine Ecosystem that is protected.

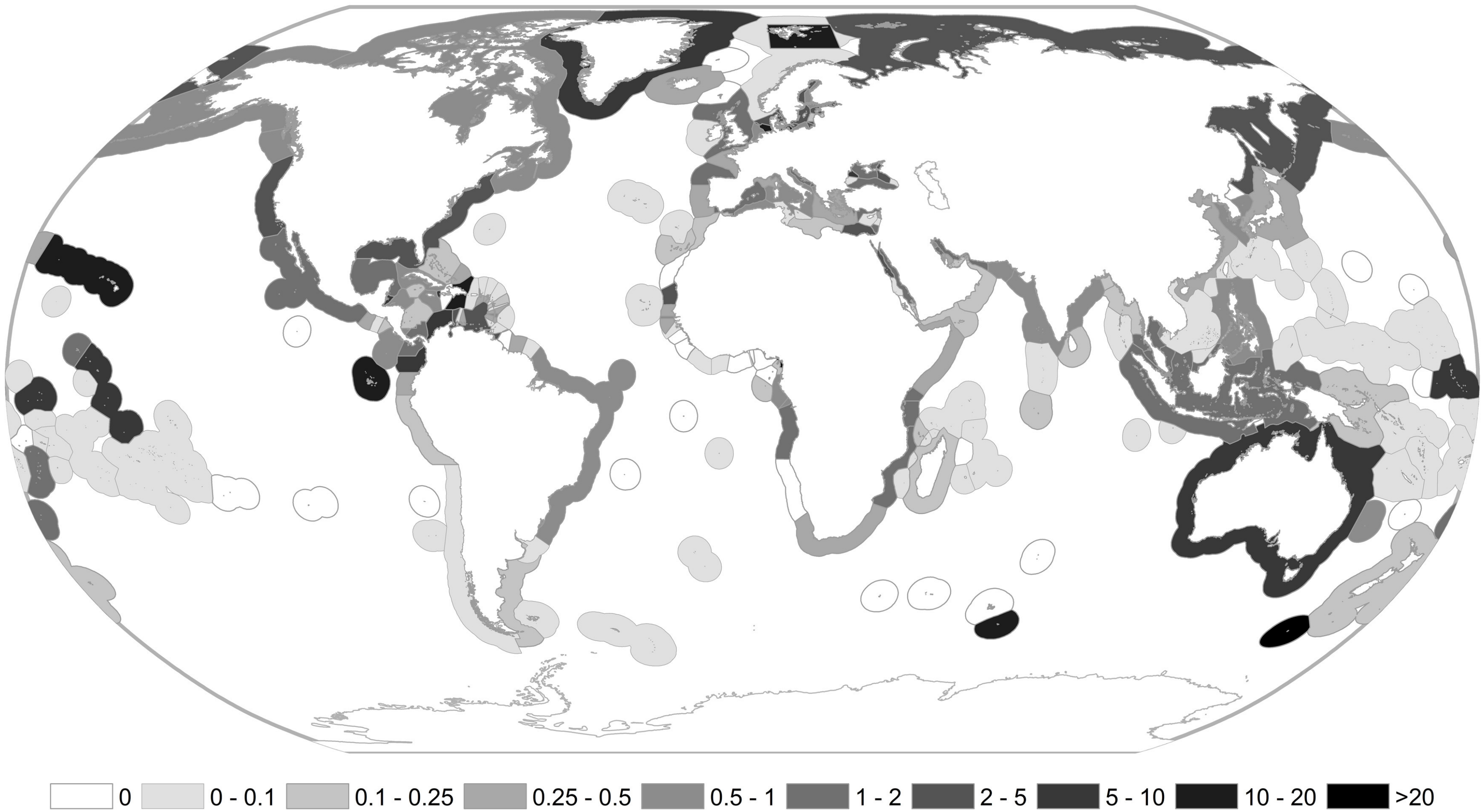

Figure 2.9 Percentage of 200nm Exclusive Economic Zones of maritime countries and territories that is protected. Approximate boundaries of the 200nm Exclusive Economic Zone (EEZ) boundaries are shown as grey lines.

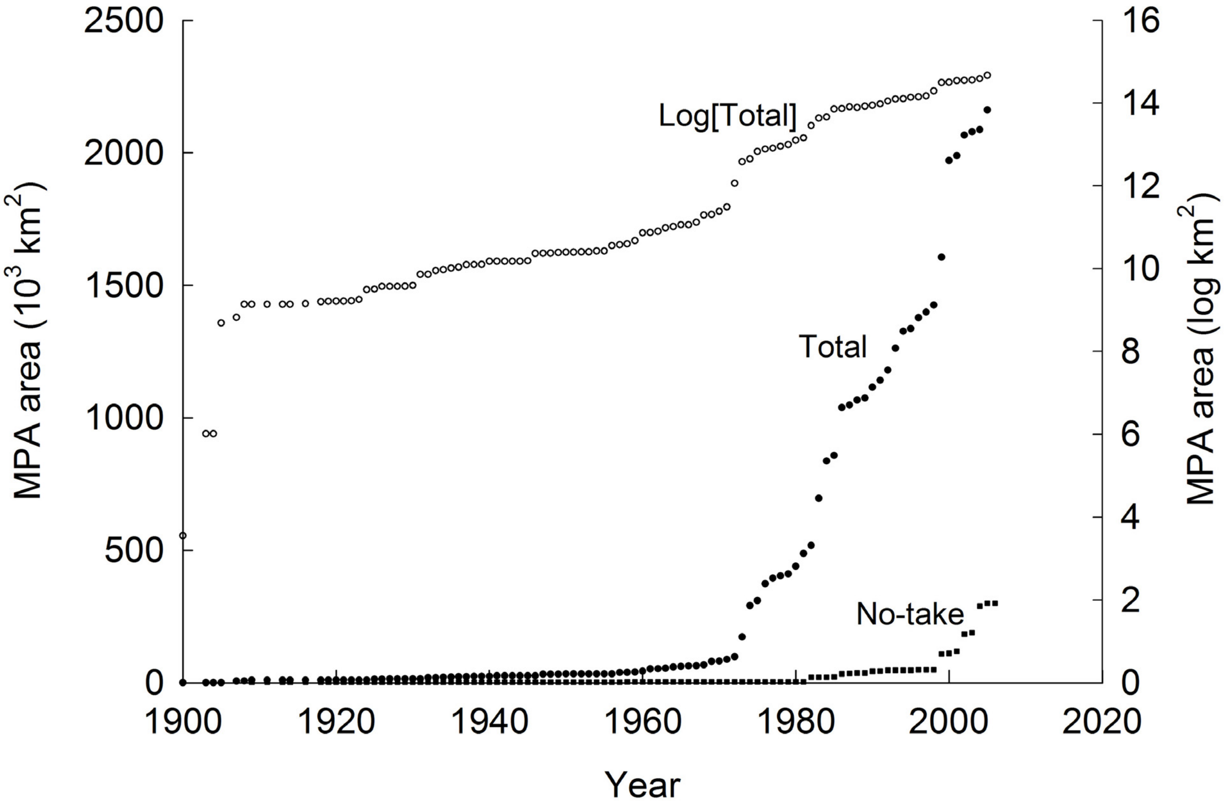

Figure 2.10 Growth in cumulative global marine area protected for: total (solid circles), log(total) (open circles) and no-take (squares) area.

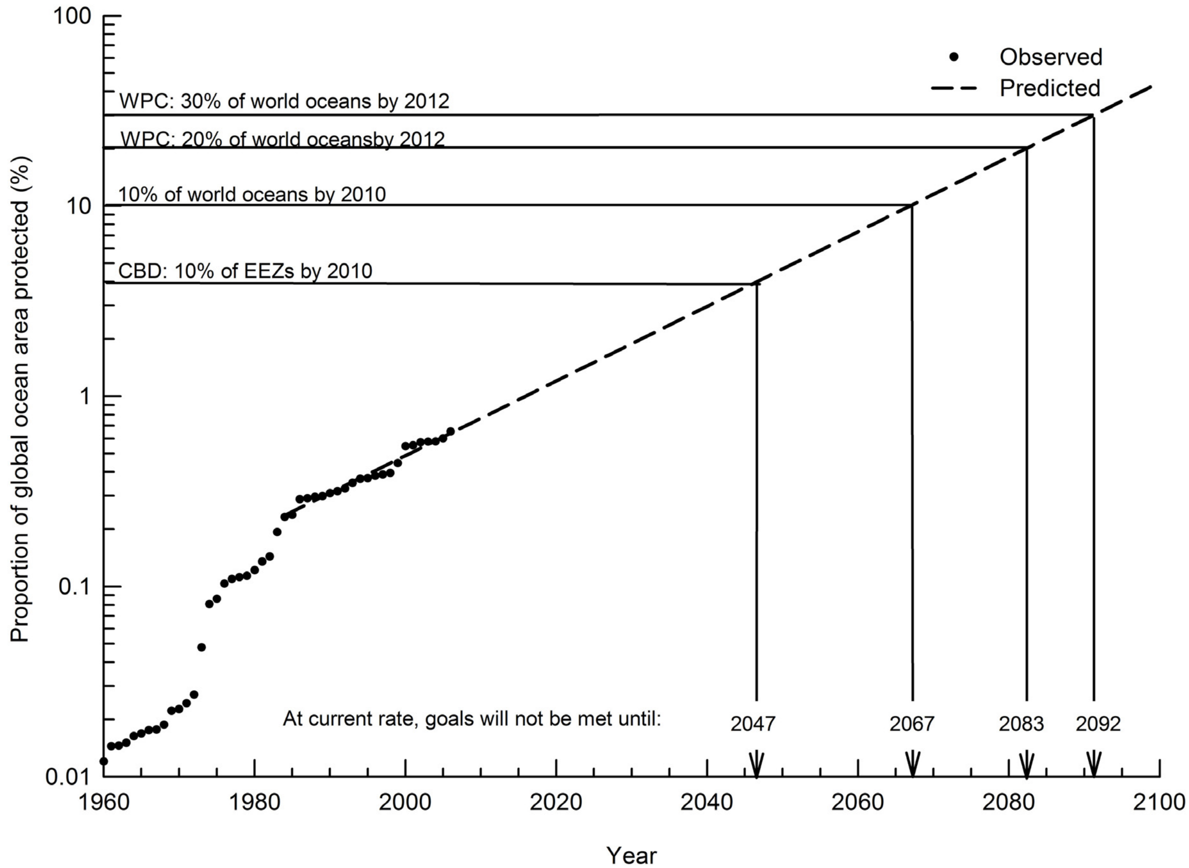

Figure 2.11 Projection of the annual rate of increase (4.6%, r^2 = 0.96) of global MPA area protected between 1984 and 2006 into the future, in relation to attainment of marine protection targets adopted by the Convention on Biological Diversity (CBD) and the World Parks Congress (WPC).

References

Agardy, T. (1994) Advances in marine conservation: the role of marine protected areas. *Trends in Ecology and Evolution*, **9**(7), 267-270.

Agardy, T., Bridgewater, P., Crosby, M.P., Day, J., Dayton, P.K., Kenchington, R., Laffoley, D., McConney, P., Murray, P.A., Parks, J.E. & Peau, L. (2003) Dangerous targets? Unresolved issues and ideological clashes around marine protected areas. *Aquatic Conservation: Marine and Freshwater Ecosystems*, **13**(4), 353-367.

Alder, J. (1996) Have tropical marine protected areas worked? An initial anaysis of their success. *Coastal Management*, **24**(2), 97-114.

Alder, J. (2003) Putting the coast in the Sea Around Us project. *Sea Around Us Project Newsletter*, **15**, 1-2.

Balmford, A., Green, R.E. & Jenkins, M. (2003) Measuring the changing state of nature. *Trends in Ecology & Evolution*, **18**(7), 326-330.

Carr, M.H., Neigel, J.E., Estes, J.A., Andelman, S., Warner, R.R. & Largier, J.L. (2003) Comparing marine and terrestrial ecosystems: Implications for the design of coastal marine reserves. *Ecological Applications*, **13**(1), S90-S107.

CBD (2003) *Marine and coastal biodiversity: review, further elaboration and refinement of the programme of work. Report of the Ad Hoc Technical Expert Group on Marine and Coastal Protected Areas.* Convention on Biological Diversity, Montreal, Canada.

CBD (2004) *Decisions adopted by the Conference of the Parties to the Convention on Biological Diversity at its seventh meeting (Decision VII/30)*. Convention on Biological Diversity, Montreal, Canada.

CBD (2005) *Draft global outcome-oriented targets for the programme of work on marine and coastal biological diversity*. Convention on Biological Diversity, Bangkok, Thailand.

CBD (2006) *Decisions adopted by the Conference of the Parties to the Convention on Biological Diversity at its eighth meeting (Decision VIII/15, Annex IV)*. Convention on Biological Diversity, Curitiba, Brazil.

Chape, S., Harrison, J., Spalding, M. & Lysenko, I. (2005) Measuring the extent and effectiveness of protected areas as an indicator for meeting global biodiversity targets. *Philosophical Transactions of the Royal Society of London. Series B: Biological Sciences*, **360**, 443-455.

Cowen, R.K., Paris, C.B. & Srinivasan, A. (2006) Scaling of connectivity in marine populations. *Science*, **311**, 522-527.

ESRI (2003) ESRI Data and Maps Media Kit.

Establishment of the Northwestern Hawaiian Islands Marine National Monument: a Proclamation by the President of the United States of America 2006 (USA). Presidential Proclamation 8031, http://www.whitehouse.gov [accessed 26th July 2006].

GBRMPA (2007) *The Great Barrier Reef Marine Park*, Great Barrier Reef Marine Park Authority. www.gbrmpa.gov.au [accessed 30th July 2007].

Great Barrier Reef Marine Park Authority (2004) *Area Statement for the Great Barrier Reef Marine Park*, Great Barrier Reef Marine Park Authority. http://www.gbrmpa.gov.au/corp_site/management/zoning/ [accessed 21st September 2005].

Halpern, B.S. & Warner, R.R. (2003) Matching marine reserve design to reserve objectives. *Proceedings of the Royal Society of London Series B-Biological Sciences*, **270**(1527), 1871-1878.

Hoyt, E. (2005) *Marine protected areas for whales, dolphins and porpoises*. Earthscan, London, UK.

IUCN (1988) *Resolution 17.38 of the 17th General Assembly of the IUCN*. IUCN, Gland, Switzerland and Cambridge, UK.

IUCN (2003) *WPC Recommendation 5.22: Building a global system of marine and coastal protected area networks*. IUCN, Durban, South Africa.

Jameson, S., Tupper, M.H. & Ridley, J.M. (2002) The three screen doors: *can* marine "protected" areas be effective? *Marine Pollution Bulletin*, **44**(11), 1177-1183.

Jones, P.J.S. (2006) Collective actions problems posed by no-take zones. *Marine Policy*, **30**(2), 143-156.

Kelleher, G., Bleakey, C. & Wells, S. (eds.) (1995) *A global representative system of marine protected areas.* The Great Barrier Reef Marine Park Authority, The World Bank, The World Conservation Union, Washington DC, USA.

Kingsland, S.E. (2002) Creating a science of nature reserve design: perspectives from history. *Environmental Modeling and Assessment*, **7**(2), 61-69.

Kirkpatrick, J.B. (1983) An iterative model for establishing priorities for the selection of nature reserves: an example from Tasmania. *Biological Conservation*, **25**(2), 127-134.

Kitchingman, A. & Lai, S. (2004) Inferences on potential seamount locations from mid-resolution bathymetric data. In *Seamounts: Biodiversity and Fisheries* (eds T. Morato & D. Pauly), Fisheries Centre, Vancouver, Canada.

Laurel, B.J. & Bradbury, I.R. (2006) 'Big' concerns with high latitude marine protected areas (MPAs): trends in connectivity and MPA size. *Canadian Journal of Fisheries and Aquatic Sciences*, **63**, 2603-2607.

Lubchenco, J., Palumbi, S.R., Gaines, S.D. & Andelman, S.J. (2003) Plugging a hole in the ocean: the emerging science of marine reserves. *Ecological Applications*, **13**(1), S3-S7.

Margules, C.R. & Pressey, R.L. (2000) Systematic conservation planning. *Nature*, **405**, 243-253.

McClanahan, T.R. (1999) Is there a future for coral reef parks in poor tropical countries? *Coral Reefs*, **18**, 321-325.

Mora, C., Andréfouët, S., Kranenburg, C., Rollo, A., Costello, M.J., Veron, J., Myers, R.A. & Gaston, K.J. (2006) Conservation of coral reefs by the global network of marine protected areas. *Science*, **312**, 1750-1751.

Palumbi, S.R. (2003) Population genetics, demographic connectivity, and the design of marine reserves. *Ecological Applications*, **13**(1), S146-S158.

Parmesan, C. (2006) Ecological and evolutionary responses to recent climate change. *Annual Review of Ecology and Systematics*, **37**, 637-669.

Pauly, D., Christensen, V., Guénette, S., Pitcher, T., Rashid Sumaila, U., Walters, C., Watson, R. & Zeller, D. (2002) Towards sustainability in world fisheries. *Nature*, **418**, 689-695.

Pauly, D. & Watson, R. (2005) Background and interpretation of the ‘Marine Trophic Index’ as a measure of biodiversity. *Philosophical Transactions of the Royal Society of London. Series B: Biological Sciences*, **360**(1454), 415-423.

Perry, A.L., Low, P.J., Ellis, J.R. & Reynolds, J.D. (2005) Climate change and distribution shifts in marine fishes. *Science*, **308**(5730), 1912-1915.

Pressey, R.L., Cowling, R.M. & Rouget, M. (2003) Formulating conservation targets for biodiversity pattern and process in the Cape Floristic Region, South Africa. *Biological Conservation*, **112**(1-2), 99-127.

Roberts, C.M., Andelman, S., Branch, G., Bustamante, R.H., Castilla, J.C., Dugan, J., Halpern, B.S., Lafferty, K.D., Leslie, H., Lubchenco, J., McArdle, D., Possingham, H.P., Ruckelshaus, M. & Warner, R.R. (2003) Ecological criteria for evaluating candidate sites for marine reserves. *Ecological Applications*, **13**(1), S199-S214.

Roberts, C.M., Hawkins, J.P. & Gell, F.R. (2005) The role of marine reserves in achieving sustainable fisheries. *Philosophical Transactions of the Royal Society: Biological Sciences*, **360**, 123-132.

Rodrigues, A.S.L., Andelman, S.J., Bakarr, M.I., Boitani, L., Brooks, T.M., Cowling, R.M., Fishpool, L.D.C., da Fonseca, G.A.B., Gaston, K.J., Hoffman, M., Long, J.S., Marquet, P.A., Pilgrim, J.D., Pressey, R.L., Schipper, J., Sechrest, W., Stuart, S.N., Underhill, L.G., Waller, R.W., Watts, M.E.J. & Yan, X. (2004) Effectiveness of the global protected area network in representing species diversity. *Nature*, **428**, 640-643.

Roff, J.C. (2005) Conservation of marine biodiversity: too much diversity, too little cooperation. *Aquatic Conservation: Marine and Freshwater Ecosystems*, **15**, 1-5.

Shanks, A.L., Grantham, B.A. & Carr, M. (2003) Propagule dispersal distance and the size and spacing of marine reserves. *Ecological Applications*, **13**(1), S159-S169.

Sherman, K. (1991) The large marine ecosystem concept: research and management strategy for living marine resources. *Ecological Applications*, **1**(4), 349-360.

Simmonds, M.P. & Isaac, S.J. (2007) The impacts of climate change on marine mammals: early signs of significant problems. *Oryx*, **41**(1), 19-26.

Soulé, M.E. & Sanjayan, M.A. (1998) Conservation targets: do they help? *Science*, **279**(5359), 2060-2061.

United Nations (2002) *Report of the World Summit on Sustainable Development, Johannesburg, South Africa, 26 August-4th September 2002*. United Nations, Johannesburg, South Africa.

Walters, C. (2000) Impacts of dispersal, ecological interactions, and fishing effort dynamics on efficacy of marine protected areas: how large should protected areas be? *Bulletin of Marine Science*, **66**(3), 745-757.

Wood, L.J. (2007) *MPA Global: a database of the world's marine protected areas*, Sea Around Us Project, UNEP-WCMC & WWF. www.mpaglobal.org [accessed 5th July 2007].

Worm, B., Lotze, H.K. & Myers, R.A. (2003) Predator diversity hotspots in the blue ocean. *Proceedings of the National Academy of Sciences*, **100**, 9884-9888.

3. Conceptual challenges to global monitoring of marine protected areas[1]

[1] A version of this chapter has been submitted for publication. Wood, L. J. Conceptual challenges to global monitoring of marine protected areas.

Introduction

Various targets recently adopted to substantially increase the level of marine protection globally (United Nations, 2002a; IUCN, 2003; CBD, 2006a) have heightened the need for effective monitoring of marine protected areas (MPAs). A fundamental premise of any monitoring program is that, in order to be monitored, the target subject of that program must be readily identified, and distinguishable from its surroundings. In other words, it needs to be defined. However, in the process of recent efforts by the author, in collaboration with others, to improve the information available on MPAs (Wood *et al.*, in press), it became apparent that the conceptual issues of what MPAs are, as well as if, how, and by whom any definition of MPA should be developed, are far from resolved. This imposes various practical limitations to monitoring MPAs at the global scale. The objective of this chapter is to explore these limitations, and consider the extent to which they affect our understanding of spatial and temporal patterns of human presence and activity in the oceans.

What is a definition and why define?

According to the Oxford English Dictionary, a definition is 'a statement of the meaning of a word or the nature of a thing'. To define something means, *inter alia,* 'to give the exact meaning of'; 'describe or explain the scope of'; or 'mark out the boundary or limits of' (Thompson, 1990). The purpose of defining thus seems to be to express concepts in words in a way that enables them to be understood by others who may be unfamiliar with them. Definitions also often undergo some degree of interpretation in order to become operational for a specific purpose and context (Mintzberg, 1992). To define or interpret is conscious, purposeful (i.e., goal-directed) human behaviour (Latham and Locke, 1991) . In the context of monitoring MPAs globally, the purpose of defining MPA is to provide a means by which to identify what to include or exclude in a global database of MPAs.

What is a 'marine protected area'?

This is not an entirely new question – see, for example, Agardy *et al* (2003) - and initially, it might also appear to be a trivial one. Cognitively, most, if not all, of us, would be able to imagine what an MPA might be, if asked. But would we be able to translate such a cognitive image into words? Over the past few decades, various attempts have been made to develop written definitions of MPA. Possibly the first, and certainly the most widely used and cited, definition is that adopted by the World Conservation Union (IUCN, 1988):

> "Any area of intertidal or subtidal terrain, together with its overlying water, and associated fauna, flora, historical, or cultural features, which has been reserved by law or other effective means to protect all or part of the enclosed environment."

Given its wide application and acceptance in the literature, I considered this definition to be the most appropriate to use to update the global MPA baseline (Wood *et al.*, in press). However, in trying to interpret this definition for global monitoring purposes, a profound conceptual problem arose: what does 'protect all or part' actually mean? What are we protecting the marine environment from? By how much? How much of the whole environment is enough to be considered part of it? In terms of scientific defensibility, one can simply avoid confronting these issues by stating assumptions and criteria explicitly. However, this has the unfortunate consequence of overlooking some critical conceptual debates.

Normatively, the phrase 'protect all or part' implies some sort of continuum of protection along which MPAs fall, based on the extent to which human activities within the area are restricted.

This notion is supported by the system developed by the IUCN to facilitate classification of protected areas. It consists of seven categories, which imply a gradation of human intervention from strict protection and preservation of species and genetic diversity, to sustainable use of natural ecosystems (IUCN, 1994). The concept of a continuum of protection is also indicated by available MPA data: globally there are some 350 different MPA designation types, which themselves reflect variation in regulations. Regulations known to apply in MPAs around the world range from complete exclusion of human access to near-zero regulation of human activities, most likely due to local circumstances and threats (Allison *et al.*, 1998; Agardy *et al.*, 2003; Wood, 2007). However, despite extensive consideration of the pluralism of MPAs, there has been scant discussion of the minimum level of regulation necessary for an area to be considered 'protected'. The sustainable use tenet of the least protected IUCN category doesn't clarify the situation, instead requiring further interpretation of 'sustainable use'. However, this minimum boundary is critical to developing a global database of MPAs. How can a database be built without knowing how to identify what to include or exclude? For example, the minimum requirement for inclusion in FishBase (Froese and Pauly, 2007) is that the organism be a fish, the definition of which has been subjected to over two centuries' worth of taxonomic classification work. No such body of work exists for MPAs.

Conceptualisation of 'protection'

Much of the difficulty in defining and interpreting terms such as 'protection' and 'sustainable use' arises from them being normative concepts. Unlike fish, MPAs are not tangible entities with established (if imperfect) methods of measurement and classification. MPAs are human constructs, the product of a complex suite of social, political, cultural, economic, and often legal, processes, which exist almost exclusively within the human mind. Displaying them on maps and marking their boundaries with buoys may lend some physical substance to MPAs, but for the most part they remain firmly within our psyches. As mental constructs, MPAs are goal-oriented, subjective, and fundamentally linked to individual values (Collet, 2006). The form that such values take is influenced by our philosophical attitude to the environment, which in turn affects our perceived responsibility to the environment, and therefore our conceptualisation of protection (Sarkar, 2005). Earlier formulations of environmental worldviews suggested a dichotomy between a resource conservation ethic and a preservation ethic (Callicott, 1991), borne of anthropocentric and biocentric philosophies, respectively (Sarkar, 2005). More recent thinking suggests a unified philosophy of conservation, with these worldviews constituting either extreme of a continuum (Callicott *et al.*, 1999). Indeed, Sarkar (2005) discusses a 'tempered anthropocentrism' which seeks to retain anthropocentric values of biodiversity without a loss of reverence for nature. Rather than being narrow and intransigent, therefore, worldviews experienced by individuals may span part or all of this continuum, depending on the sector in which they are trained and work, the scale at which they work, and the specific context of the problem at hand (Callicott *et al.*, 1999).

This presents various practical challenges to defining protection as a global concept. Firstly, the perceived need to develop a definition of protection is in itself value-laden and goal-oriented; in this case, that global targets for marine protection are appropriate and that progress towards them should be monitored at the global scale using a single definition of protection. The process of developing a definition is thus embedded within, and bounded by, the worldview(s) held by those developing the definition. Secondly, in order to develop a definition, individualistic and changeable cognitive images must be somehow extracted from a global constituency (which also needs to be defined), translated into words, and amalgamated into a single definition; this is an enormous process of social coordination. Coordination necessitates consensus-seeking through communication, the effectiveness of which is contingent upon the level of motivation within the global constituency to coordinate (Scheff, 1967), i.e., political will, as well as the diversity of worldviews held by both the constituency and the coordinators. Complete consensus occurs when there is an infinite series of reciprocating understandings between group members concerning the issue (i.e., I understand that he/she understands that I understand, and so on). Partial consensus occurs when this reciprocity breaks down. In general, more complex transactions, such as that under discussion here, require higher levels of consensus for social coordination to occur (Scheff, 1967). In planning, coordination is often synonymous with coercion (Mintzberg, 1992), particularly where political will for coordination is lacking. Consequently, there is potential for the worldviews held by those developing the definition to prevail in the final definition.

These challenges are particularly relevant in the context of marine protection, since spatial regulation of human activities has arisen in multiple sectors, notably fisheries management and biodiversity conservation. However, monitoring of MPAs has historically fallen to the conservation sector (hence the widespread use of the IUCN definition, even in the fisheries literature). Areas designated primarily for or by fisheries are seldom considered by conservationists as ‘protected areas’. They are seen, rather, as ‘area-based management tools’. The reasoning behind this linguistic dichotomy seems to be embedded within differences between actual and perceived environmental worldviews between the two sectors. In particular, much of the conservation community still leans towards preservationist worldviews (Callicott, 1991; Agardy *et al.*, 2003), and tacitly assumes that the fisheries sector leans towards resource conservationist worldviews, which inherently impose less strict regulation of human activity.

This sectoral divide is thus often considered by the conservation community to be a best approximation of the minimum boundary of protection for MPAs. However, environmental worldviews held by different sectors may be more similar than perceived (Harms and Sylvia, 2001), suggesting that the reciprocity of understanding necessary for complete consensus on a global definition of protection is lacking. Indeed, available information does not support a sectoral basis for defining the minimum boundary of protection. If the regulations applied in space are reflective of environmental worldviews, then there appears to be considerable overlap between the range of worldviews held by the two sectors (Wood, 2007). This sectoral basis for defining a minimum boundary of protection thus causes substantial consistency problems in developing a global database of MPAs. In addition, it is difficult to uphold, due to the often opportunistic use of existing legislation (e.g., fisheries laws) to create MPAs, including for biodiversity conservation objectives. Furthermore, the functional overlap between MPAs designated for different primary objectives is poorly understood and so the empirical basis of this dichotomy is very limited (Hastings and Botsford, 2003). Such a dichotomy also seems to be outdated in light of the growing emphasis on ecosystem- (i.e., space- rather than sector-) based management of the oceans (Caddy, 1999; Sumaila *et al.*, 2000; Agardy *et al.*, 2003; Agardy, 2005; Edwards, in press).

Reconciling concepts of protection for more effective global monitoring of the oceans

Fishing is currently the biggest threat to marine ecosystems, yet despite the long-running conflict between fisheries management and biodiversity conservation, their long-term goals are the same (Preikshot and Pauly, 2005). Global monitoring of MPAs using a conceptualisation of protection that excludes the fisheries sector thus seems to be the biggest shortcoming of the current system. Furthermore, the reasons for this exclusion seem to be driven by philosophical differences and misperceptions, rather than empirical ones. Explicit consideration of the conceptual and philosophical basis for these difficulties is useful in gaining a better understanding of current limitations to global monitoring efforts, but it doesn't necessarily get us any closer to solving the problem. Furthermore, given the depth and complexity of the problem, one must ask: is it even possible to achieve global consensus on individually-based, changeable philosophical views of the environment? However, given the political reality of global targets for protection, and mandates to monitor progress towards them, there remains a practical need to try to navigate these difficulties, and at least, to recognise and try to overcome some of the shortcomings of the current system. Perhaps the best solution might be to recognise that we may not be able to define a minimum boundary of protection, and embrace the concept of a continuum of protection from zero to absolute. Every location in the oceans is protected or exploited to some degree; isn't it logical to measure this on a single scale? This implies a radical transformation to a global database of all spatial restrictions on human activity in the oceans, the resource requirements of which are indeed vast. The purpose of this article is not to blindly suggest that we 'should just do this'. However, in line with Agardy *et al* (2003), we do need to think and talk more about trying to do this. Insufficient data and resources are a ubiquitous problem in conservation, but the consequences of not addressing these conceptual issues are far more profound in terms of our ability to understand, communicate about and manage our oceans.

References

Agardy, T. (2005) Global marine conservation policy versus site-level implementation: the mismatch of scale and its implications. *Marine Ecology Progress Series*, **300**, 242-248.

Agardy, T., Bridgewater, P., Crosby, M.P., Day, J., Dayton, P.K., Kenchington, R., Laffoley, D., McConney, P., Murray, P.A., Parks, J.E. & Peau, L. (2003) Dangerous targets? Unresolved issues and ideological clashes around marine protected areas. *Aquatic Conservation: Marine and Freshwater Ecosystems*, **13**(4), 353-367.

Allison, G.W., Lubchenco, J. & Carr, M. (1998) Marine reserves are necessary but not sufficient for marine conservation. *Ecological Applications*, **8**(1), S79-S92.

Caddy, J.F. (1999) Fisheries management in the twenty-first century: will new paradigms apply? *Reviews in Fish Biology and Fisheries*, **9**, 1-43.

Callicott, J.B. (1991) Conservation ethics and fishery management. *Fisheries*, **16**(2), 22-28.

Callicott, J.B., Crowder, L.B. & Mumford, K. (1999) Current normative concepts in conservation. *Conservation Biology*, **13**(1), 22-35.

CBD (2006) *Decisions adopted by the Conference of the Parties to the Convention on Biological Diversity at its eighth meeting (Decision VIII/15, Annex IV)*. Convention on Biological Diversity, Curitiba, Brazil.

Collet, S. (2006) Values at sea, value of the sea: mapping issues and divides. *Social Science Information*, **46**(1), 35-66.

Edwards, S. (in press) Ocean zoning, first possession and Coasean contracts. *Marine Policy*.

Froese, R. & Pauly, D. (2007) *FishBase*, www.fishbase.org [accessed 25th July 2007].

Harms, J. & Sylvia, G. (2001) A comparison of conservation perspectives between scientists, managers, and industry in the West Coast groundfish fishery. *Fisheries*, **26**(10), 6-15.

Hastings, A. & Botsford, L.W. (2003) Comparing designs of marine reserves for fisheries and for biodiversity. *Ecological Applications*, **13**(1), S65-S70.

IUCN (1988) *Resolution 17.38 of the 17th General Assembly of the IUCN*. IUCN, Gland, Switzerland and Cambridge, UK.

IUCN (1994) *Guidelines for protected area management categories*. IUCN & UNEP-WCMC, Gland, Switzerland and Cambridge, UK.

IUCN (2003) *WPC Recommendation 5.22: Building a global system of marine and coastal protected area networks*. IUCN, Durban, South Africa.

Latham, G.P. & Locke, E.A. (1991) Self-regulation through goal setting. *Organizational Behavior and Human Decision Processes*, **50**, 212-247.

Mintzberg, H. (1992) *The rise and fall of strategic planning: reconceiving roles for planning, plans, planners*. The Free Press, New York, USA.

Preikshot, D. & Pauly, D. (2005) Global fisheries and marine conservation: is coexistence possible? In *Marine conservation biology: the science of maintaining the sea's biodiversity* (eds E.A. Norse & L.B. Crowder), pp. 185-197. Island Press, Washington DC, USA.

Sarkar, S. (2005) *Biodiversity and environmental philosophy: an introduction*. Cambridge University Press, Cambridge, UK.

Scheff, T.J. (1967) Toward a sociological model of consensus. *American Sociological Review*, **32**(1), 32-46.

Sumaila, U.R., Guénette, S., Alder, J. & Chuenpagdee, R. (2000) Addressing ecosystem effects of fishing using marine protected areas. *ICES Journal of Marine Science*, **57**(3), 752-760.

Thompson, D. (ed.) (1990) *The Concise Oxford Dictionary of Current English, 9th edition*. Clarendon Press, Oxford, UK.

United Nations (2002) *Draft plan of implementation of the World Summit on Sustainable Development*. United Nations, Johannesburg, South Africa.

Wood, L.J. (2007) *MPA Global: a database of the world's marine protected areas*, Sea Around Us Project, UNEP-WCMC & WWF. www.mpaglobal.org [accessed 5th July 2007].

Wood, L.J., Fish, L., Laughren, J. & Pauly, D. (in press) Assessing progress towards global marine protection targets: shortfalls in information and action. *Oryx*.

4. Global marine protection targets: how S.M.A.R.T are they?[1]

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Introduction

Large-scale, typically percentage-based, conservation targets have been criticised for being ecologically irrelevant, particularly because they are rarely sufficient to ensure persistence of populations. They may also be political hindrances to further conservation efforts (Soulé and Sanjayan, 1998; Agardy *et al.*, 2003; Pressey *et al.*, 2003; Rodrigues *et al.*, 2004). As a consequence, some authors have suggested that 'policy-driven' or 'data-independent' conservation targets should be abandoned (Solomon *et al.*, 2003; Svancara *et al.*, 2005; Wiersma and Nudds, 2006). Worse still, however, is that even these apparently meagre and inadequate targets have rarely been attained. Large-scale targets are often considered to be over-ambitious, unattainable, and are thus frequently discredited and ignored (Roberts, 2005). Indeed, existing global marine protection targets seem extremely unlikely to be met on time, instead reaching only a fraction of the target by the deadlines (Wood *et al.*, in press). Thus, despite the many declarations, resolutions, conventions and protocols adopted since the United Nations Conference on the Human Environment in Stockholm, 1972, the overall state of the environment has continued to deteriorate (Elliott, 1998). Although large-scale conservation targets are widely considered to provide overarching frameworks for, and motivate, action at smaller scales (Pressey *et al.*, 2003), it seems that this rarely happens, which would appear to be a more immediate (and serious) problem than the question of their ecological relevance.

Goal-setting is a fundamental component of human behaviour: goals are expressions of values and needs, and motivate and direct behaviour and performance (Taylor, 1976; Latham and Locke, 1991; Locke and Latham, 2002), at both the individual and group levels (Erez, 2005; Latham and Pinder, 2005). As such, goals and targets have been incorporated into all scales of planning, in a range of sectors, from formal business plans and local systematic conservation planning efforts, to health services planning, to broader public policy and international regimes (Rondinelli, 1976; Mintzberg, 1992; Margules and Pressey, 2000; Broadhead, 2002; Pressey *et al.*, 2003; Thomas, 2003; United Nations, 2006). In broad terms, goals are generalised statements about an ideal state to be attained, and establish the tone for the planning process. Targets refer to specific outputs to be reached in support of goal attainment (Thomas, 2003).

Given that targets are integral to planning processes, then, it may be more appropriate in the case of large scale conservation targets to attempt to better understand the reasons for their a) general lack of uptake and subsequently b) ecological (ir)relevance, than to abandon them completely. In order to be effective, it is suggested that targets must be operationalisable, measurable, amenable to evaluation, and time-bound with clear deadlines (Thomas, 2003). These principles have been embodied in the SMART concept (Specific, Measurable, Achievable, Realistic, and Time-bound). The exact origin of the SMART concept is not known, although it has been applied in fields as varied as policy planning (HM Treasury, 2003), healthcare (van Herten and Gunning-Schepers, 2000; Busse and Wismar, 2002), financial management (Kawohl *et al.*, 2003), education (Muncey and McGinty, 1998), climate data management (Plummer *et al.*, 2005), global plant conservation (CBD, 2002), as well as the Millennium Development Goals (Roberts, 2005). Recently, it has been gaining traction in the marine conservation literature, particularly in relation to MPA objective and management effectiveness assessments (Jones, 2000; Manghubai, 2001; Day *et al.*, 2002). At a larger scale, the SMART concept has been incorporated into marine conservation and planning objectives, and assessments thereof, in, for example, Canada (Stark, 2004), the Irish Sea (Lumb *et al.*, 2004), Europe (Rice *et al.*, 2005), and South Australia (Department for Environment and Heritage, 2006), as well as in relation to the Convention on Biological Diversity indicators on sustainable use of biodiversity (Tucker, 2005). However, to date, the SMART concept has not been explicitly applied to global MPA targets.

The objective of this study is to assess three global marine protection targets using the SMART framework, and use this assessment in combination with a critical review of the current literature, and experiences in monitoring progress towards them (Wood *et al.*, in press) to attempt to better understand the challenges to their implementation. The targets to be assessed are: 1) the World Summit on Sustainable Development Plan of Implementation commitment to "the establishment of marine protected areas consistent with international law and based on scientific information, including representative networks by 2012" (United Nations, 2002b, Section IV, paragraph 32(c)); 2) Recommendation 5.22 made at the Vth World Parks Congress (2003) to "Establish by 2012 a global system of effectively managed, representative networks of marine and coastal protected areas….. these networks should be extensive and include strictly protected areas that amount to at least 20-30% of each habitat" (IUCN, 2003, p2); and 3) Decision VIII/15 adopted by the Conference of the Parties to the Convention on Biological Diversity at its eighth meeting, that "at least 10% of each of the world's ecological regions [including marine and coastal be] effectively conserved [by 2010]" (CBD, 2006a, p237).

Global marine protection targets: are they Specific?

Specific targets are clear and easy to understand, i.e., well defined, and are therefore more readily accepted by those implementing them (van Herten and Gunning-Schepers, 2000). Specific targets also help to direct behaviour towards a reduced number of potential outcomes, such that behaviour between actors is more consistent, and 'effective performance' is more evident and measurable (Latham and Locke, 1991).

All three targets appear to be quite specific: they require ‘MPAs’ to be created in ‘representative networks’ and be ‘effectively managed’. However, definitions of these concepts themselves are still widely debated, which has the potential to hinder their implementation. Some of the ambiguities surrounding the conceptual requirements imposed by the targets are summarized below.

What is an MPA?

The definition of an MPA varies in both the literature and in practice with such attributes as the objectives of the MPA; the activities regulated both within and outside it; and the effectiveness with which those regulations are implemented (Jameson *et al.*, 2002; Jones, 2002; Agardy *et al.*, 2003). However, perhaps the most critical concept to define in the context of the global targets is what the minimum level of regulation of human activities is required for that ocean space to be considered ‘protected’ – i.e., what is an MPA? What differentiates an MPA from ocean space adjacent to it? This is largely a philosophical question, based on individual conceptualisations of protection, and as such, it will always be open to debate (Wood, submitted).

What should MPAs be representative of?

Representativeness is a key component of systematic conservation planning, and refers to the inclusion of samples of all of biodiversity within protected areas (Pressey *et al.*, 1993; Sarkar *et al.*, 2006). However, biodiversity is itself very difficult to define (Sarkar, 2005). Incomplete knowledge of the spatial distributions of many aspects of biodiversity means that surrogates must be used in conservation planning exercises. Surrogates may operate at a range of spatial scales and consist of a range of biological or non-biological features. The choice of surrogate used to approximate and measure representativeness thus depends on the scale of the analysis as well as the available data for the area of interest (Margules and Pressey, 2000; Sarkar *et al.*, 2006).

What is a 'network' of MPAs?

MPA networks may be conceptualised in a multiplicity of ways. In ecological terms, they consist of MPAs that are connected oceanographically by larval dispersal and juvenile or adult migration (Ballantine, 1995; Gaines *et al.*, 2003; Lubchenco *et al.*, 2003; Palumbi, 2003; Norse and Crowder, 2005). However, as MPA boundaries are open to external influences, the design of MPA networks may also need to consider: the impacts of the MPA on the spatial redistribution of fishing effort and benefits of fishing (Walters, 2000; Sumaila and Armstrong, 2006); resilience of the MPAs (or network) over time to external factors and longer term catastrophes (Jameson *et al.*, 2002; Wagner *et al.*, 2006); the objectives of the MPA or network (Halpern and Warner, 2003; Hastings and Botsford, 2003); as well as the socio-cultural context of the area and the level of stakeholder support for the MPA or network (Sumaila *et al.*, 2000; Walmsley and White, 2003).

What is management effectiveness of MPAs?

MPA management effectiveness, rather than simple presence and size, is a more a meaningful measure of the 'actual' contribution made by MPAs to biodiversity conservation and other objectives because of the many threats to MPAs, including inadequate regulation of human activities, non-compliance, and a range of external threats (Boersma and Parrish, 1999; Jameson *et al.*, 2002). Management effectiveness consists of multiple components: design issues of protected areas and networks, appropriateness of management systems and processes, and delivery of protected area objectives (Hockings *et al.*, 2000). In addition, many methodologies for measuring MPA management effectiveness exist, ranging in application from global (Pomeroy *et al.*, 2004; Staub and Hatziolos, 2004) to regional (Wells and Mangubhai, 2004), to national or sub-national (Pollnac *et al.*, 2001), as well as MPA programme-specific, for example, World Heritage Sites (Hockings *et al.*, 2004).

These brief summaries point to considerable complexity surrounding various concepts embodied in the global marine targets. Under conditions of ambiguity such as this, there is a clear need for these concepts to be defined operationally, i.e., in practical terms that enable them to be implemented, and progress towards the targets to be monitored (Wood, submitted). However, the World Summit on Sustainable Development (WSSD) target does not provide any definition (conceptual or operational) of MPA, network, or representativeness, and it is unstated whether the WSSD target requires MPA networks to be effectively managed. The World Parks Congress (WPC) target is slightly more specific than the WSSD target in that it specifies 'marine habitats' as the features to be represented in MPA networks. However, it provides no definitions for MPA, strictly protected MPA, network, habitat, or effective management. The Convention on Biological Diversity (CBD) target is the most specific of the three targets. It provides a detailed definition of MPA:

" 'Marine and Coastal Protected Area' means any defined area within or adjacent to the marine environment, together with its overlying waters and associated flora, fauna, and historical and cultural features, which has been reserved by legislation or other effective means, including custom, with the effect that its marine and/or coastal biodiversity enjoys a higher level of protection than its surroundings" (CBD, 2003, p11).

Additional documentation expands the scope of this definition considerably:

"Other measures, such as fisheries management areas, well-functioning integrated marine and coastal area management regimes (which effectively manage land-based sources of marine pollution), prohibition of destructive practices (such as bottom trawling) may also contribute to effective protection" (CBD, 2005, p3).

Critically, however, these definitions do not specify what the minimum level of regulation of human activities is required for an area to be considered protected, and as such they are not immediately operational. While the CBD target does also specify the scale and type of biodiversity features that the representation requirement pertains to, it does not define network or management effectiveness.

Global marine protection targets: are they Measurable?

Measurable targets are easily appraised (van Herten and Gunning-Schepers, 2000), and thus allow feedback on progress to be provided in a timely manner. Progress that is reported publicly conveys the message that progress is valued, which in turn can increase motivation and commitment to achieving the targets, as well as increase public demand for further progress (Latham and Locke, 1991; Roberts, 2005). There are various aspects to measurability, which are discussed below, along with an assessment of the extent to which the three global marine targets meet them.

Targets must be quantitative

In order for a target to be measurable, it must be quantitative in some way. However, the WSSD target provides no quantification of the global representative networks of MPAs that it commits to achieve, and as such it is impossible to assess whether and/or when the target will be achieved. The WPC target is partially measurable, in that it requires 20-30% of the world's oceans to be ‘strictly protected’. However, the target also requires these strictly protected areas to be embedded within global representative networks of MPAs, the size and extent of which is not specified. The CBD target is fully measurable in that the full scope of the target is subject to a numerical target of 10% of all marine ecological regions under national jurisdiction.

What information needs to be collected?

Measurability is inherently related to the specificity of the targets, in that unambiguous, operational definitions of target requirements render measurement information needs explicit. However, as discussed, operational definitions are lacking for the conceptual requirements of all three targets.

Capacity to collect, store, and report necessary information

In addition to knowing what information is needed to monitor progress towards the targets, that information must also be available. If it is not, then there must be the capacity to collect it. There must also be sufficient capacity to store and report the information in a timely manner. The World Database on Protected Areas (WDPA), maintained by the United Nations Environment Programme World Conservation Monitoring Centre (UNEP-WCMC) has been widely used for global protected area monitoring. However, this database has significant limitations for reporting on MPAs (CBD, 2003; Chape *et al.*, 2005) and formal calls were made for better information on MPAs (CBD, 2004a). A global MPA database was developed from the WDPA, which was then expanded and updated, to enable more effective monitoring of the three global targets, to the extent that available information – and specificity of the targets – allowed. This represents an improvement in measurement capacity, but there remain important information gaps which require additional capacity and resources to fill (Wood *et al.*, in press).

Global marine protection targets: are they Achievable?

An achievable target is one that is action-oriented, and where those implementing it have the necessary knowledge, information, skills, and resources to do so (van Herten and Gunning-Schepers, 2000). However, the actions specified by the targets are, in essence, simply to 'designate MPAs', with little guidance on how this should be done, what information it should be based on, who should do it, and with no explicit recognition that much of the information needed to meet the targets (however their requirements are operationally defined) is currently unavailable. In terms of skills and resources, inadequate capacity and insufficient financial and technical resources, are all already well-cited as a major contributor to the failure of existing MPAs in meeting their management objectives (Kelleher *et al.*, 1995b; Alder, 1996; McClanahan, 1999). Achievement of the targets, which represents a 6-375 fold increase in the current global marine area protected (depending on the target & level of protection required), is thus likely to be heavily limited by this lack of resources.

Global marine protection targets: are they Realistic?

Targets are realistic when the level of change required to meet them is itself attainable. They should be ambitious enough to require commitment, motivation and effort to reach them (targets that are too low may lapse into formalities), but not so high that they cause frustration and complacency (van Herten and Gunning-Schepers, 2000). The extent to which a target is realistic is thus also related to the timeframe available to reach it. In the case of the global MPA targets, the current rate of growth of the global MPA network is on the order of 5% per year, and has not changed from that prior to adoption of the targets. At this rate, the targets will not be met for at least several decades (Figure 2.11), and it needs to increase by at least an order of magnitude in order for the targets to be met on time (Wood *et al.*, in press).

Global marine protection targets: are they Time-bound?

All three global marine protection targets have explicitly stated deadlines. The deadline for the WSSD and WPC targets is 2012. The CBD target deadline is 2010, except for MPAs in small island developing states, where the deadline is 2012 (CBD, 2006a).

Discussion

Overall, the three global marine protection targets provide a certain degree of specificity, which represents explicit attempts to address, at least in part, some of the ecological concerns expressed about earlier, purely percentage-based targets. However, this specificity is largely superficial in that the concepts embodied in the targets are not operationally defined. This reduces their measurability, which in turn reduces their achievability, because the actions and behaviours needed to meet them, as well as who is responsible for implementing them, are not readily apparent. Given these factors, in combination with the very short timeframe over which the targets were agreed to be implemented, they do not appear to be realistic, especially since additional time is necessary to gather the information, resources and capacity required to implement them. Although economies of scale are likely to occur at some point (James *et al.*, 1999; Balmford *et al.*, 2004), the disparity between the current situation and that needed to meet the targets (Wood *et al.*, in press), is so large that as yet, there is little, if any, scale to economise on. Therefore, the three global marine protection targets largely fail to meet the SMART criteria.

However, while many of the difficulties facing the achievement of the targets may appear to stem from the apparent lack of operational definitions for the target requirements, it is not at all evident that the global targets themselves should provide them, or in fact that this is the true root problem. The most comprehensive MPA network is physically and socially connected, hierarchical in structure, and functions at multiple, complementary scales through that hierarchy (Agardy, 2005). The development of such networks is ultimately the consequence of political processes, which, like ecological systems, operate on multiple scales (Agrawal and Ostrom, 2006). Accordingly, one might expect a vertically integrated system of policies, goals and targets towards a global MPA network, applicable at multiple scales (both spatial and temporal) ranging from global to local (Latham and Locke, 1991; Roberts, 2005). Global targets thus provide the overarching framework at the top of this hierarchy, providing a common context for more local efforts (Roberts, 2005). However, there has been little recognition of the need for a hierarchical system of targets, or, in particular, of the need to translate large-scale targets into intermediate-scale ones, although they are now beginning to emerge (the CBD target explicitly describes itself as an intermediate, policy-driven goal (CBD, 2005)). Nevertheless, most assessments of the ecological relevance of large-scale targets have applied them directly to very local or specific contexts, without any interpretation or modification for the scale of application (Solomon *et al.*, 2003; Wiersma and Nudds, 2006). As such the questionable ecological relevance of large-scale targets may be (at least partially) due to a mismatch between the scale at which the targets were intended to operate, and the scale at which they were assessed.

Thus, while large-scale targets have been criticised for being primarily the products of political processes (Soulé and Sanjayan, 1998; Solomon *et al.*, 2003; Svancara *et al.*, 2005), this is, in fact, inevitable in membership organizations where decisions are reached by negotiated consensus (Roberts, 2005). Furthermore, the political process through which large-scale targets are formulated also represents a public and formal commitment to action. Indeed, commitment to and belief in a particular issue (i.e., political will) is a fundamental pre-requisite for behaviour directed toward resolving it (Latham and Yukl, 1975; Routhe *et al.*, 2005). As such it may in fact be more appropriate that the primary function of large-scale targets be viewed as psychological, rather than ecological. Psychology is a key, yet often overlooked, component of biodiversity conservation, since human behaviour is a major contributor to both causing and slowing or preventing biodiversity loss (Saunders *et al.*, 2006). However, even if one accepts that the primary function of large-scale targets is to motivate action, the question remains: why do they appear to be failing in this regard?

From a psychological perspective, the formulation of the target is critical to its attainment, and the SMART concept can provide useful guidance in this regard, in particular highlighting the importance of developing operational definitions of concepts to be implemented. However, a hierarchical framework of targets as described above implicitly requires that global targets actually be flexible – and thus not overly specific – in order to retain relevance to the full diversity of contexts at smaller scales. It also implies that operational definitions of target requirements need to be developed at multiple scales. Under this framework, the responsibility for developing operational definitions must fall, not to the global targets, but to the organizations responsible for implementing them at that scale of function. As such, the process to translate global targets into operational and measurable action is fundamentally dependent on political will, which both influences and is influenced by perceived attainability of targets, and which is also a product of target formulation.

Under-ambitious targets may not solicit a change in behaviour, while over-ambitious targets may inhibit progress towards the target, especially where confidence in ability to meet it is low. Challenging targets may also hurt performance if: a) little or no strategy is provided, but the target is adopted at an early stage of learning; b) the task is heuristic; and c) there is pressure to perform well immediately (Latham and Locke, 1991). These conditions mirror the current context for the global MPA targets, in that: a) the current level of protection is near-zero, and the increase required is substantial; in addition, systematic planning of MPA networks is still a relatively nascent field that typically requires more data than are currently available, and has to date been used patchily, at relatively small scales, and mostly in an academic context (Leslie *et al.*, 2003; Sarkar *et al.*, 2006; Wood *et al.*, in press); b) the process to create global networks of MPAs is politically and ecologically complex; and c) the target deadlines are imminent.

The formal adoption of the targets does demonstrate a certain level of political will to increase the level of marine protection globally, but it is also important to recognise that this political will resides largely with those already interested in marine resource management and biodiversity conservation, and not necessarily those with the decision-making power to invest the resources necessary for the targets to be met. Arguably the most successful example of large-scale target attainment is the reduction and phase out of chlorofluorocarbons (CFCs) via the Convention for the Protection of the Ozone Layer (1985) and the Protocol on Substances that Deplete the Ozone Layer (1987). However, much of this success has been attributed to the fact that DuPont, a chemical giant which at the time accounted for about 25% of global CFC production, had already developed alternatives to the substances that were to be phased out through the Montreal Protocol. As such, industry resistance to the reduction targets was lower because economically viable (and profitable) alternatives were already available (Elliott, 1998; Broadhead, 2002).

While concerns expressed about the ecological relevance of the targets are thus indeed valid, it is also critical to consider the psychological and political context of the target-setting process. If the three global targets assessed here had been formulated so that they met all of the SMART criteria, in particular the achievability and realistic criteria, then their ecological relevance may well have been even more heavily questioned. By the same token, while the incorporation of ecological considerations of MPA design into the targets is an improvement to earlier, purely percentage-based targets, the gains made in ecological relevance may have come at a cost to the attainability of the targets. Nevertheless, the global targets can only be viewed as very ambitious when compared to the current level of protection, resources available to implement them, and the time over which they should be implemented. Over time the targets have also become more specific, the WSSD target being the most general and the CBD target being the most specific and action-oriented, as well as being adopted by nation states, rather than primarily conservationists. This indicates a consolidation of broader political support for the notion of a global network of MPAs. Indeed, formal commitments have been made by the Government of Fiji, Micronesian countries, Indonesia and Grenada to protect substantial proportions of their marine waters, as explicit responses to the global targets (WWF, 2005; CBD, 2006b). Thus, the targets do seem to have been at least partially successful at providing the political motivation necessary to result in conservation action at smaller scales.

Whether this will be sufficient for the targets to be met on time, and ultimately for the development of a truly ecologically relevant global MPA network, remains to be seen, but it does suggest that it may be premature to abandon them altogether.

References

Agardy, T. (2005) Global marine conservation policy versus site-level implementation: the mismatch of scale and its implications. *Marine Ecology Progress Series*, **300**, 242-248.

Agardy, T., Bridgewater, P., Crosby, M.P., Day, J., Dayton, P.K., Kenchington, R., Laffoley, D., McConney, P., Murray, P.A., Parks, J.E. & Peau, L. (2003) Dangerous targets? Unresolved issues and ideological clashes around marine protected areas. *Aquatic Conservation: Marine and Freshwater Ecosystems*, **13**(4), 353-367.

Agrawal, A. & Ostrom, E. (2006) Political Science and Conservation Biology: a Dialog of the Deaf. *Conservation Biology*, **20**(3), 681-682.

Alder, J. (1996) Have tropical marine protected areas worked? An initial anaysis of their success. *Coastal Management*, **24**(2), 97-114.

Ballantine, W.J. (1995) *Networks of marine reserves are practical and necessary*. Science and Management of Protected Areas Association, Halifax, Nova Scotia, Canada.

Balmford, A., Gravestock, P., Hockley, N., McClean, C.J. & Roberts, C.M. (2004) The worldwide costs of marine protected areas. *Proceedings of the National Academy of Sciences*, **101**(26), 9694-9697.

Boersma, P.D. & Parrish, J.K. (1999) Limiting abuse: marine protected areas, a limited solution. *Ecological Economics*, **31**(2), 287-304.

Broadhead, L. (2002) *International environmental politics: the limits of green diplomacy*. Lynne Reinner Publishers Inc, Boulder, Colorado, USA.

Busse, R. & Wismar, M. (2002) Health target programmes and health care services - any link? A comparative and conceptual study (Part 1). *Health Policy*, **59**, 209-221.

CBD (2002) *Global strategy for plant conservation - follow-up on SBSTTA Recommendation VII/8 Paragraph 2 on the refinement of the 16 draft targets included in the proposed strategy.* Convention on Biological Diversity, Las Palmas de Gran Canaria.

CBD (2003) *Marine and coastal biodiversity: review, further elaboration and refinement of the programme of work. Report of the Ad Hoc Technical Expert Group on Marine and Coastal Protected Areas.* Convention on Biological Diversity, Montreal, Canada.

CBD (2004) *Decisions adopted by the Conference of the Parties to the Convention on Biological Diversity at its seventh meeting (Decision VII/30)*. Convention on Biological Diversity, Montreal, Canada.

CBD (2005) *Draft global outcome-oriented targets for the programme of work on marine and coastal biological diversity*. Convention on Biological Diversity, Bangkok, Thailand.

CBD (2006a) *Decisions adopted by the Conference of the Parties to the Convention on Biological Diversity at its eighth meeting (Decision VIII/15, Annex IV)*. Convention on Biological Diversity, Curitiba, Brazil.

CBD (2006b) *A Global Island Partnership launched by the President of Palau at the Curitiba Biodiversity Conference: Press Release*. CBD, Curitiba, Brazil.

Chape, S., Harrison, J., Spalding, M. & Lysenko, I. (2005) Measuring the extent and effectiveness of protected areas as an indicator for meeting global biodiversity targets. *Philosophical Transactions of the Royal Society of London. Series B: Biological Sciences*, **360**, 443-455.

Day, J., Hockings, M. & Jones, G. (2002) *Measuring effectiveness in marine protected areas—principles and practices*. Keynote presentation in Aquatic Protected Areas. What works best and how do we know? World Congress on Aquatic Protected Areas, Cairns, Australia.

Department for Environment and Heritage (2006) *No species loss: a biodiversity strategy for South Australia 2006-2016: Draft for public review*. Department for Environment and Heritage, Victoria, Australia.

Elliott, L. (1998) *Global politics of the environment*. Macmillan Press Ltd, London, UK.

Erez, M. (2005) Goal-Setting. In *Blackwell Encyclopedic Dictionary of Organizational Behavior, 2* (eds N. Nicholson, P. Audia & M. Pillutla), pp. 138-141. Blackwell, Oxford, UK.

Gaines, S.D., Gaylord, B. & Largier, J.L. (2003) Avoiding current oversights in marine reserve design. *Ecological Applications*, **13**(1), S32-S46.

Halpern, B.S. & Warner, R.R. (2003) Matching marine reserve design to reserve objectives. *Proceedings of the Royal Society of London Series B-Biological Sciences*, **270**(1527), 1871-1878.

Hastings, A. & Botsford, L.W. (2003) Comparing designs of marine reserves for fisheries and for biodiversity. *Ecological Applications*, **13**(1), S65-S70.

HM Treasury (2003) *The Green Book: Appraisal and Evaluation in Central Government*. The Stationary Office, London, UK.

Hockings, M., Stolton, S. & Dudley, N. (2000) *Evaluating effectiveness: a framework for assessing the management of protected areas*. IUCN, Gland, Switzerland, and Cambridge, UK.

Hockings, M., Stolton, S., Dudley, N. & Parrish, J. (2004) *The World Heritage Management Effectiveness Workbook: revised edition*. UNESCO & IUCN, Paris, France.

IUCN (2003) *WPC Recommendation 5.22: Building a global system of marine and coastal protected area networks*. IUCN, Durban, South Africa.

James, A.N., Green, M.J.B. & Paine, J.R. (1999) *A global review of protected area budgets and staff*. UNEP-WCMC, WCPA, Cambridge, UK.

Jameson, S., Tupper, M.H. & Ridley, J.M. (2002) The three screen doors: *can* marine "protected" areas be effective? *Marine Pollution Bulletin*, **44**(11), 1177-1183.

Jones, G. (2000) *Outcomes-based evaluation of management for protected areas – a methodology for incorporating evaluation into management plans*. The design and management of forest protected areas: papers presented at the Beyond the Trees conference. 8-11 May 2000, Bangkok, Thailand, WWF Switzerland.

Jones, P.J.S. (2002) Marine protected area strategies: issues, divergences and the search for middle ground. *Reviews in Fish Biology and Fisheries*, **11**(3), 197-216.

Kawohl, W., Temple-Bird, C., Lenel, A. & Kaur, M. (2003) *How to manage the finances of your healthcare technology management teams*. World Health Organisation, Geneva, Switzerland.

Kelleher, G., Bleakey, C. & Wells, S. (1995) Introduction. In *A global representative system of marine protected areas.* (eds G. Kelleher, C. Bleakey & S. Wells), The Great Barrier Reef Marine Park Authority, The World Bank, The World Conservation Union, Washington DC, USA.

Latham, G.P. & Locke, E.A. (1991) Self-regulation through goal setting. *Organizational Behavior and Human Decision Processes*, **50**, 212-247.

Latham, G.P. & Pinder, C.C. (2005) Work motivation theory and research at the dawn of the twenty-first century. *Annual Review of Psychology*, **56**, 485-516.

Latham, G.P. & Yukl, G.A. (1975) A review of research on the application of goal setting in organisations. *The Academy of Management Journal*, **18**(4), 824-845.

Leslie, H., Ruckelshaus, M., Ball, I.R., Andelman, S. & Possingham, H.P. (2003) Using siting algorithms in the design of marine reserve networks. *Ecological Applications*, **13**(1), S185-S198.

Locke, E.A. & Latham, G.P. (2002) Building a practically useful theroy of goal setting and task motivation. *American Psychologist*, **57**(9), 705-717.

Lubchenco, J., Palumbi, S.R., Gaines, S.D. & Andelman, S.J. (2003) Plugging a hole in the ocean: the emerging science of marine reserves. *Ecological Applications*, **13**(1), S3-S7.

Lumb, C.M., Fowler, S.L., Atkins, S.M., Gilliland, P.M. & Vincent, M.A. (2004) *The Irish Sea Pilot: Developing marine nature conservation objectives for the Irish Sea*. DEFRA, Peterborough, UK.

Manghubai, S. (2001) *Interim Guidelines for the Assessment of Management Effectiveness of Marine Protected Areas in the western Indian Ocean.* IUCN, Gland, Switzerland.

Margules, C.R. & Pressey, R.L. (2000) Systematic conservation planning. *Nature*, **405**, 243-253.

McClanahan, T.R. (1999) Is there a future for coral reef parks in poor tropical countries? *Coral Reefs*, **18**, 321-325.

Mintzberg, H. (1992) *The rise and fall of strategic planning: reconceiving roles for planning, plans, planners*. The Free Press, New York, USA.

Muncey, J. & McGinty, J. (1998) Target setting in special schools. *British Journal of Special Education*, **25**(4), 173-173.

Norse, E.A. & Crowder, L.B. (eds.) (2005) *Marine Conservation Biology*. Island Press, Washington DC, USA.

Palumbi, S.R. (2003) Population genetics, demographic connectivity, and the design of marine reserves. *Ecological Applications*, **13**(1), S146-S158.

Plummer, N., Lipa, W., Palmer, S., Prank, G., Shortridge, J. & Stuber, D. (2005) *Guidelines on Climate Data Management Version 5.0, October 2005*. Australian Government Bureau of Meteorology, Melbourne, Australia.

Pollnac, R.B., Cawford, B.R. & Gorospe, M.L.G. (2001) Discovering factors that influence the success of community-based marine protected areas in the Visayas, Philippines. *Ocean & Coastal Management*, **44**, 683-710.

Pomeroy, R.S., Parks, J.E. & Watson, L.M. (2004) *How is your MPA doing? A guidebook of natural and social indicators for evaluating marine protected area management effectiveness*. IUCN, Gland, Switzerland and Cambridge, UK.

Pressey, R.L., Cowling, R.M. & Rouget, M. (2003) Formulating conservation targets for biodiversity pattern and process in the Cape Floristic Region, South Africa. *Biological Conservation*, **112**(1-2), 99-127.

Pressey, R.L., Humphries, C.J., Margules, C.R., Vane-Wright, R.I. & Williams, P.H. (1993) Beyond opportunism: key principles for systematic reserve selection. *Trends in Ecology and Evolution*, **8**(4), 124-128.

Rice, J., Trujillo, V., Jennings, S., Hylland, K., Hagstrom, O., Astudillo, A. & Jensen, J.N. (2005) *Guidance on the application of the ecosystem approach to management of human activities in the European Human Environment*. ICES, Copenhagen, Denmark.

Roberts, J. (2005) Millennium development goals: are international targets now more credible? *Journal of International Development*, **17**(1), 113-129.

Rodrigues, A.S.L., Andelman, S.J., Bakarr, M.I., Boitani, L., Brooks, T.M., Cowling, R.M., Fishpool, L.D.C., da Fonseca, G.A.B., Gaston, K.J., Hoffman, M., Long, J.S., Marquet, P.A., Pilgrim, J.D., Pressey, R.L., Schipper, J., Sechrest, W., Stuart, S.N., Underhill, L.G., Waller, R.W., Watts, M.E.J. & Yan, X. (2004) Effectiveness of the global protected area network in representing species diversity. *Nature*, **428**, 640-643.

Rondinelli, D.A. (1976) Public planning and political strategy. *Long Range Planning*, **IX**(2), 75-82.

Routhe, A.S., Jones, R.E. & Feldman, D.L. (2005) Using theory to understand public support for collective actions that impact the environment: alleviating water supply problems in a nonarid biome. *Social Science Quarterly*, **86**(4), 874-897.

Sarkar, S. (2005) *Biodiversity and environmental philosophy: an introduction*. Cambridge University Press, Cambridge, UK.

Sarkar, S., Pressey, R.L., Faith, D.P., Margules, C.R., Fuller, T., Stoms, D.M., Moffett, A., Wilson, K.A., Williams, K.J., Williams, P.H. & Andelman, S. (2006) Biodiversity conservation planning tools: Present status and challenges for the future. *Annual Review of Environment and Resources*, **31**, 123-159.

Saunders, C.D., Brook, A.T. & Myers Jr., O.E. (2006) Using psychology to save biodiversity and human well-being. *Conservation Biology*, **20**(3), 702-705.

Solomon, M., Van Jaarsveld, A.S., Biggs, H.C. & Knight, M.H. (2003) Conservation targets for viable species assemblages? *Biodiversity and Conservation*, **12**(12), 2435-2441.

Soulé, M.E. & Sanjayan, M.A. (1998) Conservation targets: do they help? *Science*, **279**(5359), 2060-2061.

Stark, J. (2004) *Implementing marine protected areas policy: lessons from Canada and Australia*. MSc thesis, School of Resource and Environmental Management, Simon Fraser University, Burnaby, Canada.

Staub, F. & Hatziolos, M. (2004) *Score card to assess progress in achieving management effectiveness goals for marine protected areas*. The World Bank, Washington DC, USA.

Sumaila, U.R. & Armstrong, C.W. (2006) Distributional and efficiency effects of marine protected areas: a study of the Northeast Atlantic cod fishery. *Land Economics*, **82**(3), 321-332.

Sumaila, U.R., Guénette, S., Alder, J. & Chuenpagdee, R. (2000) Addressing ecosystem effects of fishing using marine protected areas. *ICES Journal of Marine Science*, **57**(3), 752-760.

Svancara, L.K., Brannon, R., Scott, J.M., Groves, C.R., Noss, R.F. & Pressey, B. (2005) Policy-driven vs. evidence-based conservation: a review of political targets and biological needs. *Bioscience*, **55**, 989-995.

Taylor, R.N. (1976) Psychological aspects of planning. *Long Range Planning*, **IX**(2), 66-74.

Thomas, R.K. (2003) *Health Services Planning*. Kluwer Academic/Plenum Publishers, New York, USA.

Tucker, G. (2005) *A review of biodiversity conservation performance measures*. Earthwatch Institute, Oxford, UK.

United Nations (2002) *Report of the World Summit on Sustainable Development, Johannesburg, South Africa, 26 August-4th September 2002*. United Nations, Johannesburg, South Africa.

United Nations (2006) *The Millennium Development Goals Report 2006*. United Nations, New York, USA.

van Herten, L.M. & Gunning-Schepers, L.J. (2000) Targets as a tool in health policy Part I: lessons learned. *Health Policy*, **52**, 1-11.

Wagner, L.D., Ross, J.V. & Possingham, H. (2006) Catastrophe management and inter-reserve distance for marine reserve networks. *Ecological Modelling*, **201**(1), 82-88.

Walmsley, S.F. & White, A.T. (2003) Influence of social, management and enforcement factors on the long-term ecological effects of marine sanctuaries. *Environmental Conservation*, **30**(4), 388-407.

Walters, C. (2000) Impacts of dispersal, ecological interactions, and fishing effort dynamics on efficacy of marine protected areas: how large should protected areas be? *Bulletin of Marine Science*, **66**(3), 745-757.

Wells, S. & Mangubhai, S. (2004) *A workbook for assessing management effectiveness of marine protected areas in the Western Indian Ocean*. IUCN Eastern African Regional Programme, Nairobi, Kenya.

Wiersma, Y.F. & Nudds, T.D. (2006) Conservation targets for viable species assemblages in Canada: Are percentage targets appropriate? *Biodiversity and Conservation*, **15**(14), 4555-4567.

Wood, L.J. (submitted) Conceptual challenges to global monitoring of marine protected areas.

Wood, L.J., Fish, L., Laughren, J. & Pauly, D. (in press) Assessing progress towards global marine protection targets: shortfalls in information and action. *Oryx*.

WWF (2005) *Weaving a tapestry of protection and sustainability*. WWF, Suva, Fiji.

5. GIS-based multicriteria evaluation and fuzzy sets to identify priority sites for marine protection[1]

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Introduction

Marine protected areas (MPAs) are most commonly defined as:

"any area of intertidal or subtidal terrain, together with its overlying water and associated flora, fauna, historical and cultural features, which has been reserved by law or other effective means to protect part or all of the enclosed environment" (IUCN, 1988).

MPAs have the potential to contribute to a range of ecosystem goods and services, particularly the protection of marine biodiversity and the sustainable management of fisheries (Boersma and Parrish, 1999; Botsford *et al.*, 2001; Hastings and Botsford, 2003; Lubchenco *et al.*, 2003). While MPAs represent one of a suite of policies considered necessary to stop the current decline in fish catches, biomass, and biodiversity as a whole (Ward *et al.*, 2001), it is one that resonates through the recent literature and the global conservation community. For example, the World Summit on Sustainable Development (WSSD) Plan of Implementation (2002) committed to establishing a global representative network of marine protected areas by 2012 (United Nations, 2002a). In accordance with this, Recommendation 5.22 of the Vth World Parks Congress (WPC) (2003) was made to '[g]reatly increase the marine and coastal area managed in marine protected areas by 2012 …. includ[ing] strictly protected areas that amount to at least 20-30% of each habitat' (IUCN, 2003).

The decision-making process of siting new MPAs is a highly complex one that requires the consideration of multiple factors including cost, the time frame over which the MPAs will be created, and the ecological, geological, hydrological, socio-economic, political and cultural environment of the area. There is also a growing emphasis on the development of networks of MPAs (United Nations, 2002a), but the definition of network itself remains unclear. Roff (2005) states that a "network of MPAs should capture and be able to sustain the regional elements of marine biodiversity", and that the current global configuration of MPAs represents only a 'set' of MPAs and not a true network. However, the criteria for achieving a true MPA network are complex and a method to assess the extent to which a set of MPAs constitutes a network is yet to be developed. Consequently, the ability of existing MPA site selection tools to identify 'true' MPA networks has not yet been fully established, and hence it might be more appropriate to refer here to a global set of MPAs rather than a global network. The entire process is further constrained by the uncertainty conferred on the results of any site selection model by the general lack of data covering genetic, species, ecosystem and ecological processes in marine systems (Ward *et al.*, 1999). Underlying all of these complexities of the MPA site selection problem is the lack of coordination and consistent frameworks for marine conservation at national, regional and international levels (Roff, 2005; Wood, submitted).

In light of the growing momentum behind the drive for a rapid and substantial increase in the extent of marine protection at the global scale, there is a clear and urgent need for the development of a theoretical framework to guide the realisation of this global MPA goal. In order to address these issues in an objective manner, this research develops an integrated decision support framework based on geographic information systems (GIS), multicriteria evaluation (MCE) and fuzzy sets to objectively identify priority locations for future marine protection. The framework was implemented in the Pacific Canadian Exclusive Economic Zone (EEZ) using two objectives that are largely considered to be in conflict - biodiversity conservation and fisheries profit-maximisation. The chapter begins with an overview of marine protected areas and resource management, then describes the integrated decision support framework developed, implements the framework in a Western Canada case study, and then discusses its significance.

Global marine resource conservation and extraction

Currently around 0.6% of the world's marine habitats are subject to some level of protection, and only 0.08% are subject to strict 'no-take' protection (Wood *et al.*, in press). This has often been related to the generally ad hoc basis on which many MPAs have been designated. In the designation process, there is rarely a systematic and comprehensive assessment or consideration of the conflicts that may arise from partially or completely closing an area that previously had been open to resource use (Jones, 2002; Roberts *et al.*, 2003b). The development of fishing technologies over the last 50 years has made fishing essentially ubiquitous, such that previously inaccessible areas that once acted as natural refuges from fishing have now been eliminated (Agardy *et al.*, 2003). MPAs essentially represent the re-creation of these refuges, but with legal, customary, and voluntary access constraints rather than physical ones. Given that fisheries management has been conducted for the last 400 years considering fish as open access resources (Russ and Zeller, 2003), and maximising short term profits (Sumaila and Walters, 2005), fishing can reasonably be assumed to be a prolific source of conflict when imposing constraints on access to resource extraction through the designation of new MPAs. Nevertheless, the momentum to implement MPAs to assist fisheries management is also growing (Bohnsack, 1996; Ward *et al.*, 2001; Lubchenco *et al.*, 2003). Global fish catches, previously considered to be increasing, were recently shown to have been in decline since the 1980's (Watson and Pauly, 2001). Biomass of high trophic level fishes has been shown to have declined by two-thirds since 1950 in the North Atlantic (Christensen *et al.*, 2003), and of predatory fishes by 90% since industrial fisheries began globally (Myers and Worm, 2003). Fishing has been heavily implicated in these declines, and there are now 416 species from the marine biome listed as Vulnerable, Endangered, or Critically Endangered on the

IUCN Red List of Threatened Species (IUCN, 2007a). In 2002, for the first time, marine fish species were listed on Appendix II of the Convention for International Trade in Endangered Species (CITES, 2004). This creates a complex situation since one of the main drivers for increasing marine protection may also be a major challenge to realising it. While it has been suggested that the fishing sector's attitudes towards MPAs may be changing (Agardy *et al.*, 2003) and that there is a great deal of overlap between conservation goals and human (resource extraction) needs (Roberts *et al.*, 2003b), smaller scale studies showing improved compliance must be placed within the context of developing a global network of MPAs affording strict protection to 20-30% of marine habitats. This represents a 31 to 375 fold increase in protection from the status quo, depending on the target and whether any level of protection, or only no-take MPAs, are considered, respectively (Wood *et al.*, in press). The ramifications of this scale of increase for current fishing practises can only be assumed to be substantial: conflicts are inevitable. The literature on contributing factors to MPA management success and failure emphasise the need to address resource use conflicts explicitly (e.g., Jones (2002)). Such conflicts should therefore be addressed explicitly in any analysis that seeks to identify priority marine areas for future protection.

MPA site selection tools

At large scales, models provide a means to integrate diverse data and consider them simultaneously. Many techniques have been developed over the past 25 years to assist in the design of terrestrial protected area 'sets' (Kirkpatrick *et al.*, 1983; Pressey *et al.*, 1993; Pressey *et al.*, 1996; Pressey *et al.*, 1997; Possingham *et al.*, 2000; McDonnell, 2002). More recently, attention has been focused to applying this to the marine biome. For example, MARXAN uses simulated annealing to design a 'set' of MPAs for a given region that is expected to be optimally efficient (Ball, 2004). Efficiency is assumed to reflect minimal costs of implementation and is derived from the boundary length to surface area ratio. Efficiency is assumed to increase as this ratio decreases (Possingham *et al.*, 2000; Ball, 2004). All of these site selection techniques, including MARXAN, have to date been only used at relatively local scales – mostly ranging from single site to a region within a country, and they have also generally focused on primarily achieving biodiversity conservation objectives, for example, Ardron *et al* (2002) and Lewis *et al* (2003). However, given the nature of the MPA site selection problem, it would be preferable to develop an approach that a) can function from national to international scales, and b) addresses multiple objectives simultaneously and explicitly. It was recently suggested that a comprehensive efficient MPA set design approach, such as MARXAN, may only produce optimal results when the entire MPA set is implemented immediately (Meir *et al.*, 2004). There is also the suggestion that using a more simple suite of decision rules may result in an MPA set that approximates optimality more closely when the MPA set is being implemented over an extended period of time (Meir *et al.*, 2004). This could indeed be expected to be the scenario for the implementation of a global set of MPAs.

MCE and fuzzy sets for identification of priority sites for marine protection

Multicriteria evaluation (MCE), or multicriteria decision analysis (MCDA) is defined as the evaluation of a set of alternatives based on multiple criteria where the criteria are quantifiable indicators of the extent to which decision objectives are realized (Malczewski, 1999). Spatially explicit MCE requires data on the spatial distribution of criterion values. In MCE there is a one-to-one relationship between objective and criterion. Multi-objective evaluation is essentially a hierarchical extension of MCE, having a one-to-many relationship between objective and criteria. The most general objectives are at the top of the hierarchy and the most specific criteria at the lowest level (Keeney and Raiffa, 1976; Pitz and McKillip, 1984; Malczewski, 1999). In this chapter, the term MCE will be used to refer to both multicriteria and multiobjective evaluation. In MCE, criterion map layers and decision-maker preferences are aggregated according to a decision rule that yields an optimal solution (Malczewski, 1999). When objectives are in conflict, an 'optimal compromise' solution is found (Eastman *et al.*, 1993; 1995).

MCE was developed as a spatial decision support tool for land use planning when it was realized that spatial suitability analyses alone were fundamentally flawed due to their lack of consideration of decision-makers' preferences. It facilitates the integration of social, political, environmental and economic requirements with suitability analyses (Jankowski and Richard, 1994). Its integration with geographic information systems (GIS) has further enhanced this capability (Carver, 1991; Eastman *et al.*, 1995), and it has since been described as 'perhaps the most fundamental of decision support operations in geographical information systems' (Jiang and Eastman, 2000). MCE is noted for its capacity to ascribe varying importance to different criteria, according to stakeholder preferences (Ceballos-Silva and Lopez-Blanco, 2003), as well as its simplicity and its capacity to handle many different types of criteria (Jankowski and Richard, 1994). It also allows for decision-making under varying levels of uncertainty, from deterministic decisions (low uncertainty) to fuzzy decisions (high uncertainty attributable to the inherent imprecision of information used in decision-making) (Malczewski, 1999). The use of fuzzy set theory when developing criterion layers is considered to allow more flexible MCE operations, and explicitly take into account the continuity and uncertainty in the relation between the criteria and the decision set (Jiang and Eastman, 2000). For example, standardising criterion layers to fuzzy measures means that the criterion value for each cell is standardised to a measure of the possibility of belonging to the set along a continuous scale from 0-1 (real number scale) or 0-255 (byte scale) (Eastman, 2003). This is a more realistic standardisation approach than a binary set membership rule as is used in Boolean analyses, especially when there is uncertainty inherent in the input data. Finally, when used with GIS, MCE also enables the outcomes to be visualized as maps. As a consequence of these various advantages to MCE, it has been used extensively in the resolution of terrestrial resource allocation problems, in fields as varied as: industrial development (Eastman *et al.*, 1995); agricultural development (Janssen and Rietveld, 1990; Ceballos-Silva and Lopez-Blanco, 2003); route selection (Jankowski and Richard, 1994); risk analysis (Chen *et al.*, 2003); habitat suitability modelling (Store and Kangas, 2001); environmental impact assessment (Janssen, 2001); forestry (Huth *et al.*, 2004) and waste management alternatives (Carver, 1991; Chung and Poon, 1996).

The advantages of MCE described above indicate that it has high potential applicability to marine resource decision problems. The optimal compromise solutions derived from MCE are of particular relevance to a global set of MPAs being implemented over time within the context of multiple, potentially conflicting resource use objectives. However, in contrast to the widespread application of MCE to terrestrial decision making, MCE has rarely been used in spatial decision-making for marine natural resource management, including fisheries (Mardle and Pascoe, 1999), and even less so for MPAs. Through an extensive literature search, I identified only three studies applying MCE to MPAs and they were all applied at a local level. Brown *et al* (2001) used the decision support aspects of MCE as a means to facilitate stakeholder involvement in a trade off analysis in a Caribbean MPA, but did not make use of the integration of MCE with GIS. Killpack *et al* (2001) and Villa *et al* (2002) used MCE to develop a zoning plan for a single MPA in the USA and Italy respectively. In the latter two examples, MCE was integrated with GIS to produce spatially explicit results in the form of maps. The work presented in this chapter differs from previous marine MCE analyses in four ways. First, it focuses on large scales. The feasibility of applying the MCE approach in ocean basin and global scale models of new MPA location is assessed for the Pacific Canadian Exclusive Economic Zone (EEZ). Second, MCE is applied in the context of identifying priority areas for future protection and to guide smaller scale analyses in the context of MPA networks, rather than fine-tuning the management of existing MPAs. Third, it uses fuzzy decision-making within the MCE to address the uncertainty associated with the coarse scale marine data and the global MPA 'set' design decision-making process. Further, it differs from many MARXAN applications in that it addresses both biodiversity conservation and fisheries management objectives explicitly to produce optimized trade off results.

Methods

Study site

Canada declared its EEZ under the 1996 Oceans Act (Oceans Act, 1996), and in accordance with the United Nations Convention on the Law of the Sea, which Canada ratified in 2003 (UNCLOS, 2004). The EEZ extends to 200 nautical miles from the coastline and on the Pacific Canadian coast the EEZ covers an area of approximately 458,000km^2 adjacent to the province of British Columbia (Figure 5.1). The British Columbia provincial government mandated a Protected Areas Strategy to protect a minimum of 12% of the province, including its waters, by the year 2000 (British Columbia, 1993), cited in (Zacharias and Howes, 1998). However, five years after the deadline, this target still remains to be met; a recent assessment indicated that only 0.2% of the EEZ is protected, and only 0.0007% of the EEZ is protected by no-take

MPAs (Ardron *et al.*, 2002). Furthermore, most MPAs in the Pacific Canadian EEZ have been, to date, designated on a relatively ad hoc basis, associated with human recreation values rather than ecosystem or species conservation *per se* (Jamieson and Levings, 2001). While there are efforts underway to identify potential networks of MPAs in regions of the Pacific Canadian EEZ (Ardron *et al.*, 2002; Rumsey *et al.*, 2004), none has addressed this issue at the scale of the entire EEZ. This situation is analogous to other locations around the world and is representative of the lack of a framework observed for marine conservation globally. The efforts of the various agencies in Canada that have some mandate for marine conservation (provincial and federal) are neither nationally nor regionally coordinated (Roff, 2005).

Data sources and software

The *Sea Around Us* Project (SAUP), based at the Fisheries Centre, University of British Columbia, has spatially distributed global data on marine fisheries catches, fishing access agreements and a wide variety of marine fisheries and biodiversity-related data in a 0.5° latitude and longitude grid of spatial cells (Watson *et al.*, 2005). Of the 259,200 cells that cover the world, more than 180,000 contain some marine area (Watson *et al.*, 2004). The Pacific Canadian EEZ consists of 295 cells. As these cells are defined by decimal degrees, they vary in size, decreasing in surface area from low to high latitudes. The effect of this on spatial analyses is mitigated by pro-rating cell values according to the surface area of water contained within them. While 0.5° resolution might be considered coarse in analogous analyses in terrestrial ecosystems, and the preconception that oceans are homogenous and resilient has been refuted in recent times (e.g., Agardy (1994)), the high connectivity of the oceans can generally be said to aggregate the scale at which marine processes occur (Jones, 2002). A resolution of 0.5° is a compromise that seeks to address the complexity of the oceans while allowing for large scale analyses. Furthermore, from a practical perspective, obtaining comprehensive data over such large areas at higher resolutions is extremely difficult. The *Sea Around Us* Project database, with its extensive spatially explicit marine data at relatively fine resolutions was used for this analysis. Data processing was performed in the ArcGIS™ and IDRISI Version 14.0 (Kilimanjaro) GIS software systems.

MCE procedure

The decision problem was formulated based on the guidelines suggested by Malczewski (1999). Figure 5.2 shows a general criteria structure that can be used to identify priority sites for MPAs. Due to the problem complexity, and that the goal of this research is to explore the utility of MCE to identify candidate areas for protection, the focus will be on a subset of realistic criteria available for each objective. Each objective was evaluated separately, before being assessed as a multiobjective evaluation during the analysis procedure.

The validity of the MCE outputs was maintained by standardising all criterion input layers such that their scales of measurement were commensurate. Linear scale transformation and fuzzy set membership functions are two available standardisation methods (Malczewski, 1999). The criterion layers in this study were based on continuous data with different units of measurement. They all possessed a level of uncertainty as is inherent in coarse scale marine data. Hence, standardisation of the criterion layers as fuzzy measures was considered to be the most appropriate standardisation technique. The fuzzy set membership function used was a monotonically increasing sigmoid membership model (Figure 5.3).

Once the criterion layers have been standardised, the user assigns weights to them. These weights enable the solution to reflect the importance (as perceived by the user) of the input criteria relative to each other. Similarly, different objectives (which themselves are comprised of multiple, individually weighted criteria) can also be weighted relative to each other. Various weighting schemes both within and between objectives were investigated and their details are presented below. Once weights have been specified, the criteria (or objectives) are combined according to a decision rule. The simple additive weighting method, also known as weighted linear combination, is the most common type of decision rule used in GIS-based decision-making (Malczewski, 1999). This type of rule is implemented by multiplying each criterion layer by its weight and then summing the results (Eastman, 2003) according to the following equation (Malczewski, 1999):

$$A_i = \sum_j w_j x_{ij}$$

where A_i is the final suitability score in each pixel, x_{ij} is the score of the ith pixel with respect to the jth criterion, and weight w_i is a normalised weight so that $\Sigma w_i = 1$. The final result, A_i, is a layer of suitability scores of each pixel to fulfilling the objective under assessment. In this study, 30% of the most suitable cells were selected from the results of the procedure to reflect current directions in the international conservation community towards a global network of MPAs protecting up to 30% of the world's oceans (IUCN, 2003), and to indicate how such a level of protection might be represented spatially on a map.

It is important to recognise that quantitative modelling with GIS, such as MCE, produces results in the form of maps that provide no indication of the level of error associated with those results. However, the robustness of such models is inherently dependent upon the quality of the spatial data used, the quality of the model, and the way the data and the model interact

(Burrough and McDonnell, 1998). Sensitivity analysis is considered to be a valid robustness assessment method, and is defined as “a procedure for determining how the recommended course of action is affected by changes in the inputs of the analysis” (Malczewski, 1999). Sensitivity analysis was performed at various stages of the analysis by varying the input parameters and changing the weighting regimes used within and between objectives.

MCE procedure applied to Objective 1: Biodiversity conservation

Biodiversity conservationists, in the process of identifying new areas to protect, might generally be considered to seek to: (i) fully represent biodiversity, and (ii) ensure persistence of biodiversity (Soulé and Terborgh, 1999). This implies the consideration of all species, habitats and processes in the ecosystem(s) under study. A conceptualisation of this is shown in Figure 5.2. The sub-objectives of species representation and persistence were selected to represent the objective of biodiversity conservation, and the criterion of species distribution selected to represent these sub-objectives. The *Sea Around Us* Project database contains distribution data for all species that have been reported as having been caught in commercial fishing operations, globally. This includes targeted and by-caught species, and all marine mammals, thereby including species of both some and no economic value. Distributions for all species occurring in the Pacific Canadian EEZ, totalling 178, were obtained, as a binary layer of presence/absence by cell. These distributions were used to develop species richness criterion layers. Suitability to protection was assumed to increase with cell species richness. Two scenarios were investigated using these data.

Scenario 1A: All species are equally important to protect

As all species were weighted equally, a total species richness layer was sufficient to represent this scenario (Table 5.1). This layer was developed by running an iterative model to union all species distributions. The layer was then fuzzy standardised according to the monotonically increasing sigmoid membership function illustrated in Figure 5.3, and the top 30% most suitable cells were selected as candidate priority sites for marine protection.

Scenario 1B: some species are more (or less) important to protect than others.

The World Conservation Union (IUCN) Red List of Endangered Species (IUCN, 2007a) classifies species according to their risk of extinction into the following categories, listed in descending order of risk of extinction: Critically Endangered; Endangered; Vulnerable; Near Threatened; Lower Risk; Data Deficient; Not Evaluated. Twenty nine species occurring in the Pacific Canadian EEZ are listed in one of the six categories from Critically Endangered to Data Deficient, although none was classified as Least Concern, so this category was excluded from the analysis. The rest were categorised as Not Evaluated. Species richness per IUCN Red List category was evaluated using the same iterative union model described in scenario 1A, resulting in 6 input factors to the MCE. Each factor was then weighted according to the relative importance of each to the overall objective of biodiversity conservation. In general it may be reasonable to weight species more heavily as their vulnerability to extinction increases, but in reality the configuration of the weighting scheme can vary substantially, and the outputs of the MCE can be heavily influenced by this. A sensitivity analysis of the robustness of results to changes in criterion weights was performed by investigating the results generated by various user-defined weighting schemes; two of them are presented here. The first weighting scheme was linear, thus, the weight increased linearly with the risk of extinction, as defined by the

IUCN Red List categories. The second scheme sought to ascribe higher weights to species at higher risk of extinction, and lower weights to those at lower risk, than linear weighting allowed for. This was achieved by predicting weights according to a logistic growth function, similar to the curve illustrated in Figure 5.3. Actual weights used for all scenarios are shown in Table 5.1.

The two categories, Data Deficient and Not Evaluated, were weighted equally, because in both cases a categorisation of the risk of extinction was unavailable, either due to lack of data or lack of resources. A further sensitivity analysis of the results was conducted by removing one IUCN Red List group at a time and re-running the MCE.

MCE procedure applied to Objective 2: Fisheries management

Fisheries have largely functioned to date under the concept of short-term profit maximisation (Sumaila and Walters, 2005) and the assumption of open access (Russ and Zeller, 2003). The identification of new locations for MPAs by fishing industry stakeholders can therefore be expected to be heavily influenced by the primary goal of minimising the costs experienced by fishers and fishing companies through loss of fishing grounds to increased area under protection. This primary goal is conceived here as the identification of locations of greatest value to the fishery, whose suitability to resource extraction is greatest and restriction of access to which, through the creation of MPAs, is least desirable. The cost minimisation sub-objective (Figure 5.2) was selected to represent the overall objective of identifying priority sites for marine protection for fisheries management.

For such an economic analysis, it was assumed that only species of commercial value should be considered, rather than all species, as for the biodiversity conservation objective. Catch data, by species and by cell, were obtained for the year 2000 from the SAUP database, 90 species in total. Catch data from 2000 were used to provide a current indication of the relative importance of areas to fisheries, and from one year only to reflect the generally short-term, profit-maximising approach to fisheries. In essence, the economic value of the species represents a weighting for that species. Ex-vessel landed price data for 2000 (DFO, 2004) was used to calculate catch value per cell for each species. Only fifteen species were found to constitute 80% of the total catch value, and it was assumed that using the criterion layers for these fifteen species would sufficiently represent the total catch value. The catch value distributions were then combined using the iterative union model used in previous analyses, and standardised as fuzzy measures using the same membership function as described above, to produce a total catch value distribution layer. The 30% most valuable cells were selected from this layer as being the most suitable for continued access to resource extraction.

Multi-objective space allocation analysis

A multi-objective space allocation analysis was completed using the result from Scenario 1B.ii for the biodiversity conservation objective with the result for the fisheries management objective as inputs. The required output was set such that 30% of the EEZ would be allocated to the biodiversity conservation objective (i.e., it would be allocated as highly suitable for future protection). The remaining 70% of the EEZ would be allocated to fishing, and as such considered as more suitable for continued fishing than for protection. A sensitivity analysis of these results was performed by weighting the two objectives differently; this process also served as an exploration of how different stakeholders' perceptions and needs might affect the results of such an analysis, and to have the effects represented spatially. The weights used are summarised in Table 5.2.

Results

Objective 1: Biodiversity Conservation

The results of the MCE for the three scenarios for the biodiversity conservation objective are shown in Figure 5.4. The results are consistent, showing mainly inshore areas as the most suitable for protection based on species richness criteria. There are some subtle differences between these three results. Firstly, in comparison to Scenario 1A (Figure 5.4A), both weighting regimes of Scenario 1B (Figures 5.4B,C) result in a slight increase in the number of offshore pixels selected. These offshore pixels also have a higher suitability score than the few offshore pixels selected in Scenario 1A. This is most likely because many of the species listed in the 'more' endangered IUCN Red List categories have distributions that extend offshore, and so this region is prioritised more strongly in the MCE when IUCN Red Listed species are weighted according to their risk of extinction. However, these species' distributions generally covered both inshore and offshore regions; hence the relatively slight increase in suitability offshore. Secondly, the two weighting regimes yield slightly different results, particularly in the distribution of suitability scores. The suitability scores resulting from the linear weighting regime (Figure 5.4B) are very similar to those of Scenario 1A (Figure 5.4A). This is most likely because the weights are not widely spread in this scenario (Table 5.1) – the heaviest weight is only around 3 times that of the lightest weight. The logistic weighting regime for Scenario 1B.ii generated an inshore region of uniformly high suitability score (Figure 5.4C), which in the previous two analyses was more heterogeneous, although still highly suitable. It is possible that this was caused by the high weight given to the Critically Endangered category, which in this analysis contained only one species (Bocaccio rockfish, *Sebastes paucispinus*), and whose distribution overlaps almost perfectly with this inshore area of high suitability. However, the sensitivity analysis that excluded the Critically Endangered IUCN Red List category yielded a similar result (Figure 5.4D). In fact, this area remained of uniformly high suitability for protection in every sensitivity analysis of this weighting scheme, indicating that the results for this assessment are quite robust.

Objective 2: Fisheries Management

The standardised catch value distribution is shown in Figure 5.5A, and the 30% most valuable cells are shown in Figure 5.5B. There are some areas in the Canadian EEZ that are of no commercial value to fisheries (blank cells within the EEZ), because nothing was caught there in 2000. Therefore, no immediate losses would be incurred by fisheries in the event of these areas becoming protected. As such, these cells currently represent areas of zero conflict for protection. However, very few of these cells, if any, overlap with the areas indicated as highly suitable for protection by any of the biodiversity conservation scenarios (Figures 5.3A-C). In fact, areas of the highest value to the fishery overlap substantially with the areas indicated as highly suitable for protection according to biodiversity conservation objectives, indicating a classic conflicting multiobjective decision problem.

Multiobjective space allocation analysis

The multiobjective analysis was implemented using the result Scenario 1B.ii (Figure 5.4C) for the biodiversity conservation objective and the result for the fisheries management objective (Figure 5.5A) as inputs. The results of using the three different weighting schemes outlined in Table 5.2 are illustrated in Figure 5.6. The results in Figure 5.6 show absolute allocations of each cell to one objective or the other, without an indication of suitability. Figure 5.6A shows the result of weighting biodiversity conservation at 0.3 and fisheries management at 0.7. The weighting of fisheries management is so heavy that very little of the area considered highly suitable for biodiversity conservation is allocated to that objective (Figure 5.4C). Figure 5.6B shows the results of weighting fisheries management and biodiversity objectives equally. Quite a large area that is highly suitable to biodiversity conservation and valuable to fisheries is selected for biodiversity conservation, and vice versa (compare Figure 5.4C with Figure 5.5). The weighting regime also imposes the selection of some areas for biodiversity conservation which are of no value to fisheries management i.e. zero conflict areas (Figure 5.5A), even though they are of lower suitability for biodiversity conservation. Figure 5.6C shows the results of weighting biodiversity conservation at 0.7 and fisheries management at 0.3. This solution is very similar to the biodiversity conservation solution shown in Figure 5.4C.

Discussion

The focus of this study was not to develop policy recommendations, but to assess the utility of MCE to spatially identify marine locations for future protection using a reduced set of criteria. The results should therefore be interpreted in terms of how the MCE process functioned, the questions it raised, and how the MCE results can be investigated for their robustness.

The analysis presented here indicates that MCE does not offer prescriptive solutions to a given resource allocation problem, but instead offers a range of scenarios that address different decision makers' preferences to varying extents. This is generally considered to be preferable as it enables decision-makers to explore different solutions (Possingham *et al.*, 2000) or use it as an integral part of a spatial decision process (Balram *et al.*, 2003). Furthermore, the visualisation of these scenarios as maps can encourage stakeholder discussions. The importance of considering multiple objectives has been made evident, even when using a restricted data set, as was the case here. The high level of overlap between optimal outcomes for biodiversity conservation and fisheries management objectives illustrates that decision-making might easily cause conflicts with different resource users if the process fails to explicitly consider different objectives. The results also show that even with a limited amount of weighting of factors, MCE can explicitly incorporate different priorities, as expressed by decision-makers or stakeholders, and the effects of changing these priorities on the ability of other stakeholders to meet their own objectives are readily visible as a map. Presentation of a spatial resource allocation problem using MCE with GIS also enabled areas of zero conflict to be identified. When spatial resource allocation objectives are in conflict to the extent that biodiversity conservation and resource extraction appear to be in the Pacific Canadian EEZ, areas of no conflict are hard to envisage. This methodological framework has identified such areas and enabled their locations to be visualized. While it may seem most conciliatory to decision-makers to select areas of zero conflict for protection, the suitability scores enable different areas to be compared using a standardised, semi-quantitative scale, and as such inform the decision-maker as to the contribution of the area of zero conflict to both objectives. This improved interpretive ability is conferred to the decision maker by the use of fuzzy standardisation of criteria which would not be possible with the binary outcomes from Boolean analyses of 'suitable' or 'unsuitable' areas. Fuzzy standardisation of criteria also takes into account some of the uncertainty associated with the input data. Finally, the sensitivity analysis performed provided insights into the potential causes behind a particular outcome, as well as an indication of the robustness of the results of the MCE. The application of a sensitivity analysis by changing the weighting regime used also shows that the results can differ depending on the views of the stakeholders in the decision process.

This chapter has shown that MCE has high transferability to the decision-making problem of developing a large scale theoretical framework that can guide the implementation of smaller scale MPA network design initiatives within the context of a global set of MPAs. One challenge that MCE faces with respect to its applicability to MPA site selection is how to model a true 'network', as defined by Roff (2005), rather than simply a set. However, this is a challenge that currently faces all site selection tools, particularly at very large scales. Similarly, while the success of this framework is heavily contingent on the appropriate selection of criteria and the use of reliable data and models by which to represent them, this is also a challenge common to all site selection models.

The MCE framework and implementation has shown substantial potential to support MPA planning and management. However, there are some aspects of the analysis that require further attention, and these provide the basis for future work. Firstly, the framework could be expanded to incorporate as many of the criteria and sub-objectives illustrated in the hierarchical structure of the problem (Figure 5.2) as available data permit. This is critical to obtaining results that are relevant to the scale of study and also reliable for decision-making. For example, the criterion for biodiversity conservation used here was species richness, which is considered to be a useful, but incomplete, measure of biodiversity (Gaston, 1996), cited in (Ward *et al.*, 1999). However, there are various measures of diversity, which function at different geographical scales, and it would be preferable to better represent these in this framework. Community species richness is largely considered to be a measure of alpha diversity (within-community scale), whereas information on variation in habitat types and its use by species can confer information about beta diversity (between-community scale) (Whittaker, 1972). There is an array of literature demonstrating the utility of additional information such as habitat type (e.g., Ward, *et al*. (1999) and Worm, *et al* (2003)), level of threat to habitats (e.g., Roberts, *et al* (2002), as well as biophysical parameters and processes such as latitude and productivity (Worm *et al.*, 2003) to the identification of priority areas for protection of varying levels and scales of biodiversity. The incorporation of additional data sets including those suggested in Figure 5.2 can reasonably be expected to improve the ability of the analysis to capture diversity over multiple scales. Secondly, this framework could be developed to further extend the spatial scale of the model, specifically to the ocean basin or global scale. Thirdly, the results obtained from this framework could be compared to those obtained from using other site selection approaches, particularly complementarity-based place prioritization algorithms, such as MARXAN (Ball and Possingham, 2000) and ResNet (Garson *et al.*, 2002).

Tables

Table 5.1 showing hypothetical criterion weighting schemes, which were applied in this MCE for the Biodiversity Conservation objective.

IUCN Red List Category	Equal Weighting (Scenario 1A)	Linear Weighting (Scenario 1B.i)	Logistic Weighting (Scenario 1B.ii)
Critically Endangered	0.167	0.261	0.321
Endangered	0.167	0.217	0.309
Vulnerable	0.167	0.174	0.242
Lower Risk	0.167	0.130	0.093
Data Deficient	0.167	0.087	0.017
Not Evaluated	0.167	0.087	0.017

Table 5.2 Table showing hypothetical multiobjective objective weighting schemes, which were used in the multiobjective space allocation analysis.

Objective	Scenario A	Scenario B	Scenario C
Biodiversity Conservation	0.3	0.5	0.7
Fisheries Management	0.7	0.5	0.3

Figures

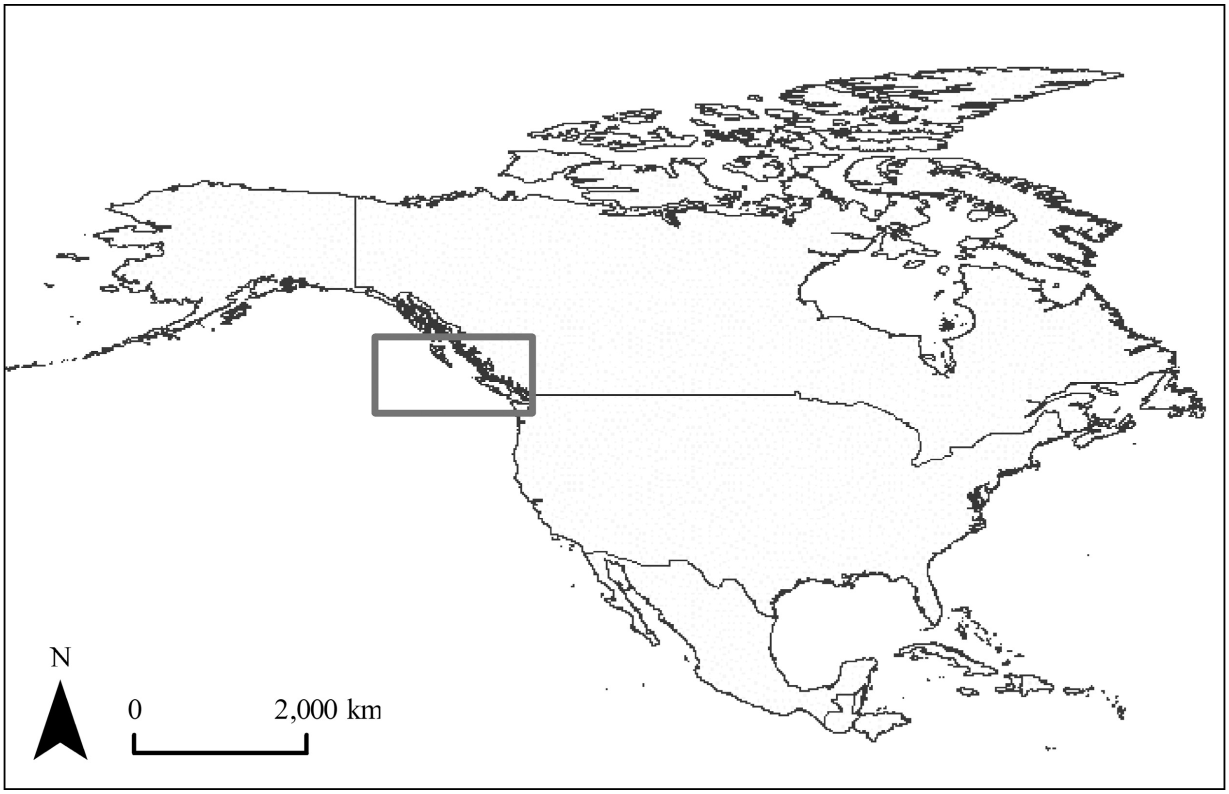

Figure 5.1 Map showing the approximate location of study area, in the heavy box: the Pacific Canadian Exclusive Economic Zone (EEZ)

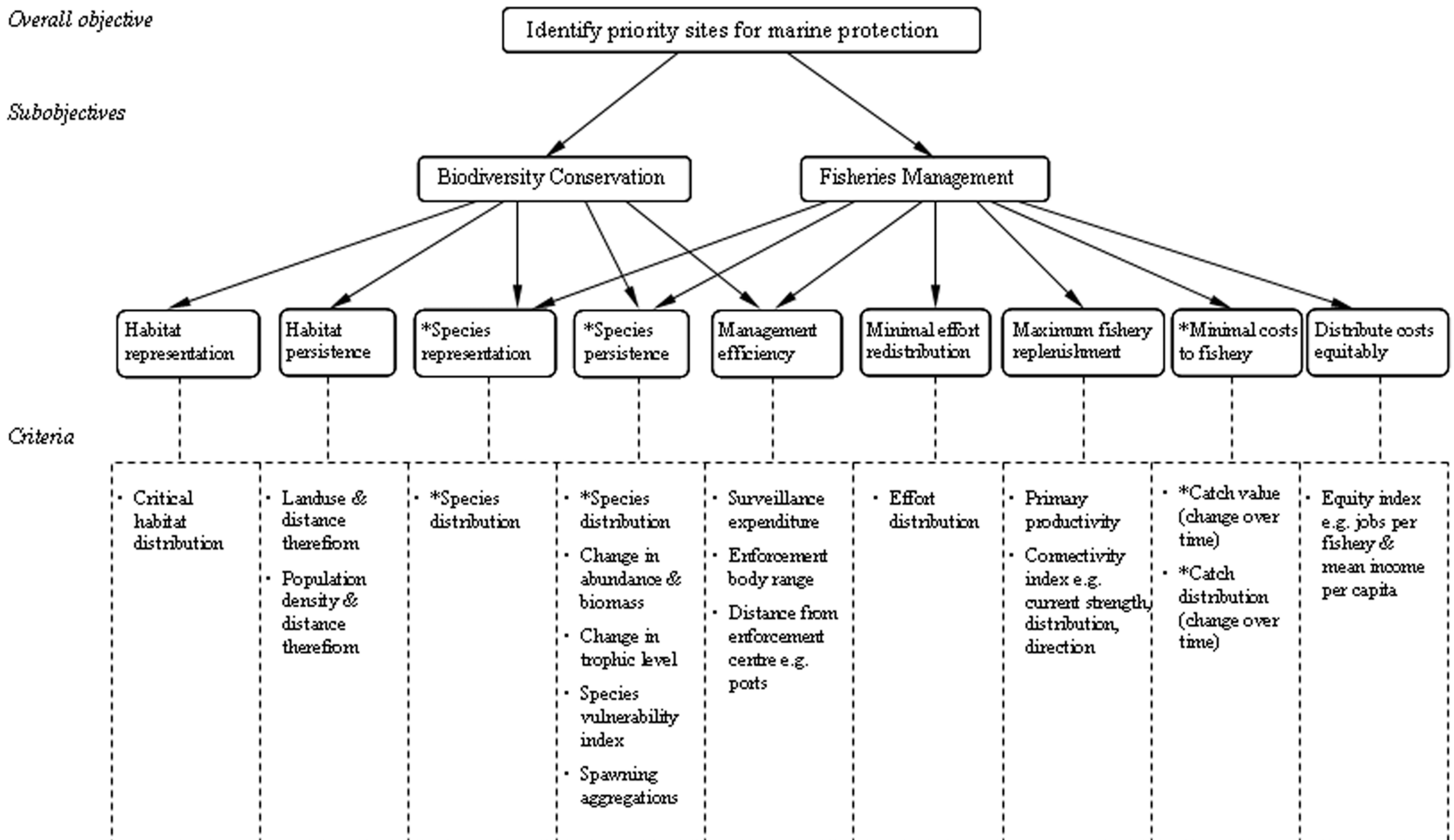

Figure 5.2 Hypothetical formulation of the decision problem of identifying priority sites for marine protection using the hierarchy defined by Malczewski (1999). The criteria listed here do not represent an exhaustive list.

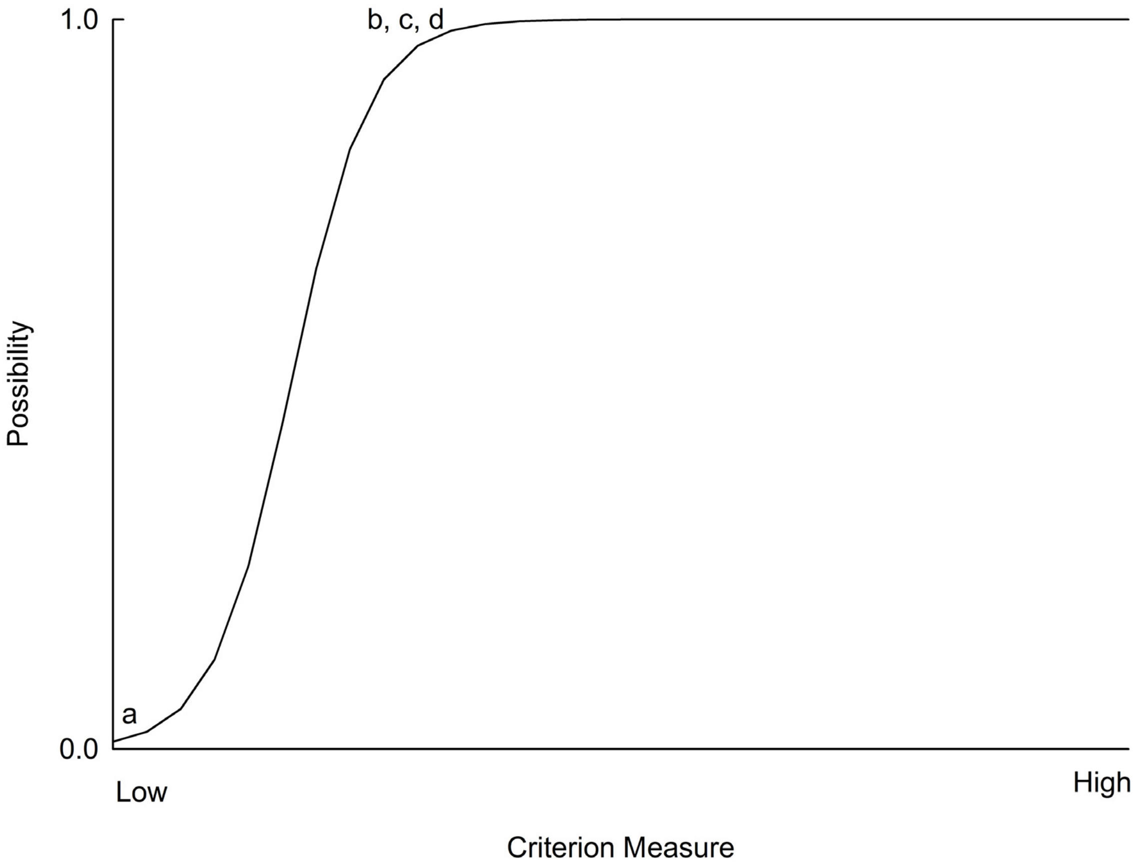

Figure 5.3 Monotonically increasing, sigmoidal fuzzy membership function used to standardize MCE criteria layers. a, b c, and d represent inflection points where the membership function rises above zero, approaches one, falls below one and approaches zero again, respectively. A monotonically increasing function rises to one and never falls again. Hence b, c, and d all have the same criterion value of 1.

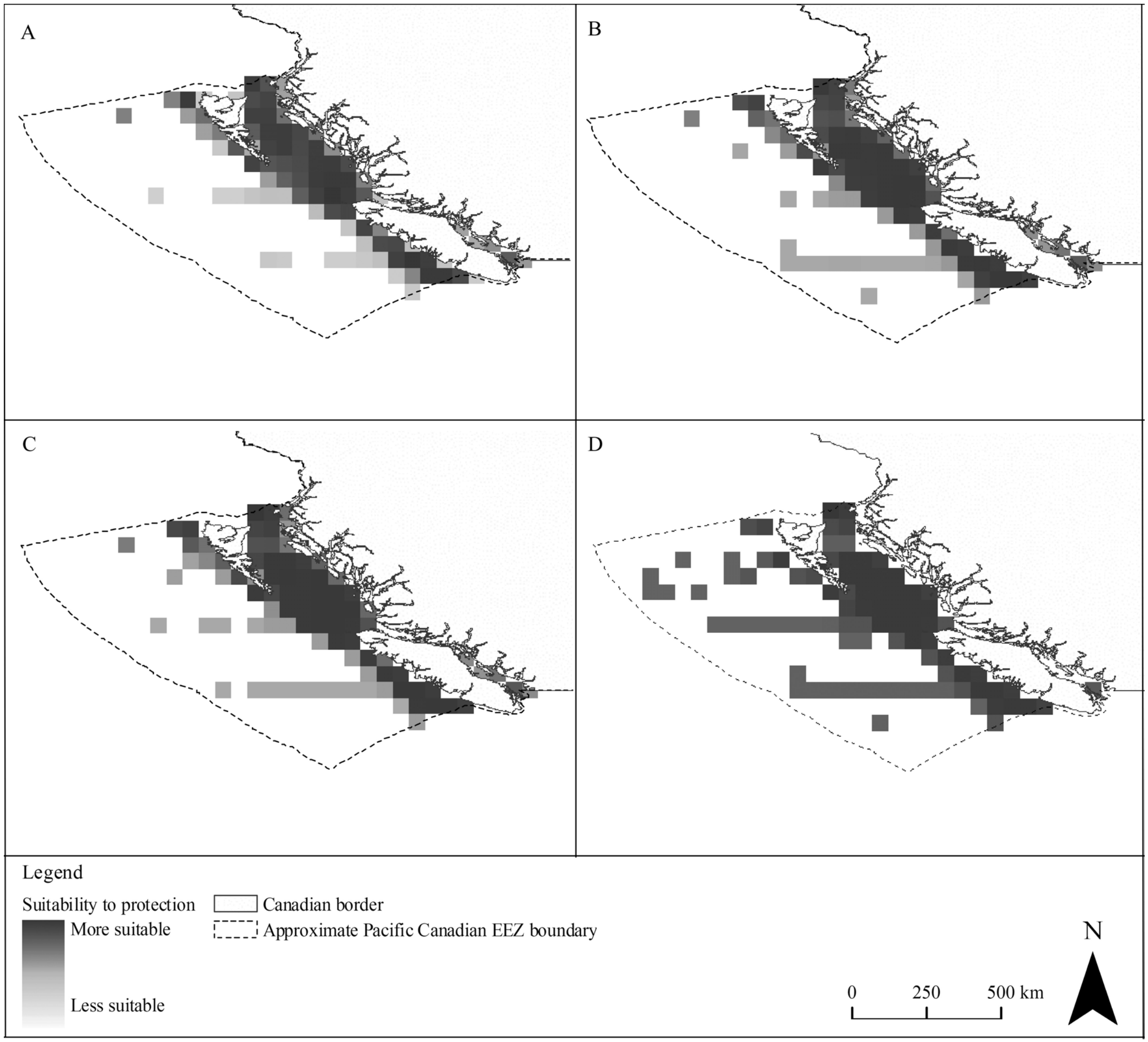

Figure 5.4 Outputs of MCE for Objective 1 – Biodiversity Conservation. Maps show the 30% most suitable cells for protection from within the Pacific Canadian EEZ identified by the following scenarios: A) Equal weights for all species; B) Linear weighting regime for IUCN Red Listed species; C) Logistic weighting regime for IUCN Red Listed species; D) Logistic weighting regime for IUCN Red Listed species with Critically Endangered category removed.

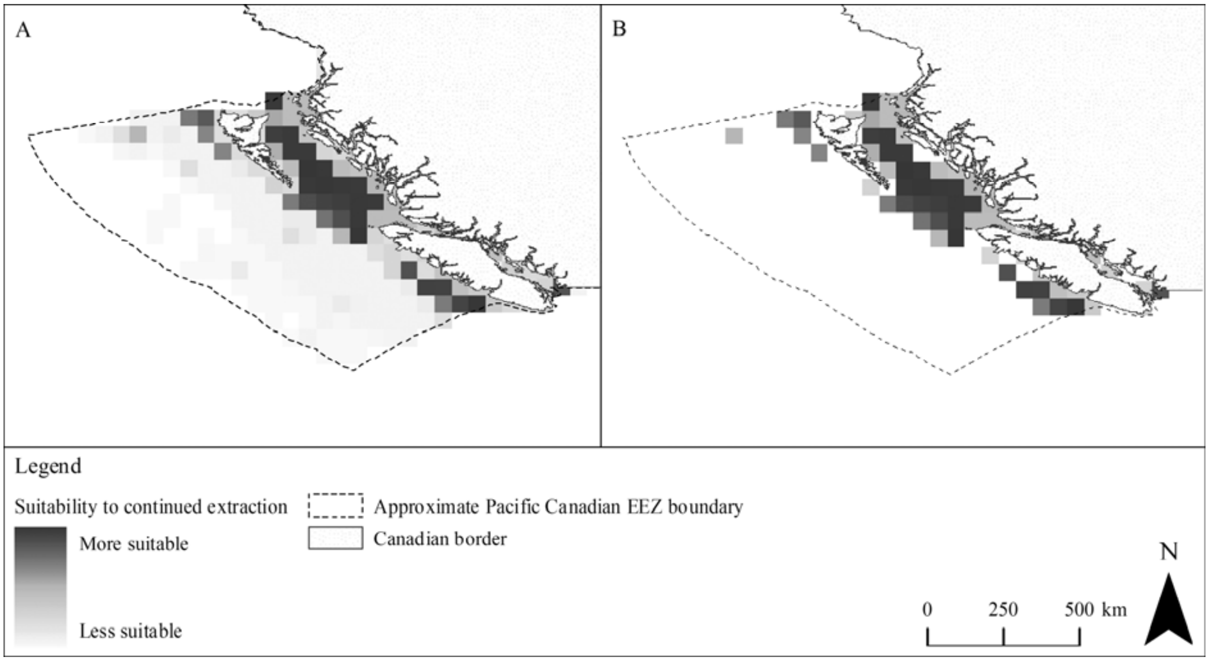

Figure 5.5 Outputs of MCE for Objective 2 – Fisheries Management. Map A shows the catch value distribution across the entire Pacific Canadian EEZ. Map B shows the 30% most suitable cells for continued resource extraction from with the Pacific Canadian EEZ, as identified by catch value distribution.

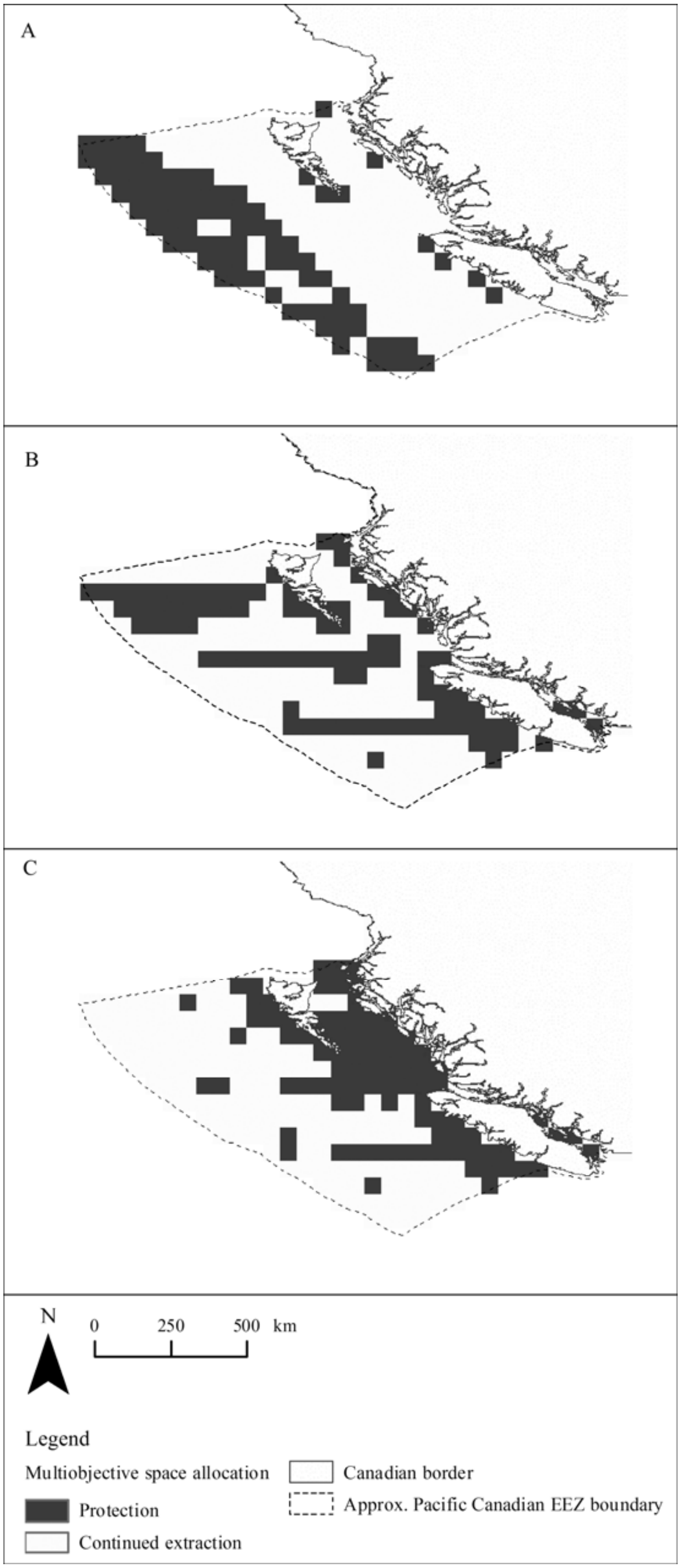

Figure 5.6 Outputs of Multiobjective space allocation analysis, with biodiversity conservation and fisheries management as the two, conflicting objectives. Maps show the 30% most suitable cells for protection from within the Pacific Canadian EEZ, as identified by the following objective weighting schemes, listed biodiversity conservation : fisheries management: A = 0.3 : 0.7; B = 0.5 : 0.5; C = 0.7 : 0.3.

References

Agardy, T. (1994) Advances in marine conservation: the role of marine protected areas. *Trends in Ecology and Evolution*, **9**(7), 267-270.

Agardy, T., Bridgewater, P., Crosby, M.P., Day, J., Dayton, P.K., Kenchington, R., Laffoley, D., McConney, P., Murray, P.A., Parks, J.E. & Peau, L. (2003) Dangerous targets? Unresolved issues and ideological clashes around marine protected areas. *Aquatic Conservation: Marine and Freshwater Ecosystems*, **13**(4), 353-367.

Ardron, J., Lash, J. & Haggarty, D. (2002) *Modelling a network of marine protected areas for the central coast of British Columbia. Version 3.1.* Living Oceans Society, Sointula, Canada.

Ball, I. (2004) *MARXAN - a reserve system selection tool*, The Ecology Centre, University of Queensland. http://www.ecology.uq.edu.au/marxan.htm [accessed 11th Feb 2004].

Ball, I.R. & Possingham, H.P. (2000) *Marxan (v1.8.2). Marine reserve deisgn using spatially explicit annealing. A manual prepared for the Great Barrier Reef Marine Park Authority*. University of Queensland, Brisbane, Australia.

Balram, S., Dragicevic, S. & Meredith, T. (2003) Achieving effectiveness in stakeholder participation using the GIS-based Collaborative Spatial Delphi methodology. *Journal of Environmental Assessment Policy and Management*, **5**(3), 365-394.

Boersma, P.D. & Parrish, J.K. (1999) Limiting abuse: marine protected areas, a limited solution. *Ecological Economics*, **31**(2), 287-304.

Bohnsack, J.A. (1996) Maintenance and recovery of reef fishery productivity. In *Reef Fisheries* (eds N.V.C. Polunin & C.M. Roberts), pp. 283-313. Chapman and Hall, London, UK.

Botsford, L.W., Hastings, A. & Gaines (2001) Dependence of sustainability on the configuration of marine reserves and larval dispersal distance. *Ecology Letters*, **4**(2), 144-150.

British Columbia (1993) *A protected areas strategy for British Columbia.* Government of British Columbia, Victoria, Canada.

Brown, K., Adger, W.N., Tompkins, E., Bacon, P., Shim, D. & Young, K. (2001) Trade-off analysis for marine protected area management. *Ecological Economics*, **37**(3), 417-434.

Burrough, P.A. & McDonnell, R.A. (1998) *Principles of Geographic Information Systems*. Oxford University Press, Inc., New York, USA.

Carver, S. (1991) Integrating multi-criteria evaluation with geographical information systems. *International Journal of Geographical Information Systems*, **5**(3), 321-339.

Ceballos-Silva, A. & Lopez-Blanco, J. (2003) Delineation of suitable areas for crops using a Multi-Criteria Evaluation approach and land use/cover mapping: a case study in Central Mexico. *Agricultural Systems*, **77**(2), 117-136.

Chen, K., Blong, R. & Jacobson, C. (2003) Towards an integrated approach to natural hazards risk assessment using GIS: With reference to bushfires. *Environmental Management*, **31**(4), 546-560.

Christensen, V., Guénette, S., Heymans, J.J., Walters, C., Watson, R., Zeller, D. & Pauly, D. (2003) Hundred-year decline of North Atlantic predatory fisheries. *Fish and Fisheries*, **4**, 1-24.

Chung, S.S. & Poon, C.S. (1996) Evaluating management alternatives by the multiple criteria approach. *Resources, Conservation and Recycling*, **17**(3), 189-210.

CITES (2004) *Convention on International Trade In Endangered Species of Wild Fauna and Flora*, CITES. http://www.cites.org/index.html [accessed 19th May 2004].

DFO (2004) *DFO's Statistical Services*, Fisheries and Oceans Canada. http://www.dfo-mpo.gc.ca/communic/statistics/main_e.htm [accessed 30th May 2004].

Eastman, J.R. (2003) *IDIRISI Kilimanjaro Guide to GIS and Image Processing*. Clark Labs, Clark University, Worcester, Massachusetts, USA.

Eastman, J.R., Jin, W.G., Kyem, P.A.K. & Toledano, J. (1995) Raster Procedures for Multicriteria / Multiobjective Decisions. *Photogrammetric Engineering and Remote Sensing*, **61**(5), 539-547.

Eastman, J.R., Kyem, P.A.K. & Toledano, J. (1993) *A procedure for multi-objective decision making in GIS under conditions of conflicting objectives*. 4th European Conference on Geographical Information Systems, 29 March - 1 April 1993, Genoa, Italy.

Garson, J., Aggarwal, A. & Sarkar, S. (2002) *ResNet Manual Ver 1.2 30 October 2002*. University of Texas at Austin, Austin, Texas, USA.

Gaston, K.J. (1996) Species richness: measure and measurement. In *Biodiversity a biology of numbers and difference.* (ed K.J. Gaston), pp. 77-113. Blackwell Science, Oxford, UK.

Hastings, A. & Botsford, L.W. (2003) Comparing designs of marine reserves for fisheries and for biodiversity. *Ecological Applications*, **13**(1), S65-S70.

Huth, A., Drechsler, M. & Kohler, P. (2004) Multicriteria evaluation of simulated logging scenarios in a tropical rain forest. *Journal of Environmental Management*, **71**(4), 321-333.

IUCN (1988) *Resolution 17.38 of the 17th General Assembly of the IUCN*. IUCN, Gland, Switzerland and Cambridge, UK.

IUCN (2003) *WPC Recommendation 5.22: Building a global system of marine and coastal protected area networks*. IUCN, Durban, South Africa.

IUCN (2007) *2007 IUCN Red List of Threatened Species*, IUCN Species Survival Commission. http://www.redlist.org/ [accessed 4th October 2007].

Jamieson, G.S. & Levings, C.O. (2001) Marine protected areas in Canada - implications for both conservation and fisheries management. *Canadian Journal of Fisheries and Aquatic Sciences*, **58**(1), 138-156.

Jankowski, P. & Richard, L. (1994) Integration of GIS-based suitability analysis and multicriteria evaluation in a spatial decision-support system for route selection. *Environment and Planning B - Planning & Design*, **21**(3), 323-340.

Janssen, R. (2001) On the use of multi-criteria analysis in environmental impact assessment in the Netherlands. *Journal of Multi-Criteria Decision Analysis*, **10**, 101-109.

Janssen, R. & Rietveld, P. (1990) Multicriteria analysis and geographic information systems: and application to agricultural land use in the Netherlands. In *Geographical information systems for urban and regional planning* (eds H.J. Scholten & J.C.H. Stillwell), Kluwer Academic Publishing, Dordrecht, Germany.

Jiang, H. & Eastman, J.R. (2000) Application of fuzzy measures in multi-criteria evaluation in GIS. *International Journal of Geographical Information Science.*, **14**(2), 173-184.

Jones, P.J.S. (2002) Marine protected area strategies: issues, divergences and the search for middle ground. *Reviews in Fish Biology and Fisheries*, **11**(3), 197-216.

Keeney, R.L. & Raiffa, H. (1976) *Decisions with muliple objectives: preferences and value tradeoffs.* Wiley, New York, USA.

Killpack, D., Waltenberger, B. & Fowler, C. (2001) *The Channel Islands - Spatial Support and Analysis Tool.* 2nd Biennial Coastal GeoTools Conference, Charleston, South Carolina, NOAA Coastal Services Center.

Kirkpatrick, S., Gelatt Jr, C.D. & Vecchi, M.P. (1983) Optimisation by simulated annealing. *Science*, **220**, 671-680.

Lewis, A., Fernandes, L., Lowe, D., Slegers, S., Nuller, L. & Day, J. (2003) *Re-zoning the Great Barrier Reef Marine Park to better protect biodiversity*. Coastal GIS Workshop, July 7-8 2003, Wollongong, Australia.

Lubchenco, J., Palumbi, S.R., Gaines, S.D. & Andelman, S.J. (2003) Plugging a hole in the ocean: the emerging science of marine reserves. *Ecological Applications*, **13**(1), S3-S7.

Malczewski, J. (1999) *GIS and Multicriteria Decision Analysis*. John Wiley & Sons, Inc., New York, USA.

Mardle, S. & Pascoe, S. (1999) A review of applications of multiple-criteria decision-making techniques to fisheries. *Marine Resource Economics*, **14**(1), 40-63.

McDonnell, M.D., Possingham, H. P., Ball, I. R., Cousins, E. A. (2002) Mathematical methods for spatially cohesive reserve design. *Environmental Modeling and Assessment*, **7**(2), 107-114.

Meir, E., Andelman, S. & Possingham, H. (2004) Does conservation planning matter in a dynamic and uncertain world? *Ecology Letters*, **7**, 615-622.

Myers, R.A. & Worm, B. (2003) Rapid worldwide depletion of predatory fish communities. *Nature*, **423**(6937), 280-283.

Oceans Act 1996 (Canada). http://www.canlii.org/ca/sta/o-2.4/whole.html [accessed 1st August 2005].

Pitz, G.F. & McKillip, J. (1984) *Decision analysis for program evaluators*. Sage Publications, Thousand Oaks, California, USA.

Possingham, H., Ball, I. & Andelman, S. (2000) Mathematical methods for identifying representative reserve networks. In *Quantitative methods for conservation biology* (ed S. Ferson, Burgman, M.), pp. 291-305. Springer-Verlag, New York, USA.

Pressey, R.L., Humphries, C.J., Margules, C.R., Vane-Wright, R.I. & Williams, P.H. (1993) Beyond opportunism: key principles for systematic reserve selection. *Trends in Ecology and Evolution*, **8**(4), 124-128.

Pressey, R.L., Possingham, H.P. & Day, J.R. (1997) Effectiveness of alternative heuristic algorithms for identifying indicative minimum requirements for conservation reserves. *Biological Conservation*, **80**(2), 207-219.

Pressey, R.L., Possingham, H.P. & Margules, C.R. (1996) Optimality in reserve selection algorithms: When does it matter and how much? *Biological Conservation*, **76**(3), 259-267.

Roberts, C.M., Branch, G., Bustamante, R.H., Castilla, J.C., Dugan, J., Halpern, B.S., Lafferty, K.D., Leslie, H., Lubchenco, J., McArdle, D., Ruckelshaus, M. & Warner, R.R. (2003) Application of ecological criteria in selecting marine reserves and developing reserve networks. *Ecological Applications*, **13**(1), S215-S228.

Roberts, C.M., McClean, C.J., Veron, J.E.N., Hawkins, J.P., Allen, G.R., McAllister, D.E., Mittermeier, C.G., Schueler, F.W., Spalding, M., Wells, F., Vynne, C. & Werner, T.B. (2002) Marine biodiversity hotspots and conservation priorities for tropical reefs. *Science*, **295**(5558), 1280-1284.

Roff, J.C. (2005) Conservation of marine biodiversity: too much diversity, too little cooperation. *Aquatic Conservation: Marine and Freshwater Ecosystems*, **15**, 1-5.

Rumsey, C., Ardron, J., Ciruna, K., Curtis, T., Doyle, F., Ferdana, Z., Hamilton, T., Heinmeyer, K., Iachetti, P., Jeo, R., Kaiser, G., Narver, D., Noss, R., Sizemore, D., Tautz, A., Tingey, R. & Vance-Borland, K. (2004) *An ecosystem analysis for Haida Gwaii, Central Coast and North Coast British Columbia.* Coast Information Team, Victoria, Canada.

Russ, G.R. & Zeller, D. (2003) From *Mare Liberum* to *Mare Reservarum*. *Marine Policy*, **27**(1), 75-78.

Soulé, M.E. & Terborgh, J. (1999) Conserving nature at regional and continental scales - a scientific program for North America. *Bioscience*, **49**(10), 809-817.

Store, R. & Kangas, J. (2001) Integrating spatial multi-criteria evaluation and expert knowledge for GIS-based habitat suitability modelling. *Landscape and Urban Planning*, **55**(2), 79-93.

Sumaila, U.R. & Walters, C. (2005) Intergenerational discounting: a new intuitive approach. *Ecological Economics*, **52**(2), 135-142.

UNCLOS (2004) *Chronological list of ratifications / accessions / successions to the United Nations Convention on the Law of the Sea*, UNCLOS. http://www.un.org/Depts/los/convention_agreements/convention_agreements.htm [accessed 30th August 2004].

United Nations (2002) *Draft plan of implementation of the World Summit on Sustainable Development*. United Nations, Johannesburg, South Africa.

Villa, F., Tunesi, L. & Agardy, T. (2002) Zoning marine protected areas through spatial multiple-criteria analysis: the case of the Asinara Island National Marine Reserve of Italy. *Conservation Biology*, **16**(2), 515-526.

Ward, T., Heinemann, D. & Evans, N. (2001) *The role of marine reserves as fisheries management tools. A review of concepts, evidence and international experience.* Bureau of Rural Sciences, Canberra, Australia.

Ward, T., Vanderklift, M.A., Nicholls, A.O. & Kenchington, R. (1999) Selecting marine reserves using habitats and species assemblages as surrogates for biological diversity. *Ecological Applications*, **9**(2), 691-698.

Watson, R., Alder, J., Kitchingman, A. & Pauly, D. (2005) Catching some needed attention. *Marine Policy*, **29**(3), 281-284.

Watson, R., Kitchingman, A., Gelchu, A. & Pauly, D. (2004) Mapping global fisheries: sharpening our focus. *Fish and Fisheries*, **5**, 168-177.

Watson, R. & Pauly, D. (2001) Systematic distortions in world fisheries catch trends. *Nature*, **414**, 534-536.

Whittaker, R.H. (1972) Evolution and measurement of species diversity. *Taxon*, **21**(2/3), 213-251.

Wood, L.J. (submitted) *Global marine protection targets: how S.M.A.R.T. are they?*

Wood, L.J., Fish, L., Laughren, J. & Pauly, D. (in press) Assessing progress towards global marine protection targets: shortfalls in information and action. *Oryx*.

Worm, B., Lotze, H.K. & Myers, R.A. (2003) Predator diversity hotspots in the blue ocean. *Proceedings of the National Academy of Sciences*, **100**, 9884-9888.

Zacharias, M.A. & Howes, D.E. (1998) An analysis of marine protected areas in British Columbia, Canada, using a marine ecological classification. *Natural Areas Journal*, **18**(1), 4-13.

6. Priority areas for a global network of marine protected areas [1]

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Introduction

Much of current activity in marine conservation is framed within the context of three global targets, adopted in the last five years. Firstly, the Johannesburg Plan of Implementation, adopted at the World Summit on Sustainable Development in 2002, committed to developing global representative networks of marine protected areas by 2012 (United Nations, 2002a). Secondly, the World Parks Congress adopted a recommendation in 2003 to strictly protect 20-30% of each marine habitat by 2012 (IUCN, 2003). Finally, in 2006, the Convention on Biological Diversity made a decision to effectively conserve 10% of the world's ecological (including marine) regions (within Exclusive Economic Zones (EEZs)) by 2010 (CBD, 2006a). However, none of these global targets are likely to be met on time, due to both the magnitude of increase in protection required as well as the imminence of the target deadlines (Wood *et al.*, in press; Wood, submitted). Since the adoption of these targets, however, no global strategy or framework has been developed to guide the identification and selection of areas for protection. This is a substantial hindrance to the attainment of the global targets, not least because it is likely to result in continued 'ad hoc' creation of MPAs, which is widely considered to be inefficient and ineffective (Possingham *et al.*, 2000; Agardy *et al.*, 2003; Stewart, 2003). The objective of this study was to use an heuristic place prioritisation algorithm to identify priority areas for protection in the world's oceans, under various scenarios that reflect the quantitative global marine protection targets adopted by the World Parks Congress and the Convention on Biological Diversity. This is by far the largest geographical area over which a place prioritization exercise has been conducted to date.

Reserve selection, more recently termed conservation area selection or place prioritization (Sarkar *et al.*, 2004; Sarkar *et al.*, 2006), is an explicit stage in systematic conservation planning (Margules and Pressey, 2000). It is the identification of locations which aim to ensure that biodiversity is adequately represented within networks of conservation areas as effectively and efficiently as possible, such that biodiversity will persist into the future (Pressey *et al.*, 1993; Margules and Pressey, 2000; Wilson *et al.*, 2005b; Possingham *et al.*, 2006; Sarkar *et al.*, 2006). Place prioritization approaches to achieving these overall goals of representation and persistence have been evolving over the last twenty five years (Kirkpatrick, 1983; Kingsland, 2002), and are centred around the concept of complementarity, as well as a suite of additional guiding principles including irreplaceability, flexibility, vulnerability, spatial economy, and computational efficiency (Pressey *et al.*, 1993; Reyers *et al.*, 2000; Williams, 2001; Faith *et al.*, 2003; Possingham *et al.*, 2006; Sarkar *et al.*, 2006).

Place prioritization algorithms have been broadly categorized as optimal, heuristic, or metaheuristic (Sarkar *et al.*, 2006). Heuristic algorithms have been criticized because they are not guaranteed to find optimal (i.e. maximally spatially economical) solutions, and the degree of suboptimality is unpredictable, varying with the dataset and the algorithm (Underhill, 1994; Rodrigues *et al.*, 2000; Rodrigues and Gaston, 2002). However, the level of sub-optimality can be very small and in some cases heuristic solutions may even match optimal solutions (Csuti *et al.*, 1997; Pressey *et al.*, 1997; Sarkar *et al.*, 2004). Furthermore, the nature of place prioritization problems is such that even relatively simple problems (which seldom have much practical relevance) may be intractable for optimal algorithms, a problem which is not necessarily alleviated by increasing computer processing speed. Heuristic algorithms, while sacrificing some spatial economy, are more computationally efficient (Sarkar *et al.*, 2006). This is especially the case for large datasets and/or complex constraint scenarios (e.g., proportional representation targets), which are more commonly encountered in practice

(Pressey *et al.*, 1996). Metaheuristics, such as simulated annealing (Kirkpatrick *et al.*, 1983; Ball and Possingham, 2000; Possingham *et al.*, 2000), may achieve greater spatial economy than heuristic algorithms, but are currently less computationally efficient (Kelley, 2002).

Place prioritization studies require that surrogates for biodiversity be used, because our knowledge of global biodiversity is incomplete, and more detailed data cannot typically be obtained in the timeframe within which landscape- or seascape-altering decisions are made (Margules and Pressey, 2000; Favreau *et al.*, 2006; Possingham *et al.*, 2006), or target deadlines have passed. Despite concerns about, and the difficulties of testing, the effectiveness of surrogates in representing total biodiversity (van Jaarsveld *et al.*, 1998; Reyers *et al.*, 2000; Favreau *et al.*, 2006), they remain a practical necessity (Flather *et al.*, 1997; Margules and Pressey, 2000). Possingham *et al* (2006) thus recommend that the best use of all available data should be made, and Beger *et al* (2003) recommend the use of surrogates with high levels of distribution heterogeneity. Surrogates can be either biological or non-biological features, or a combination of both, and must have distributions that can be easily assessed in the field or reliably modelled at a scale relevant to the scale of the planning operation (Zacharias and Howes, 1998; Ward *et al.*, 1999; Margules and Pressey, 2000; Gladstone, 2002; Sarkar *et al.*, 2004; Stevens and Connolly, 2004; Wilson *et al.*, 2005b; Rondinini *et al.*, 2006). While the global marine protection targets under consideration focus on the representation of habitats and/or ecological regions, there is as yet no habitat data or ecological classification available with both global extent and sufficiently high resolution to render a habitat-based global place prioritization exercise feasible.

Place prioritization can be formulated either as minimum-set problems or maximum-coverage problems. Minimum set problems identify the minimum area required to meet the specified targets for all surrogates, while maximum coverage problems maximize the level of surrogate representation for a given cost, for example, area (Williams, 2001; Sarkar *et al.*, 2006). Formulation of place prioritization as a maximum coverage problem is sometimes considered to be more realistic than minimum-set formulation because conservation goals are usually more heavily constrained by financial and/or spatial budgets than by notions of representation (Underhill, 1994; Williams, 2001; Sarkar *et al.*, 2006). Minimum set problems may also fail to achieve representation of biodiversity if more remote surrogates, such as habitat types or higher taxonomic groupings, are used (Williams, 2001). However, maximum coverage problems run the risk of representing some surrogates beyond that which is biologically relevant, while not achieving adequate representation of others. As such it is still advisable to set minimum representation targets for surrogates in a maximum-set problem (Sarkar *et al.*, 2004).

The process of developing targets is a complex one. Large-scale, percentage-based targets, such as those under consideration here, have been criticized for being ecologically irrelevant and often insufficient to ensure persistence of biodiversity (Soulé and Sanjayan, 1998; Pressey *et al.*, 2003; Solomon *et al.*, 2003). However, such conclusions have typically been drawn from applications of global targets directly to local situations. This is unlikely to be appropriate given the multiscale nature of both ecological systems and political organisation. Instead, a hierarchical system of targets that spans spatial and temporal scales of application is likely to be needed (Wood, submitted). In practical conservation planning, two broad rounds of analysis are recommended: firstly, identification of global priority areas, followed by regional reserve network selection within global priority areas (Vane-Wright *et al.*, 1991; Pressey *et al.*, 1993). In this context, global targets would best be applied at the stage of identifying global priority areas. To date, systematic conservation planning techniques have not yet been used for either analysis.

Methods

Data Sets

This analysis makes use of the global spatial grid system adopted by the *Sea Around Us* Project, which consists of 259,200 cells of 0.5° latitude/longitude resolution, and whose area is prorated according to sea surface area (Watson *et al.*, 2004; Sea Around Us, 2007). Cells containing no sea water, and cells whose percent mean ice cover between 1979 and 2002 was 100 (Cavalieri *et al.*, 1996, updated 2006) were excluded, leaving a total of 176093 cells.

Global distributions have recently been developed through the *Sea Around Us* Project for 923 fish and invertebrate species that have been reported by the Food and Agriculture Organisation of the United Nations (FAO) as being part of commercial fisheries catches between 1950 and 2003. The distributions were developed using a rigorous rule-based revision of geographic ranges (Close *et al.*, 2006) and are available online (Sea Around Us, 2007). Global distributions are also available for 115 species of marine mammal using the same 0.5° global grid (Kaschner *et al.*, 2006). This makes a total of 1038 species with a broad taxonomic representation (Table 6.1). Both datasets provide relative probabilities of occurrence, i.e., the probabilities across all cells for each species summed to one. However, the use of relative probabilities of occurrence is problematic here because summing them (a step in implementing the place prioritization algorithm) is based on unrealistic assumptions about the independence of probabilities of occurrence between surrogates and through space. Interpreting the relative probabilities as expectations, by dividing the relative probabilities by the maximum relative probability for that species, circumvents this problem (Sarkar *et al.*, 2006). Relative probabilities were thus converted to expectation values and truncated to a precision of 0.1. Values less than 0.1 were truncated to zero. Richness of a) all species, b) fish and invertebrates, and c) marine mammals, based on these expectation values are shown in Figures 6.1-6.3.

Place prioritisation protocols

This study made use of the rarity-complementarity heuristic algorithm encoded in the ResNet software package (Garson *et al.*, 2002). Rarity-complementarity heuristics have generally been found to produce the most spatially economical solutions (Csuti *et al.*, 1997; Pressey *et al.*, 1997). Sarkar *et al* (2004) found a complementarity-only algorithm produced more spatially economical solutions than a rarity-complementarity algorithm with probabilistic surrogate data. However, I still selected the rarity-complementarity algorithm because the differences in economy were small (0.2%), and excluding rarity considerations from the algorithm may result in under-representation of locations that, on the basis of the data used, currently appear to be species-depauperate but, as more data come available, may in fact not be, as may be the case in the Antarctic (Brandt *et al.*, 2007).

I developed five scenarios to investigate the WPC and CBD targets, and variations thereof, as minimum-set problems. I used minimum-set problems because I did not want to constrain the total area available for MPAs at this stage of the global planning process. Although the targets pertain specifically to habitat types or ecological regions, I used the 1038 global species distributions described earlier as surrogates for biodiversity in all scenarios. This is because of both the lack of habitat data at sufficient resolution and with global extent, and also to avoid difficulties with minimum-set problems failing to represent biodiversity when using more remote surrogates (see Introduction). Two scenarios were configured as direct interpretations of the World Parks Congress (WPC) targets, with representation targets of 20% and 30% of all species' global distributions, and a study area of the world's oceans. Two scenarios addressed the Convention on Biological Diversity (CBD) target. The study area for both was the combined area contained within the EEZs of the maritime countries of the world, as specified by the target. The first scenario set a representation target of 10% of the combined within-EEZ distributions of all species, and is intended as a direct interpretation of the CBD target. The second set a target of 10% of the global distributions of all species. The purpose of this scenario was to investigate the possibility of representing 10% of species' global distributions completely within EEZs. This is of interest because, although progress has been made to develop the legal framework necessary to implement MPA networks on the high seas (i.e., the area beyond EEZs, which constitutes around 56% of the surface area of the world's oceans), it is not yet in place (Gjerde and Kelleher, 2005; Norse and Crowder, 2005; Sumaila *et al.*, 2007). Finally, a fifth scenario was investigated, with a representation target of 10% of all species' global distributions and a study area of the world's oceans. This last scenario is intended as an extension of the CBD target to the world's oceans, since the CBD target also recommends that areas beyond national jurisdiction be afforded urgent and increased protection through international cooperation and action (CBD, 2005). It is also intended as a precursor to the higher percentage targets recommended by the World Parks Congress. Each of the five scenarios was then also re-run using fish and invertebrate distributions only, i.e., excluding marine mammals, in order to investigate the effect of the wide-ranging annual movements of many marine mammal species (Kaschner *et al.*, 2006) on the size and spatial configuration of the solutions. This resulted in a total of ten scenarios (Table 6.2).

There are usually many solutions to a particular place prioritisation problem (Pressey *et al.*, 1994), but there are often practical constraints to implementing some of them. Providing a range of flexible solutions can facilitate the decision-making process (Pressey *et al.*, 1993; Possingham *et al.*, 2000). However, heuristic solutions typically only produce a single solution (Leslie *et al.*, 2003; Sarkar *et al.*, 2006). Alternative solutions can be produced by the algorithm used here by using multiple, randomly re-ordered input files (Sarkar *et al.*, 2002; Fuller *et al.*, 2006). I produced 100 randomised input files in order to generate 100 alternative solutions for each scenario. Existing MPAs were not locked into the solutions for two reasons. Firstly, existing MPAs have largely been created in an ad hoc manner (Agardy *et al.*, 2003; Agardy, 2005), and the inclusion of them in systematic conservation planning processes has been shown to result in less efficient solutions (Stewart, 2003; Stewart *et al.*, 2007). Secondly, the vast majority of MPAs are much smaller than the cell size used here (Wood *et al.*, in press), so locking in entire cells would be inappropriate, and may have a compounding effect on the inefficiency resulting from the ad hoc nature of their creation. Finally, the optional redundancy rule of ResNet was also invoked in all runs. This part of the algorithm iterates over the final selection of cells to check for redundancy, i.e., that cells selected earlier in the process were not made redundant in terms of the representation targets by subsequent cell selections. Redundant cells are removed from the solution, improving the spatial economy of the final solution (Garson *et al.*, 2002).

Pilot tests indicated that the processing time required to generate 100 solutions for each of the 10 scenarios listed in Table 6.2 would be on the order of four years if run serially on a single PC. In order to generate solutions in a more timely manner, I ran the jobs on the Glacier cluster of 1680 computing nodes provided through WestGrid, a high performance computing, collaboration and visualization infrastructure for western Canada (WestGrid, 2007). This reduced the real-time processing to a few weeks. For each scenario, the 100 solutions were summed to produce maps showing frequency of selection of each cell, which is sometimes referred to as irreplaceability (Pressey *et al.*, 1994). Finally, the degree of spatial overlap between cells selected 100% of the time for solutions developed including and excluding marine mammals was measured using Jaccard's coefficient of similarity (Rice and Belland, 1982).

Results

Table 6.3 provides summary statistics for the solutions identified under each scenario. Maps showing the frequency of cell selection for the 100 solutions generated for each scenario are shown in Figures 6.4-6.13 inclusive. The WPC targets to represent 20-30% of all species (WPC20_All and WPC30_All) can be met by protecting 73-110 million km^2, or 20-30% of the world's oceans, respectively. Under both scenarios, around half of the selected area falls inside EEZs, representing 26-36% of the global combined EEZ area (Figures 6.4 & 6.5). The CBD target of 10% of the distribution of species within EEZs (CBD_All) can be met for all species in 15 million km^2, representing 4% of the world's oceans and 10% of global combined EEZ area (Figure 6.6). The modified CBD target of 10% of species' global distributions (CBD_GAll) can be met for 1022 species in 37 million km^2, or 10% of the world's oceans and 25% of the global combined EEZ area (Figure 6.7). This target could not be met for 16

Antarctic species (2 marine mammals, 2 invertebrates, and 12 fish), because less than 10% of their global distributions occur within EEZs. Finally, the target of 10% of all species' global distributions in the world oceans (Global10_All) can be met within 37 million km^2, or 10% of the world's oceans (Figure 6.8). However, in contrast to CBD_GAll, where the entire solution fell within EEZs, only 57% of this solution occurred within EEZs, representing 14% of the global combined EEZ area.

Exclusion of marine mammal distributions from the scenarios resulted in significantly smaller solutions, on average, by about 27% ($p<0.05$, paired t-test; Figures 6.9-6.13). The overlap in the spatial configuration of solutions generated for a given target scenario using a) all species or b) fish and invertebrates is quite low, with Jaccard's coefficients ranging from 0.32 to 0.42 (Table 6.4). Expressed differently, the average overlap area represents 45% of the mean solution area for scenarios including marine mammals, and 60% of mean solution area for scenarios excluding marine mammals (Table 6.4; see also Figures 6.4-6.13). To illustrate this, I combined and remapped the cells selected 100% of the time under scenarios WPC30_All and WPC30_Fi (i.e., Figures 6.8 and 6.9), but differentiating between the cells selected 1) for fish and invertebrates only, 2) for marine mammals only, and 3) all species (Figure 6.14).

Discussion

This study is the largest area over which systematic conservation planning techniques have been applied to date, and the first time that they have been used to identify global priorities for protection in the world's oceans. It is also the first globally synthetic test of the species distribution data used. The results generated for the World Parks Congress scenarios, using all species, provide the broadest indication of what a truly global network of MPAs might look like, with around half of the area selected falling in the high seas. The solutions also make intuitive sense, in that they highlight some major oceanographic and bathymetric phenomena that broadly identify areas of high productivity, diversity, and ecosystem value, such as: coastal areas; reefs, such as the Coral Triangle; upwelling regions, e.g. the Benguela Current system; and seamounts, for example, the Azores (Shannon *et al.*, 1988; Pauly and Christensen, 1995; Costanza, 1999; Probert, 1999; Roberts *et al.*, 2002; Worm *et al.*, 2003; Rogers, 2004). However, it is worth noting that selected areas are not necessarily species-rich, for example parts of the Sea of Okhotsk, the coast of Greenland, and south of French Polynesia. Similarly, not all species-rich areas are selected, such as the coastal area of the Great Australian Bight, and parts of the Mediterranean. This is attributable to the rarity-complementarity basis of the algorithm used, which selects areas that contain the next rarest surrogate and whose biodiversity content is most different from previously selected cells (Garson *et al.*, 2002).

Despite the spatial restriction of CBD targets to areas within EEZs, even a target of 10% of species' global distributions is largely attainable. However, Antarctic waters and many oceanic features are completely unrepresented in these solutions, meaning that the biodiversity associated with these habitats is also under-represented (Figures 6.6 & 6.7). Indeed, even based on the surrogate data set, it was not possible to represent all species by constraining the study area to EEZs. Furthermore, given the ocean-wide distances between EEZs, and therefore selected areas, solutions such as those shown in Figure 6.7 are unlikely to function as effectively as their global ocean equivalent (Figure 6.8) as a connected network of MPAs. To some extent, the possibility of protecting 10% of all species' global distributions purely by designating MPAs within EEZs may be appealing, as it provides an alternative route to achieving higher representation targets more quickly while the legal complexities of creating functional MPAs on the high seas are resolved. However, in light of the representation and connectivity challenges likely to be faced by a network of MPAs created exclusively within EEZs, high seas MPAs should still be actively pursued, through, for example, strengthening of international governance mechanisms such as Regional Fisheries Management Organisations (Foster *et al.*, 2005; Gjerde and Kelleher, 2005; Grant, 2005). Furthermore, the costs to global high seas fisheries of creating MPAs are relatively low. Although the distribution of costs between countries is likely to be uneven, borne largely by countries that are already fishing the high seas (Sumaila *et al.*, 2007), the costs of creating MPAs exclusively within EEZs is likely to be higher in absolute terms, borne by more countries, and potentially with even less equitable distribution (Figure 6.7).

The inclusion of marine mammals in the ResNet runs resulted in solutions that are significantly larger than solutions using fish and invertebrates only, and the overlap between the two is limited. Given the highly migratory nature of marine mammals, and therefore generally transitory occupancy of a given location (Kaschner *et al.*, 2006), the substantial increase in area required to meet representation targets for marine mammals raises questions over the appropriateness of MPAs as a tool for their conservation, especially given the limited resource base for implementing them (Boersma and Parrish, 1999; Norse and Crowder, 2005). Detailed exploration of these issues is somewhat beyond the scope of this chapter; however, it is conceivable that overlap between solutions generated with and without marine mammals may serve as an alternative type of prioritization (see Figure 6.14 for an example of this). Although Figure 6.14 no longer shows an efficient solution to a specific minimum-set problem, it could be used to identify areas which are high priority for high levels of protection (areas of overlap between solutions), as well as candidate areas for different management or regulatory regimes, including, for example, dynamic MPAs for areas selected primarily for marine mammals (and, indeed, other wide-ranging species) (Hyrenbach *et al.*, 2000; Hoyt, 2005).

Setting these questions aside, however, the inclusion of marine mammals is important to this analysis in terms of addressing the broader goal of representing biodiversity in a global network of MPAs. Furthermore, the inclusion of marine mammals goes some way to offsetting obvious shortcomings of the fish and invertebrate distribution data, specifically the substantial data gaps in the Southern Ocean (Figure 6.2), and potential biases resulting from them being commercially caught species. The inclusion of marine mammals thus broadens the taxonomic and distributional range of biodiversity surrogates used, and makes this study a proactive, or strategic, approach to conservation planning, rather than a reactive one, which would tend to focus on individual species, for example those at imminent risk of extinction (Williams, 2001). By the same token, persistence of biodiversity into the future is related to its vulnerability to current or impending threatening processes (Pressey *et al.*, 1996; Wilson *et al.*, 2005a). Fishing is currently the biggest and most imminent threat to marine ecosystems (Preikshot and Pauly, 2005) and so the use of primarily commercially caught species in this study represents an explicit attempt to incorporate vulnerability considerations into the priority-setting process. Beyond this, ensuring persistence of biodiversity through time also means incorporating spatiotemporally dynamic ecological, evolutionary and socio-political processes into systematic conservation planning, a challenge which is at the frontier of research in this field, but which should largely be addressed at more local scales than this study (Vane-Wright *et al.*, 1991; Pressey *et al.*, 1993; Balmford *et al.*, 1998; Nicholls, 1998; Araújo and Williams, 2000; Cabeza and Moilanen, 2001; Sarkar *et al.*, 2006).

In all scenarios investigated, most cells were selected 100% of the time, i.e., there were relatively low levels of flexibility between the 100 solutions identified for each scenario. However, high flexibility may actually be undesirable in the early stages of a global planning process, where the goal is to identify priority areas within which MPA network design processes will be undertaken. Indeed, the fundamental purpose of prioritisation is to reduce, or at the very least, order, choices, but certainly not to increase them. Nevertheless, it is worth exploring the possible reasons for the low flexibility observed, because the number of possible solutions to systematic conservation planning problems typically increases with problem size (Pressey *et al.*, 1993; Arthur *et al.*, 1997; Csuti *et al.*, 1997). However, this effect is weakened when the algorithms are using real numbers with many possible values such as selection unit (cell) area or number of features per unit (Pressey *et al.*, 1997). In this case, cell area varied over 4 orders of magnitude from ~3km^2 to >3000km^2. In addition, a large number of surrogates were used, represented by expectations of occurrence (rather than presence/absence), which increases the range of possible biodiversity content scores for a given cell. Precision of surrogate expectation values during pilot tests was 10^{-8}, but was reduced to 10^{-1} because solutions generated using the former were almost completely identical, and the use of such high levels of precision is hard to justify ecologically given the resolution of the data and uncertainties associated with the species distributions (Close *et al.*, 2006). In addition, it is also possible that the relatively low flexibility observed may be at least partly attributable to the somewhat (albeit necessarily) coarse resolution of the species distributions used, which might best be viewed as refined extents of occurrence, as opposed to areas of occupancy (Gaston, 1991). Patterns of biodiversity, and therefore complementarity, are dependent on the spatial scale at which they are measured, and this in turn affects the selection of sites by the algorithm (Williams, 2001; Warman *et al.*, 2004). For example, flexibility decreases as the number of rare species increases (Cabeza and Moilanen, 2001). Decreasing resolution effectively increases the number of rare species, as well as decreasing the number of cells in which they occur (Pressey *et al.*, 1993). This increases the number of cells that are required in all solutions to ensure their representation targets are met.

The various considerations discussed here pertaining to surrogate choice, data sources, scale, targets, and gaps, all point to the main conclusion that none of the solutions presented here should be viewed as definitive MPA network configurations. The maps simply illustrate hypothetical priorities identified at a global scale, designed to match the scale of the targets that they are addressing, and that represent known biodiversity efficiently. Furthermore, as alluded to earlier, these results make no assessment of the costs or political feasibility of implementing them, or how equitably the costs are distributed between countries. Political will is a critical requirement for the global targets to be implemented on time (Wood, submitted), but this can be reduced when broad objectives for resource are in conflict, as is the case for an up to 375-fold expansion of the current global MPA network, while maintaining or even increasing fishing activities (Wood and Dragicevic, 2007). Multicriteria decision-making (MCDM) techniques can be used to explore decision-makers preferences for different alternative solutions to a particular resource allocation problem (Malczewski, 1999; Wood and Dragicevic, 2007). They are becoming incorporated into the design of conservation area networks, although at generally smaller scales than global priority-setting exercises. (Fuller *et al.*, 2006; Moffett *et al.*, 2006; Moffett and Sarkar, 2006).

Finally, it is worth noting that none of the products of these analyses necessarily result in conservation action. Indeed, despite the increase in use of conservation area selection algorithms over the last 2 decades, only two conservation area networks designed using them have been implemented fully, in Tasmania (Kirkpatrick, 1983) and the Great Barrier Reef (Fernandes *et al.*, 2005). This is analogous to the almost ubiquitous failure of formal planning in the business world (Wildavsky, 1973; Mintzberg, 1992), yet, ironically, the business model is currently viewed as the best model for conservation planning (Margules and Pressey, 2000; Possingham, 2001; Leslie *et al.*, 2003). This is not to say that ad hoc approaches to MPA designation are necessarily better, particularly in terms of representing biodiversity efficiently (Stewart, 2003), but the limitations of the reductionist approaches inherent in formal planning need to be more explicitly recognised. The analytic approach provides not a solution, but a perspective, another way to look at the problem (Mintzberg, 1992). The critical, yet least well understood, part of planning processes is that where ideas are translated into behaviour and action (Wack, 1985; Mintzberg, 1992). The changes in behaviour needed for the global targets to be met on time is at least partly dependent on the perception that they are attainable, which is itself a function of political will and adequate information and resources (Wood, submitted). It is hoped that the analysis presented here will help to motivate some of the necessary behavioural changes, by illustrating how the 361 million km^2 of the world's oceans can be reduced to a more manageable area for focused MPA network design and implementation efforts.

Tables

Table 6.1Taxonomic summary of species whose distributions were used in ResNet analyses.

Fish		**No. species**
	Pelagic fish	159
	Demersal fish	231
	Bathypelagic fish	17
	Bathydemersal fish	34
	Benthopelagic fish	117
	Reef-associated fish	58
	Sharks and rays	83
Total fish		**699**
Invertebrates		
	Crustaceans	117
	Molluscs	99
	Echinoderms	4
	Other invertebrates	4
Total invertebrates		**224**
Marine Mammals		
	Baleen whales	14
	Toothed whales	69
	Pinnipeds	32
Total marine mammals		**115**
Total species		**1038**

Table 6.2 Attributes of the ten scenarios investigated. WPC = World Parks Congress, CBD = Convention on Biological Diversity.

Scenario	Official target	Study area	# cells	# surrogates	Target (%)	Notes
WPC20_All	WPC	World	176093	1038	20	All species
WPC20_Fi	WPC	World	176093	923	20	Fish & invertebrates only
WPC30_All	WPC	World	176093	1038	30	All species
WPC30_Fi	WPC	World	176093	923	30	Fish & invertebrates only
CBD_All	CBD	EEZs only	70305	1038	10	All species Target is 10% of species distributions contained within EEZs
CBD_Fi	CBD	EEZs only	70305	923	10	Fish & invertebrates only Target is 10% of species distributions contained within EEZs
CBD_GAll	CBD	EEZs only	70305	1038	10	All species Target is 10% of global species distributions
CBD_GFi	CBD	EEZs only	70305	923	10	Fish & invertebrates only Target is 10% of global species distributions
Global10_All	n/a	World	176093	1038	10	All species
Global10_Fi	n/a	World	176093	923	10	Fish & invertebrates only

Table 6.3 Summary statistics for solutions found for each of the 10 scenarios investigated. See Table 6.2 for more details on the scenarios. Values presented are means based on 100 solutions generated for each scenario.

Scenario	Solution statistics (mean values)			
	Area (million km^2)	% of world ocean in solution	% of solution within EEZs	% of EEZs in solution
WPC20_All	73.4	20	52	26
WPC20_Fi	52.5	15	46	17
WPC30_All	109.5	30	48	36
WPC30_Fi	80.4	22	46	26
CBD_All	14.9	4	100	10
CBD_Fi	11.1	3	100	8
CBD_GAll	36.5	10	100	25
CBD_GFi	28.3	8	100	19
Global10_All	37.0	10	57	14
Global10_Fi	25.5	7	47	8

Table 6.4 Area of overlap between solutions including and excluding marine mammal distributions for each target scenario. Data are based on cells selected in 100% of solutions for each scenario, and are expressed as absolute area, as well as percentage of the mean solution area for that scenario, both including and excluding marine mammal distributions.

Scenario (target)	Overlap area (million km^2)	Percent of mean solution area that overlaps with the solution		Jaccard's coefficient of similarity
		Scenario (species composition)		
		All species	Fish & invertebrates	
WPC20	30.2	41.1	57.5	0.32
WPC30	47.1	43.0	58.6	0.33
CBD	6.5	43.5	58.4	0.33
CBD_G	19.3	52.8	68.1	0.42
Global10	16	43.3	62.8	0.34

Figures

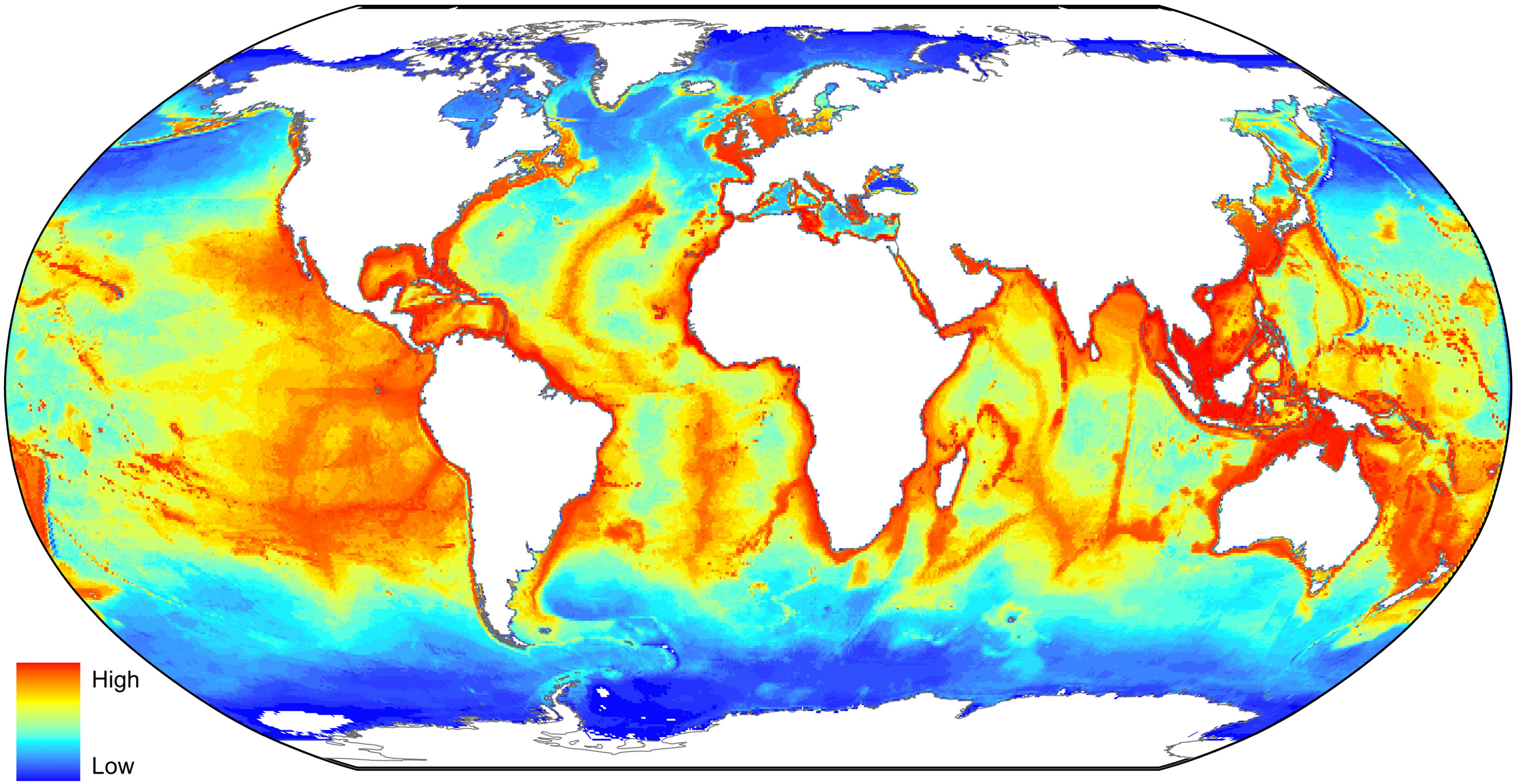

Figure 6.1 Total richness of 1038 fish, invertebrate, and marine mammal species, calculated using expected occurrence data (see text). White areas of sea in the Arctic and Antarctic are where there are no data.

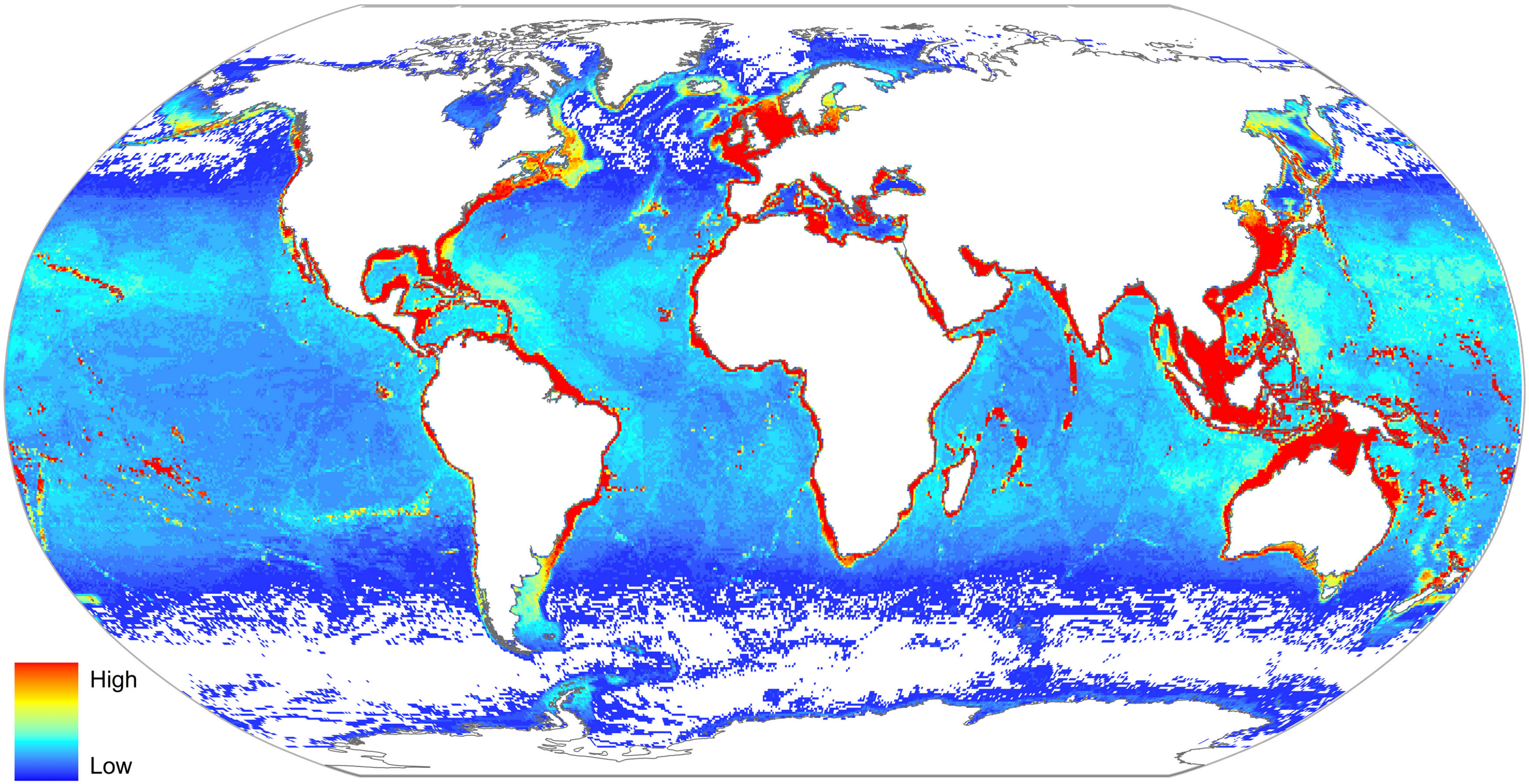

Figure 6.2 Richness of 923 commercially targeted fish and invertebrate species. White areas of sea in high latitudes are areas where there are no data.

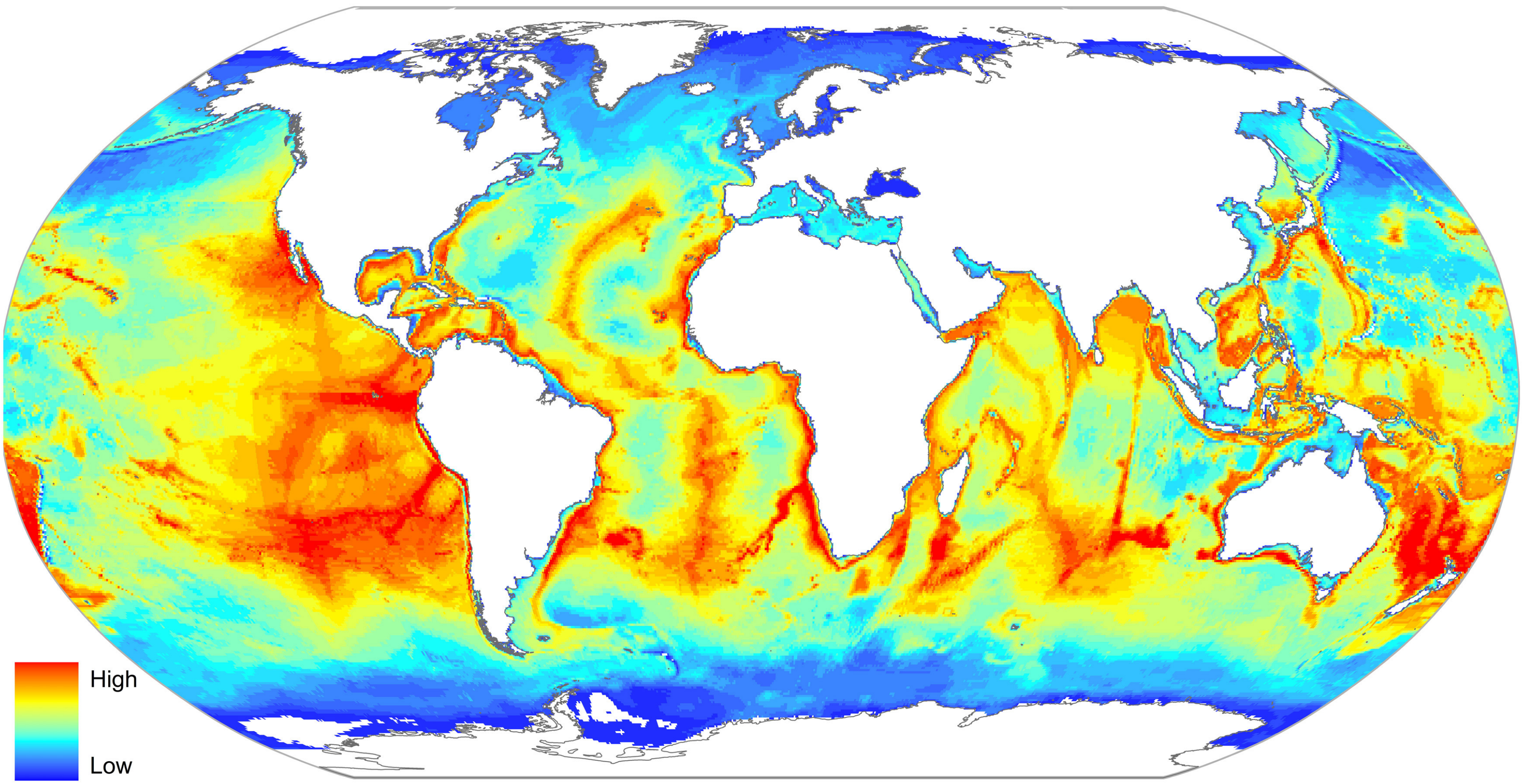

Figure 6.3 Richness of 115 marine mammal species. White areas of sea in the Arctic and Antarctic are where there are no data.

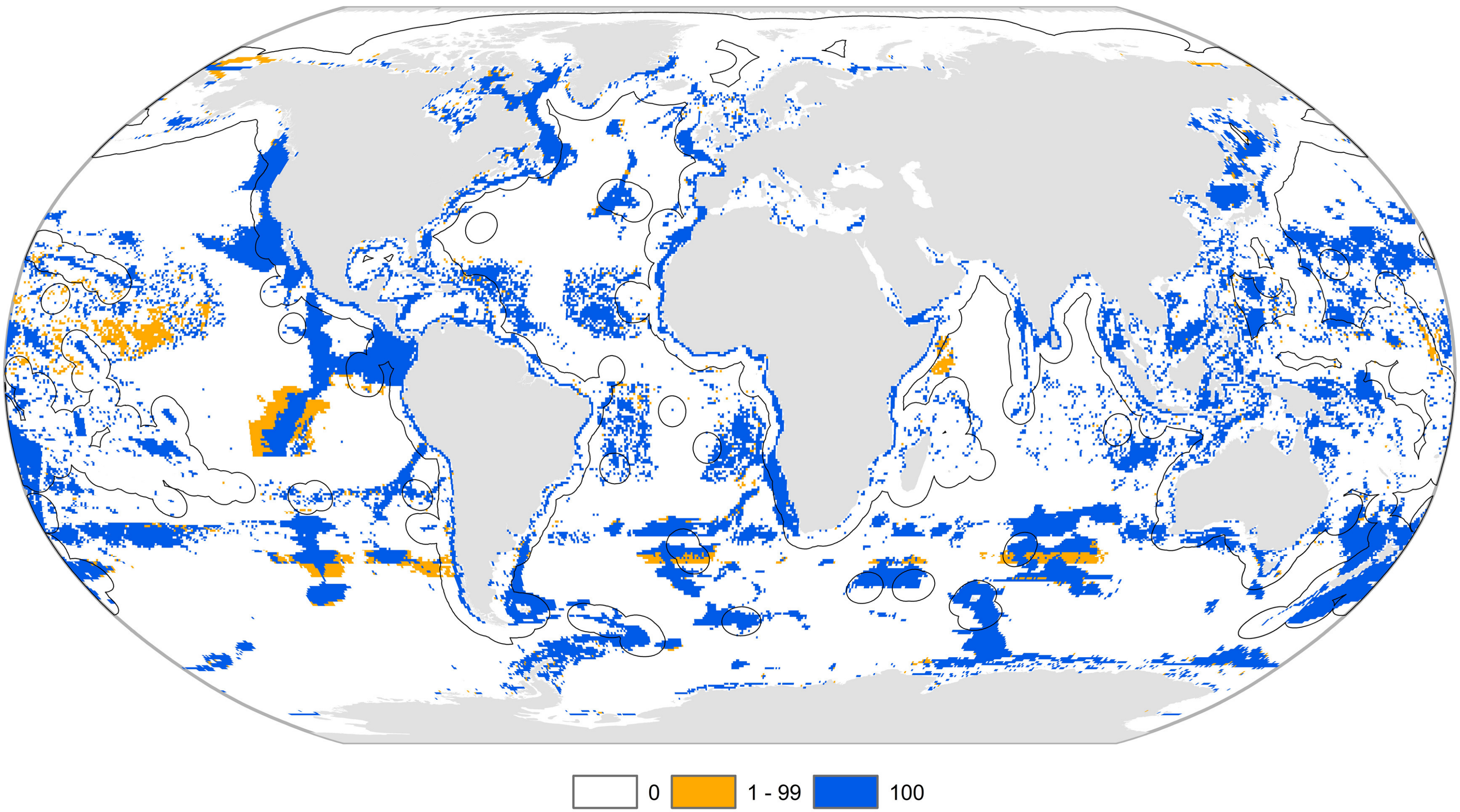

Figure 6.4 Frequency of selection of cells over 100 iterations of ResNet for scenario WPC_20All. Approximate boundaries of the 200nm Exclusive Economic Zone (EEZ) boundaries are shown in black.

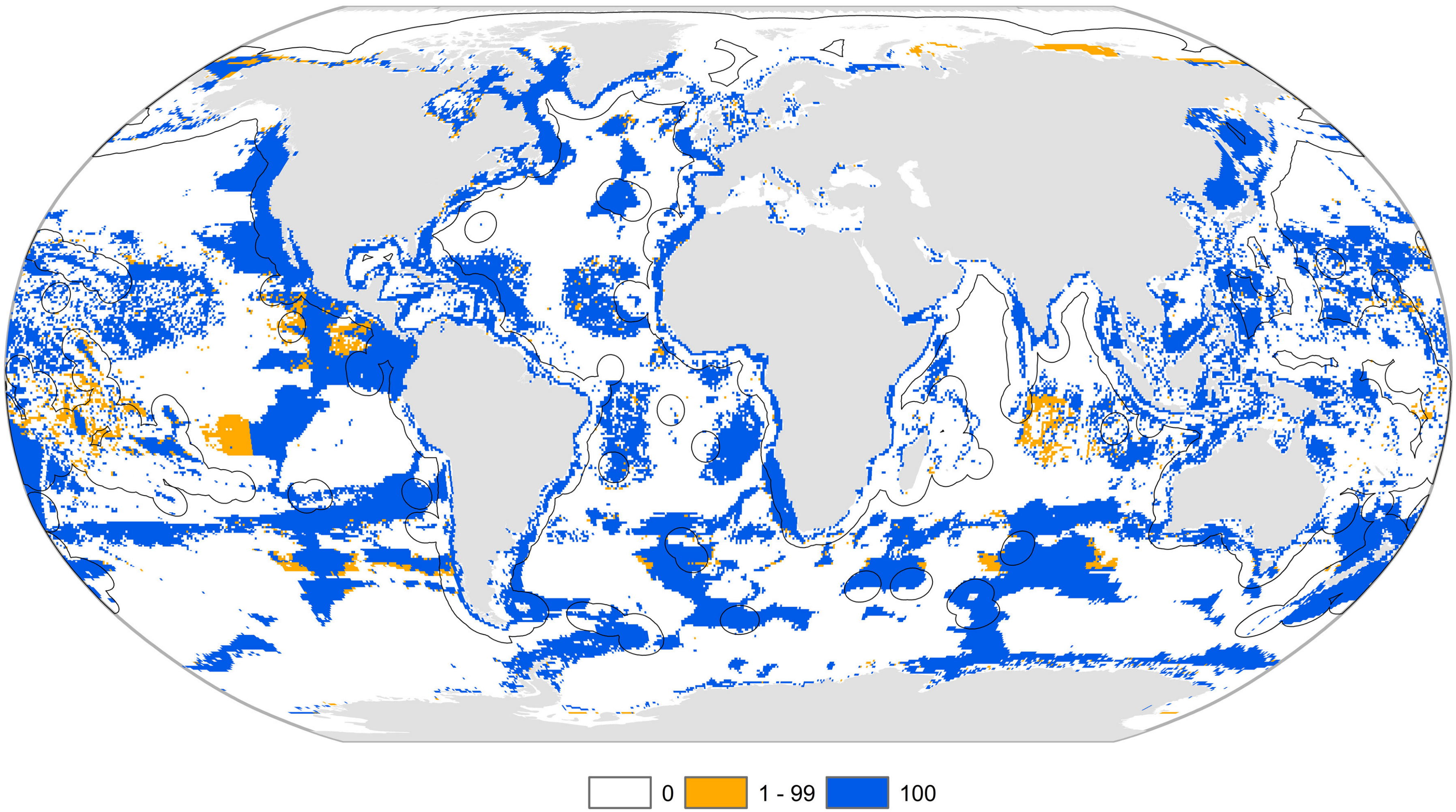

Figure 6.5 Frequency of selection of cells over 100 iterations of ResNet for scenario WPC_30All. Approximate boundaries of the 200nm Exclusive Economic Zone (EEZ) boundaries are shown in black.

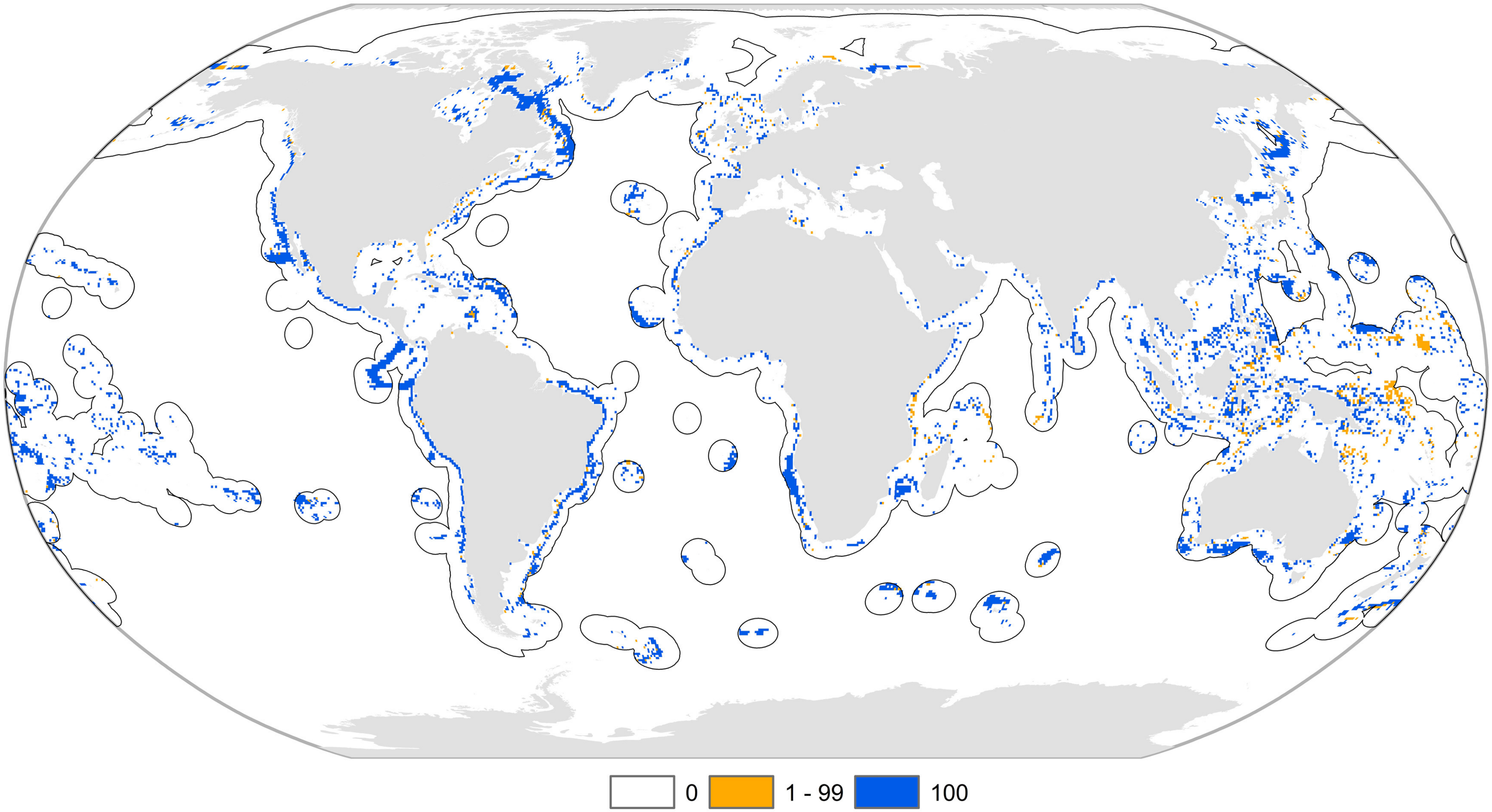

Figure 6.6 Frequency of selection of cells over 100 iterations of ResNet for scenario CBD_All. Approximate boundaries of the 200nm Exclusive Economic Zone (EEZ) boundaries are shown in black.

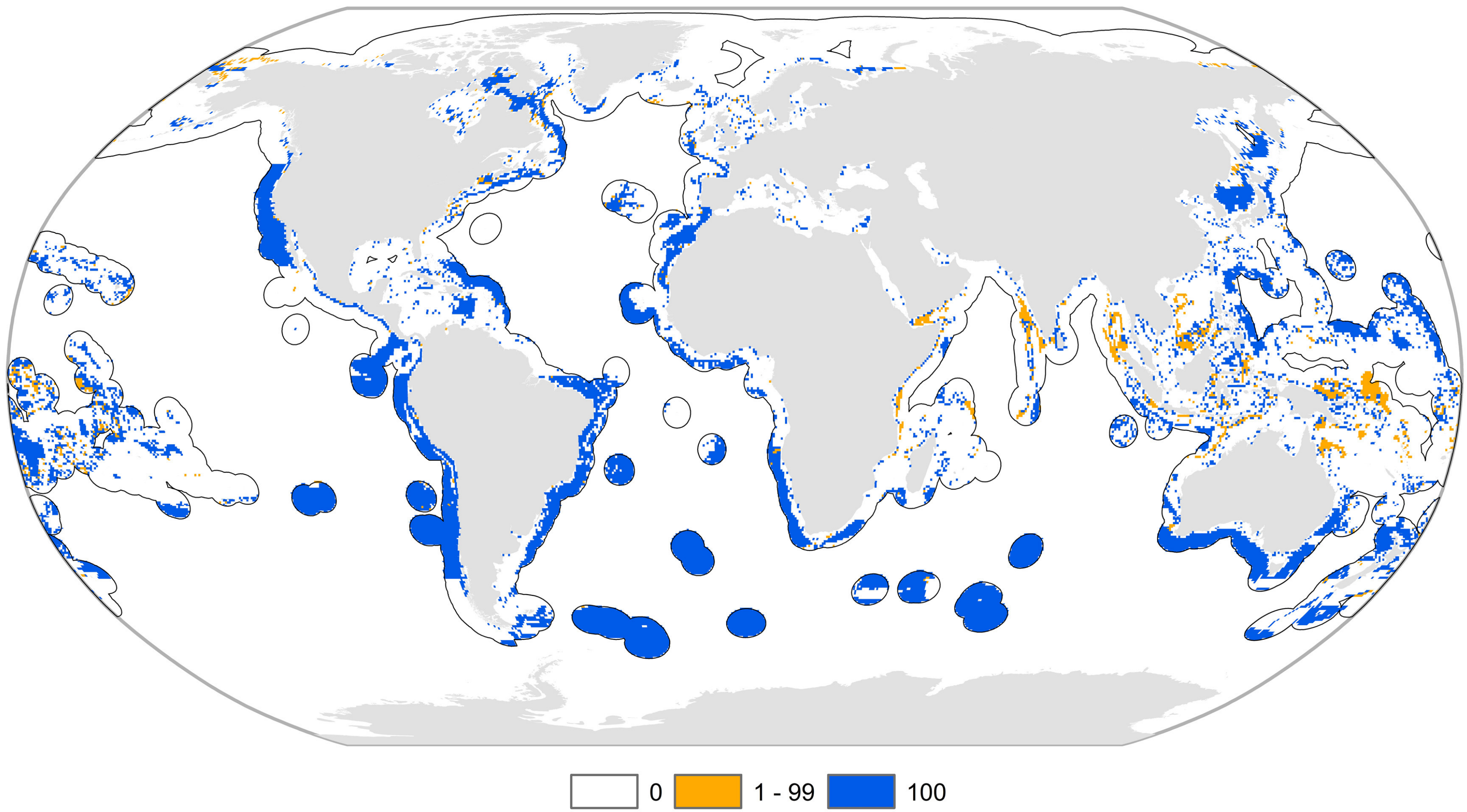

Figure 6.7 Frequency of selection of cells over 100 iterations of ResNet for scenario CBD_GAll. Approximate boundaries of the 200nm Exclusive Economic Zone (EEZ) boundaries are shown in black.

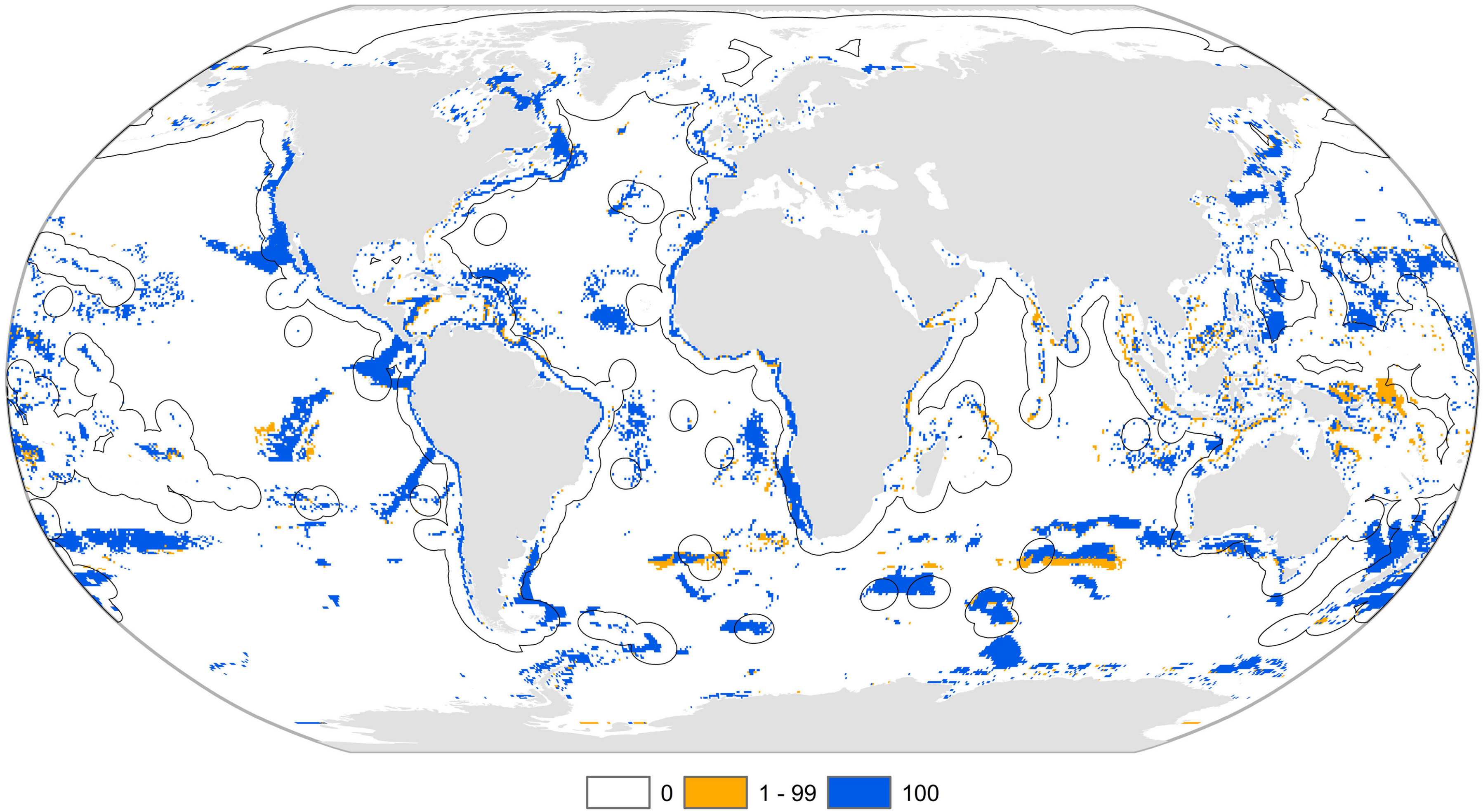

Figure 6.8 Frequency of selection of cells over 100 iterations of ResNet for scenario Global_10All. Approximate boundaries of the 200nm Exclusive Economic Zone (EEZ) boundaries are shown in black.

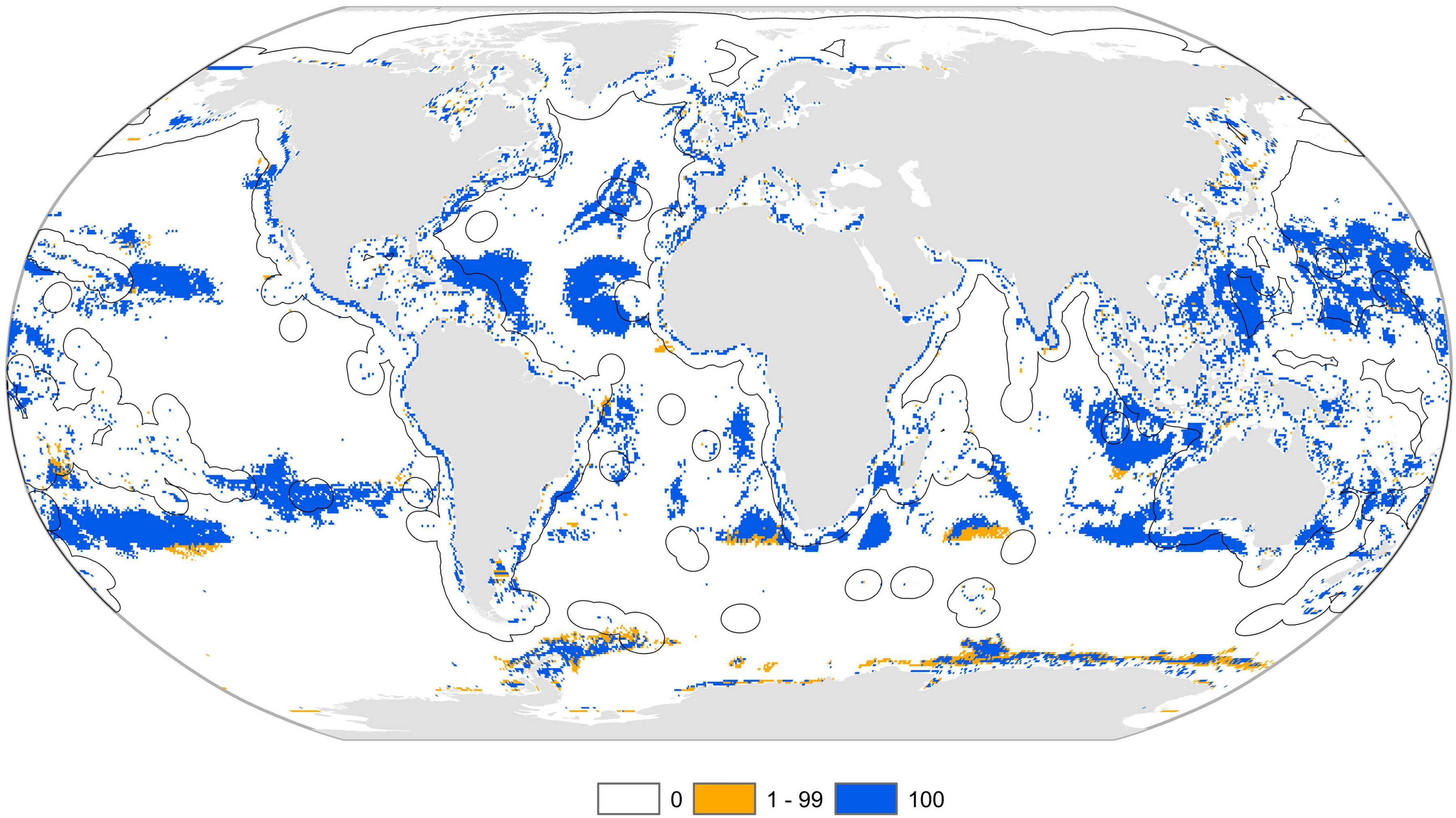

Figure 6.9 Frequency of selection of cells over 100 iterations of ResNet for scenario WPC_20Fi. Approximate boundaries of the 200nm Exclusive Economic Zone (EEZ) boundaries are shown in black.

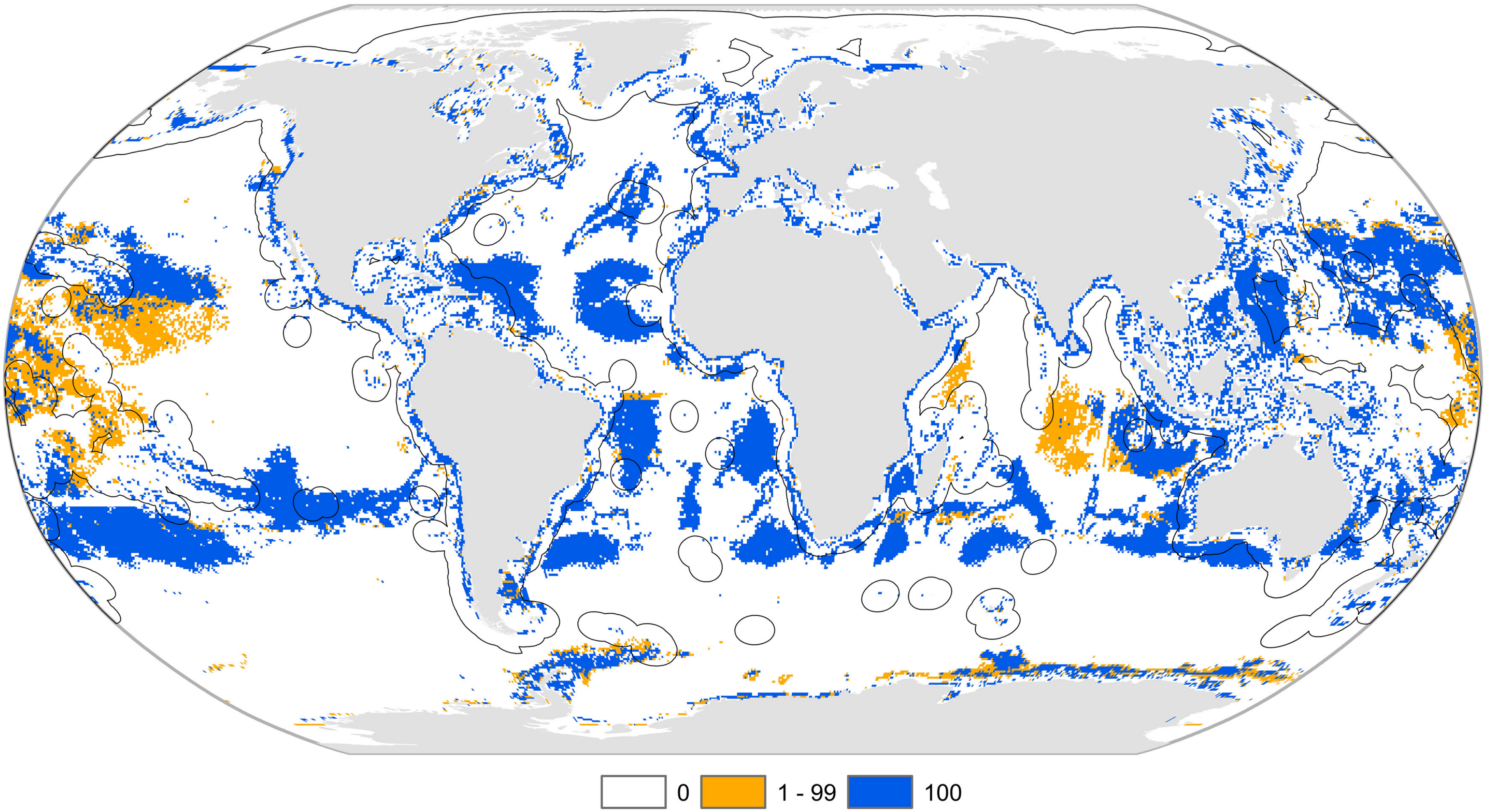

Figure 6.10 Frequency of selection of cells over 100 iterations of ResNet for scenario WPC_30Fi. Approximate boundaries of the 200nm Exclusive Economic Zone (EEZ) boundaries are shown in black.

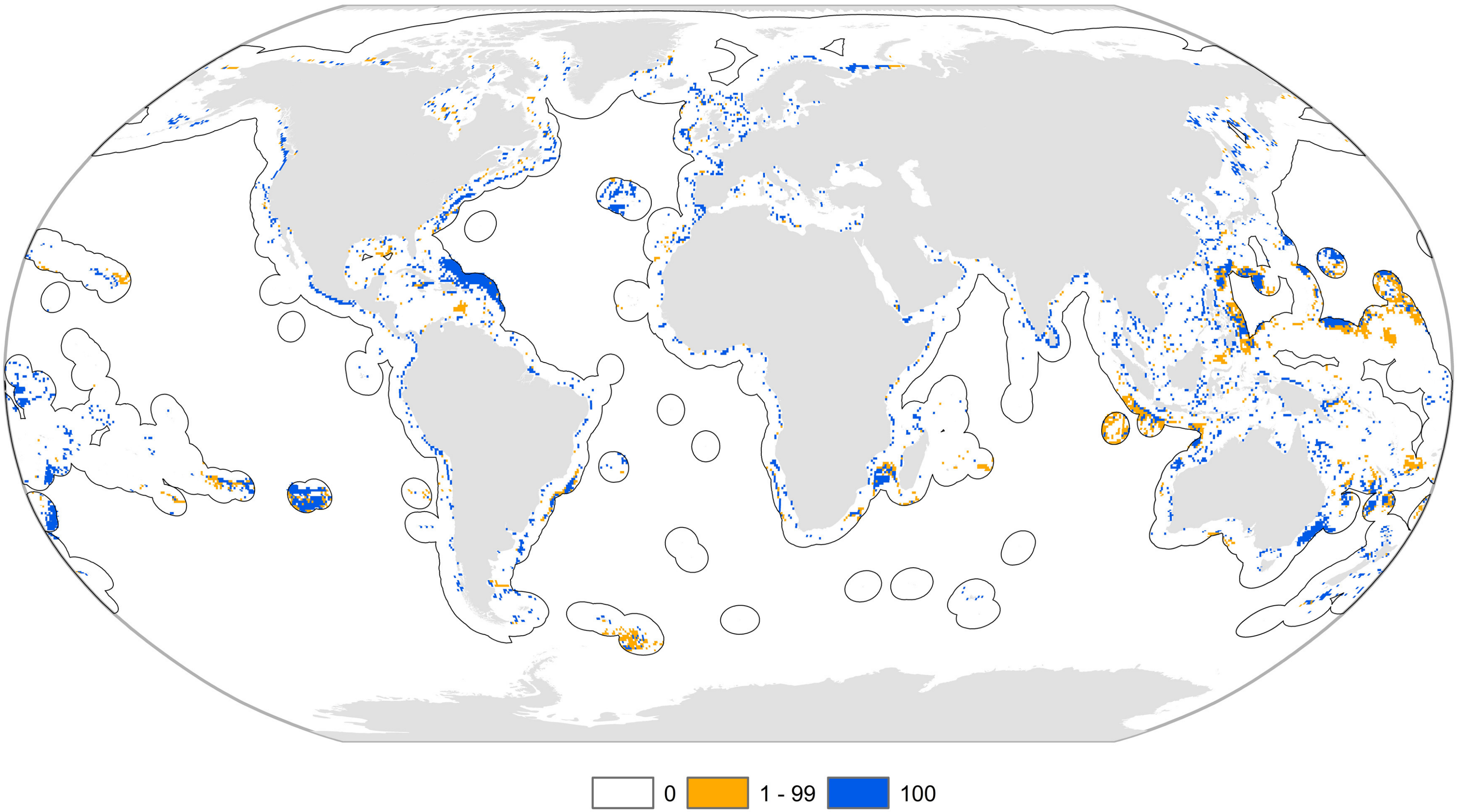

Figure 6.11 Frequency of selection of cells over 100 iterations of ResNet for scenario CBD_Fi. Approximate boundaries of the 200nm Exclusive Economic Zone (EEZ) boundaries are shown in black.

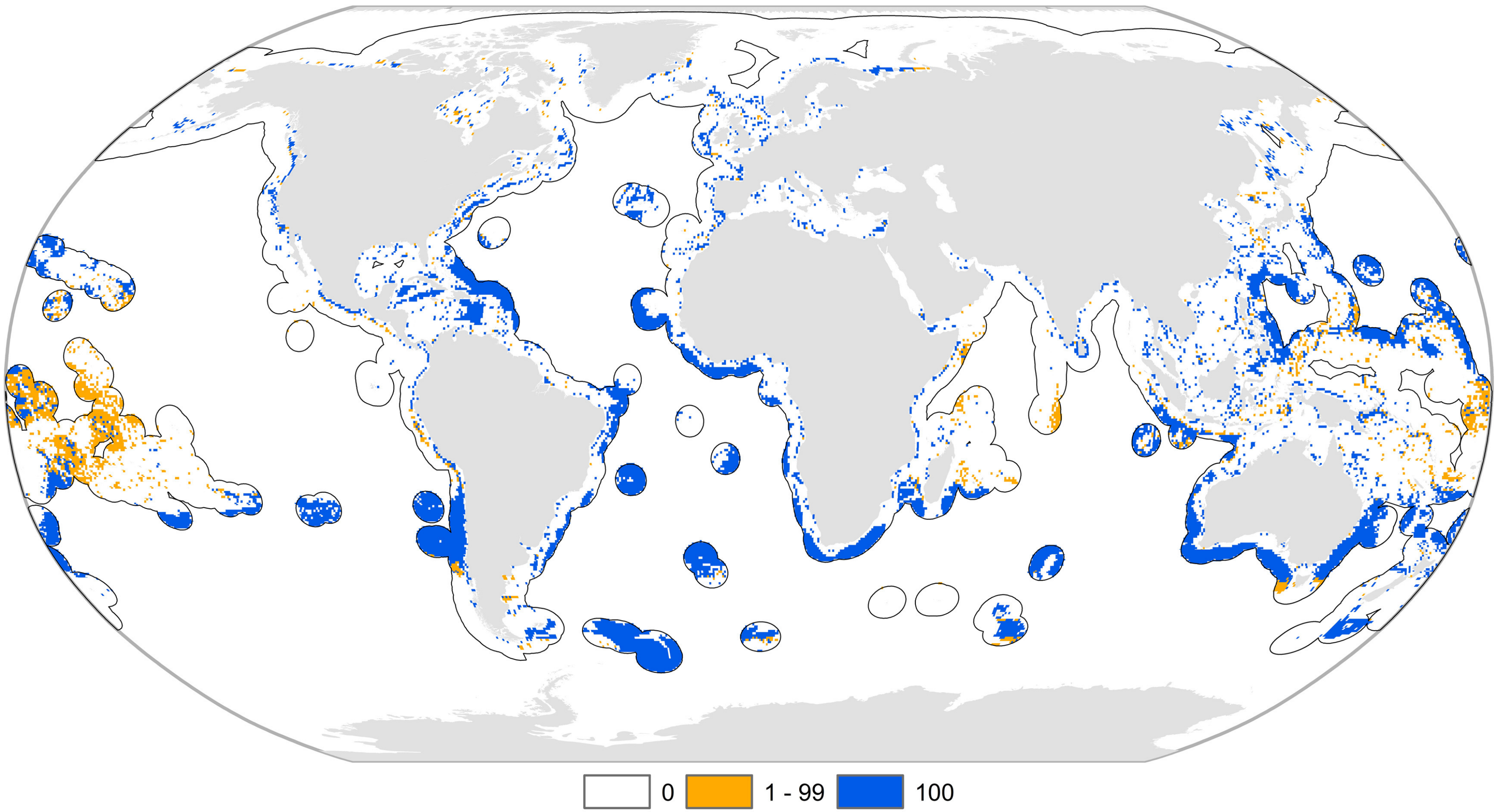

Figure 6.12 Frequency of selection of cells over 100 iterations of ResNet for scenario CBD_GFi. Approximate boundaries of the 200nm Exclusive Economic Zone (EEZ) boundaries are shown in black.

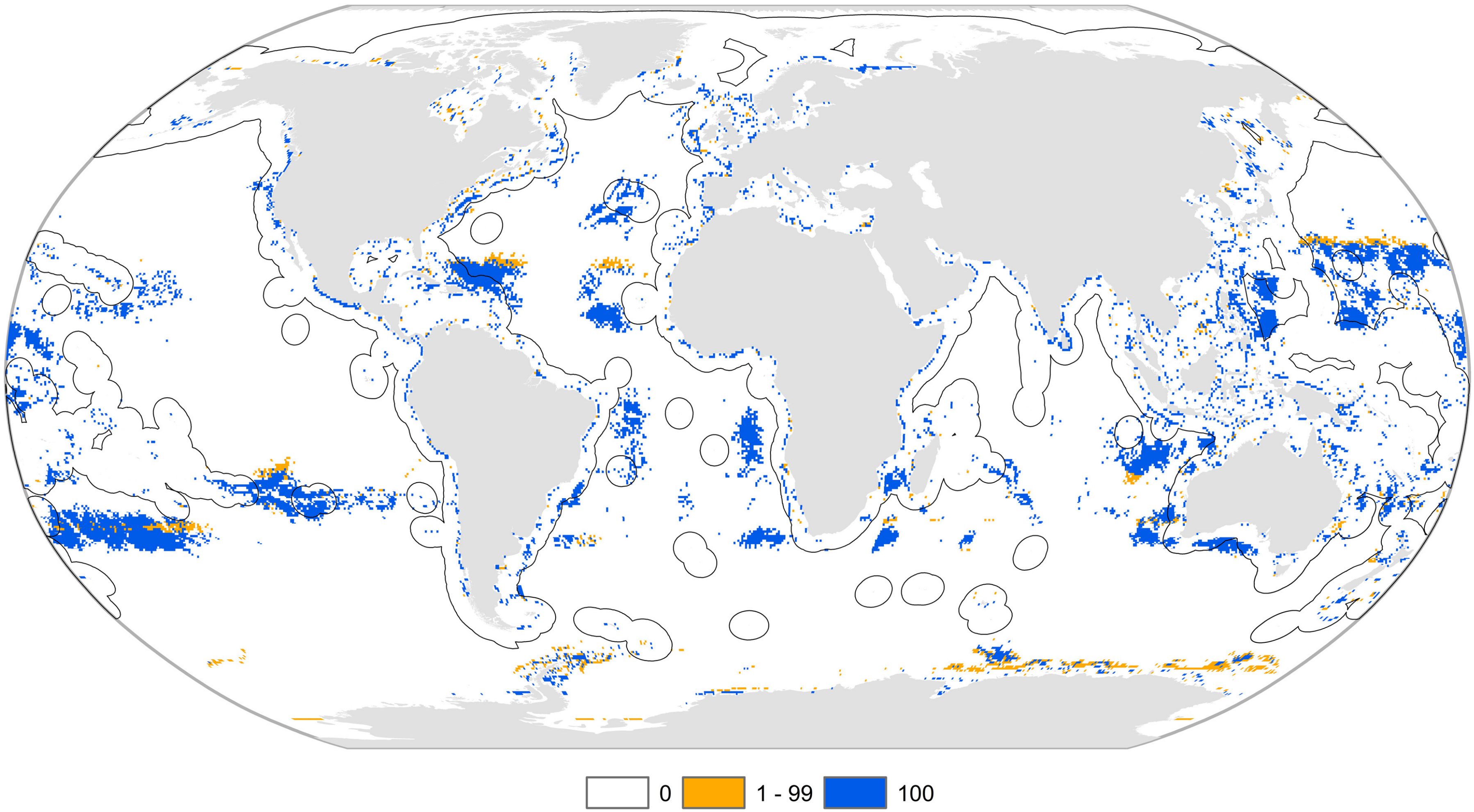

Figure 6.13 Frequency of selection of cells over 100 iterations of ResNet for scenario Global_10Fi. Approximate boundaries of the 200nm Exclusive Economic Zone (EEZ) boundaries are shown in black.

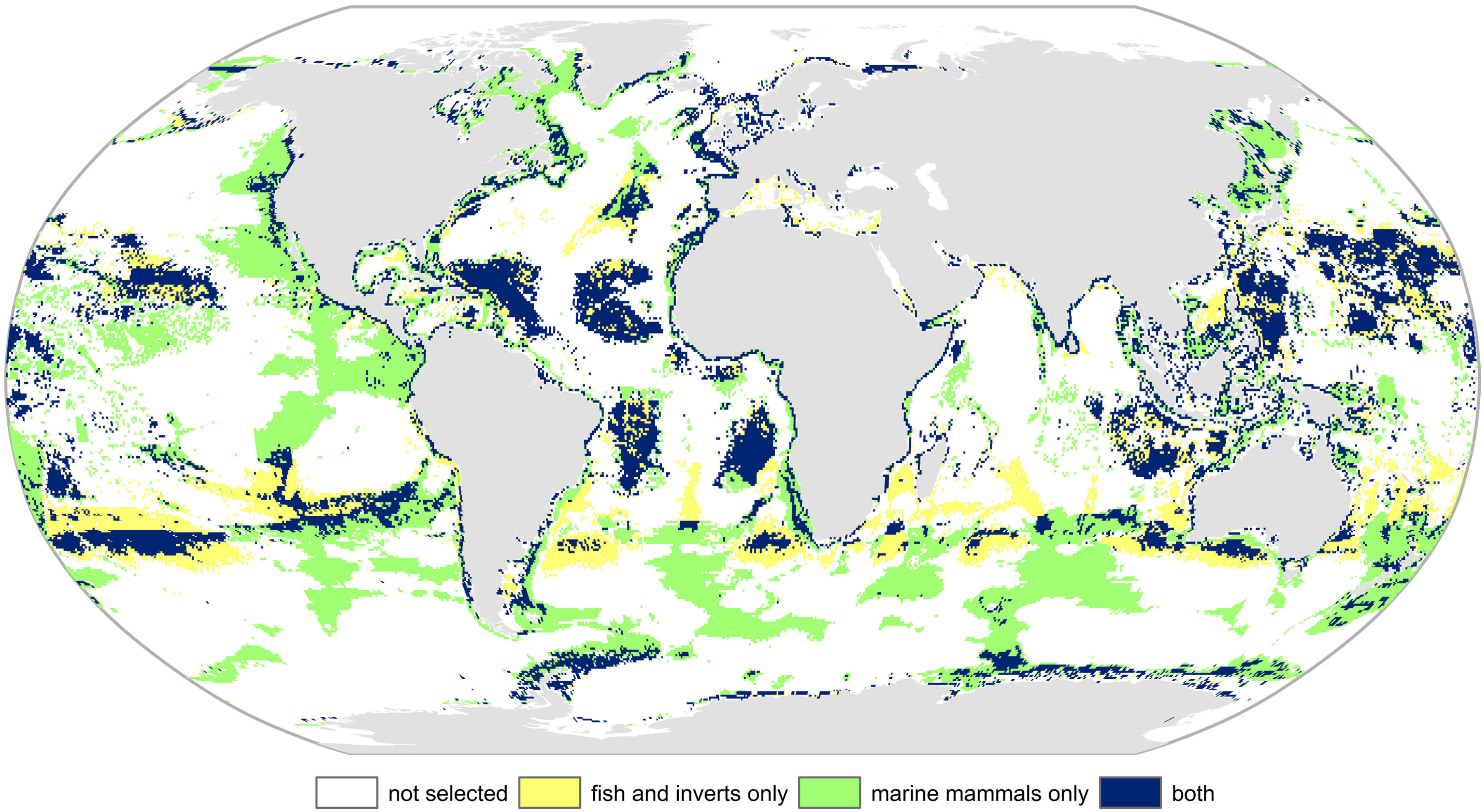

Figure 6.14 Overlap between cells selected 100% of the time in scenarios WPC30_All and WPC30_Fi. Approximate boundaries of the 200nm Exclusive Economic Zone (EEZ) boundaries are shown in black.

References

Agardy, T. (2005) Global marine conservation policy versus site-level implementation: the mismatch of scale and its implications. *Marine Ecology Progress Series*, **300**, 242-248.

Agardy, T., Bridgewater, P., Crosby, M.P., Day, J., Dayton, P.K., Kenchington, R., Laffoley, D., McConney, P., Murray, P.A., Parks, J.E. & Peau, L. (2003) Dangerous targets? Unresolved issues and ideological clashes around marine protected areas. *Aquatic Conservation: Marine and Freshwater Ecosystems*, **13**(4), 353-367.

Araújo, M.B. & Williams, P.H. (2000) Selecting areas for species persistence using occurrence data. *Biological Conservation*, **96**(3), 331-345.

Arthur, J.L., Hachey, M., Sahr, K., Huso, M. & Kiester, A.R. (1997) Finding all optimal solutions to the reserve site selection problem: formulation and computational analysis. *Environmental and Ecological Statistics*, **4**, 153-165.

Ball, I.R. & Possingham, H.P. (2000) *Marxan (v1.8.2). Marine reserve deisgn using spatially explicit annealing. A manual prepared for the Great Barrier Reef Marine Park Authority*. University of Queensland, Brisbane, Australia.

Balmford, A., Mace, G.M. & Ginsberg, J.R. (1998) The challenges to conservation in a changing world: putting processes on the map. In *Conservation in a changing world* (eds G.M. Mace, A. Balmford & J.R. Ginsberg), pp. 1-28. Cambridge University Press, Cambridge, UK.

Beger, M., Jones, G.P. & Munday, P.L. (2003) Conservation of coral reef biodiversity: a comparison of reserve selection procedures for corals and fishes. *Biological Conservation*, **111**(1), 53-62.

Boersma, P.D. & Parrish, J.K. (1999) Limiting abuse: marine protected areas, a limited solution. *Ecological Economics*, **31**(2), 287-304.

Brandt, A., Gooday, A.J., Brandao, S.N., Brix, S., Brokeland, W., Cedhagen, T., Choudhury, M., Cornelius, N., Danis, B., De Mesel, I., Diaz, R.J., Gillan, D.C., Ebbe, B., Howe, J.A., Janussen, D., Kaiser, S., Linse, K., Malyutina, M., Pawlowski, J., Raupach, M. & Vanreusel, A. (2007) First insights into the biodiversity and biogeography of the Southern Ocean deep sea. **447**(7142), 307-311.

Cabeza, M. & Moilanen, A. (2001) Design of reserve networks and the persistence of biodiversity. *Trends in Ecology and Evolution*, **16**(5), 242-248.

Cavalieri, D., Parkinson, C., Gloersen, P. & Zwally, H.J. (1996, updated 2006) Sea ice concentrations from Nimbus-7 SMMR and DMSP SSM/I passive microwave data. National Snow and Ice Data Center, Boulder, Colorado, USA.

CBD (2005) *Draft global outcome-oriented targets for the programme of work on marine and coastal biological diversity*. Convention on Biological Diversity, Bangkok, Thailand.

CBD (2006) *Decisions adopted by the Conference of the Parties to the Convention on Biological Diversity at its eighth meeting (Decision VIII/15, Annex IV)*. Convention on Biological Diversity, Curitiba, Brazil.

Close, C., Cheung, W.L., Hodgson, S., Lam, V., Watson, R. & Pauly, D. (2006) Distribution ranges of commercial fishes and invertebrates. In *Fishes in databases and ecosystems, Fisheries Centre Research Reports 14(4)* (eds M.L.D. Palomares, K.I. Stergiou & D. Pauly), pp. 27-37. University of British Columbia, Vancouver, Canada.

Costanza, R. (1999) The ecological, economic, and social importance of the oceans. *Ecological Economics*, **31**(2), 199-213.

Csuti, B., Polasky, S., Williams, P.H., Pressey, R.L., Camm, J.D., Kershaw, M., Kiester, A.R., Downs, B., Hamilton, R., Huso, M. & Sahr, K. (1997) A comparison of reserve selection algorithms using data on terrestrial vertebrates in Oregon. *Biological Conservation*, **80**, 83-97.

Faith, D., Carter, G., Cassis, G., Ferrier, S. & Wilkie, L. (2003) Complementarity, biodiversity viability analysis, and policy-based algorithms for conservation. *Environmental Science and Policy*, **6**(3), 311-328.

Favreau, J., Drew, C., Hess, G., Rubino, M., Koch, F. & Eschelbach, K. (2006) Recommendations for assessing the effectiveness of surrogate species approaches. *Biodiversity and Conservation*, **15**(12), 3949-3969.

Fernandes, L., Day, J., Lewis, A., Slegers, S., Kerrigan, B., Breen, D., Cameron, D., Jago, B., Hall, J., Lowe, D., Innes, J., Tanzer, J., Chadwick, V., Thompson, L., Gorman, K., Simmons, M., Barnett, B., Sampson, K., De'ath, G., Mapstone, B., Marsh, H., Possingham, H., Ball, I., Ward, T., Dobbs, K., Aumend, J., Slater, D. & Stapleton, K. (2005) Establishing representative no-take areas in the Great Barrier Reef: Large-scale implementation of theory on marine protected areas. *Conservation Biology*, **19**(6), 1733-1744.

Flather, C.H., Wilson, K.R., Dean, D.J. & McComb, W.C. (1997) Identifying gaps in conservation networks: of indicators and uncertainty in geographic-based analyses. *Ecological Applications*, **7**(2), 531-542.

Foster, E., Flood, T., Graham, A. & Exel, M. (2005) Improved oceans governance to conserve high seas biodiversity. *Parks*, **15**(3), 19-23.

Fuller, T., Munguía, M., Mayfield, M., Sánchez-Corderdo, V. & Sarkar, S. (2006) Incorporating connectivity into conservation planning: A multi-criteria case study from central Mexico. *Biological Conservation*, **133**(2), 131-142.

Garson, J., Aggarwal, A. & Sarkar, S. (2002) *ResNet Manual Ver 1.2 30 October 2002*. University of Texas at Austin, Austin, Texas, USA.

Gaston, K.J. (1991) How large is a species' geographic range? *Oikos*, **61**(3), 434-438.

Gjerde, K.M. & Kelleher, G. (2005) High seas marine protected areas on the horizon: legal framework and recent progress. *Parks*, **15**(3), 11-18.

Gladstone, W. (2002) The potential value of indicator groups in the selection of marine reserves. *Biological Conservation*, **104**, 211-220.

Grant, S.M. (2005) Challenges of marine protected area development in Antarctica. *Parks*, **15**(3), 40-47.

Hoyt, E. (2005) *Marine protected areas for whales, dolphins and porpoises*. Earthscan, London, UK.

Hyrenbach, K.D., Forney, K.A. & Dayton, P.K. (2000) Marine protected areas and ocean basin management. *Aquatic Conservation: Marine and Freshwater Ecosystems*, **10**(6), 437-458.

IUCN (2003) *WPC Recommendation 5.22: Building a global system of marine and coastal protected area networks*. IUCN, Durban, South Africa.

Kaschner, K., Watson, R., Trites, A.W. & Pauly, D. (2006) Mapping world-wide distributions of marine mammal species using a relative environmental suitability (RES) model. *Marine Ecology Progress Series*, **316**, 285-310.

Kelley, C., Garson, J., Aggarwal, A., and S. Sarkar (2002) Place Prioritization for Biodiversity Reserve Network Design: A Comparison of the SITES and ResNet Software Packages for Coverage and Efficiency. *Diversity and Distributions*, **8**, 297–306.

Kingsland, S.E. (2002) Creating a science of nature reserve design: perspectives from history. *Environmental Modeling and Assessment*, **7**(2), 61-69.

Kirkpatrick, J.B. (1983) An iterative model for establishing priorities for the selection of nature reserves: an example from Tasmania. *Biological Conservation*, **25**(2), 127-134.

Kirkpatrick, S., Gelatt Jr, C.D. & Vecchi, M.P. (1983) Optimisation by simulated annealing. *Science*, **220**, 671-680.

Leslie, H., Ruckelshaus, M., Ball, I.R., Andelman, S. & Possingham, H.P. (2003) Using siting algorithms in the design of marine reserve networks. *Ecological Applications*, **13**(1), S185-S198.

Malczewski, J. (1999) *GIS and Multicriteria Decision Analysis*. John Wiley & Sons, Inc., New York, USA.

Margules, C.R. & Pressey, R.L. (2000) Systematic conservation planning. *Nature*, **405**, 243-253.

Mintzberg, H. (1992) *The rise and fall of strategic planning: reconceiving roles for planning, plans, planners*. The Free Press, New York, USA.

Moffett, A., Dyer, J.S. & Sarkar, S. (2006) Integrating biodiversity representation with multiple criteria in North-Central Namibia using non-dominated alternatives and a modified analytic hierarchy process. *Biological Conservation*, **129**(2), 181-191.

Moffett, A. & Sarkar, S. (2006) Incorporating multiple criteria into the design of conservation area networks: a minireview with recommendations. *Diversity & Distributions*, **12**(2), 125-137.

Nicholls, A.O. (1998) Integrating population abundance, dynamics and distribution into broad-scale priority setting. In *Conservation in a changing world* (eds G.M. Mace, A. Balmford & J.R. Ginsberg), pp. 251-272. Cambridge University Press, Cambridge, UK.

Norse, E.A. & Crowder, L.B. (eds.) (2005) *Marine Conservation Biology*. Island Press, Washington DC, USA.

Pauly, D. & Christensen, V. (1995) Primary production required to sustain global fisheries. *Nature*, **374**, 255-257.

Possingham, H. (2001) *The business of biodiversity: applying decision theory principles to nature conservation*. Australian Conservation Foundation, Fitzroy, Victoria, Australia.

Possingham, H., Ball, I. & Andelman, S. (2000) Mathematical methods for identifying representative reserve networks. In *Quantitative methods for conservation biology* (ed S. Ferson, Burgman, M.), pp. 291-305. Springer-Verlag, New York, USA.

Possingham, H.P., Wilson, K.A., Andelman, S. & Vynne, C.H. (2006) Protected Areas. In *Principles of Conservation Biology* (eds M.J. Groom, G.K. Meffe & C.R. Carroll), pp. 509-551. Sinauer Associates Inc, Sunderland, MA, USA.

Preikshot, D. & Pauly, D. (2005) Global fisheries and marine conservation: is coexistence possible? In *Marine conservation biology: the science of maintaining the sea's biodiversity* (eds E.A. Norse & L.B. Crowder), pp. 185-197. Island Press, Washington DC, USA.

Pressey, R.L., Cowling, R.M. & Rouget, M. (2003) Formulating conservation targets for biodiversity pattern and process in the Cape Floristic Region, South Africa. *Biological Conservation*, **112**(1-2), 99-127.

Pressey, R.L., Humphries, C.J., Margules, C.R., Vane-Wright, R.I. & Williams, P.H. (1993) Beyond opportunism: key principles for systematic reserve selection. *Trends in Ecology and Evolution*, **8**(4), 124-128.

Pressey, R.L., Johnson, I.R. & Wilson, P.D. (1994) Shades of irreplaceability: Towards a measure of the contribution of sites to a reservation goal. *Biodiversity and Conservation*, **3**, 242-262.

Pressey, R.L., Possingham, H.P. & Day, J.R. (1997) Effectiveness of alternative heuristic algorithms for identifying indicative minimum requirements for conservation reserves. *Biological Conservation*, **80**(2), 207-219.

Pressey, R.L., Possingham, H.P. & Margules, C.R. (1996) Optimality in reserve selection algorithms: When does it matter and how much? *Biological Conservation*, **76**(3), 259-267.

Probert, P.K. (1999) Seamounts, sanctuaries and sustainability: moving towards deep-sea conservation. *Aquatic Conservation: Marine and Freshwater Ecosystems*, **9**, 601-605.

Reyers, B., van Jaarsveld, A.S. & Kruger, M. (2000) Complementarity as a biodiversity indicator strategy. *Proceedings: Biological Sciences*, **267**(1442), 505-513.

Rice, J. & Belland, R.J. (1982) A simulation study of moss floras using Jaccard's Coefficient of Similarity. *Journal of Biogeography*, **9**(5), 411-419.

Roberts, C.M., McClean, C.J., Veron, J.E.N., Hawkins, J.P., Allen, G.R., McAllister, D.E., Mittermeier, C.G., Schueler, F.W., Spalding, M., Wells, F., Vynne, C. & Werner, T.B. (2002) Marine biodiversity hotspots and conservation priorities for tropical reefs. *Science*, **295**(5558), 1280-1284.

Rodrigues, A.S.L., Cerdeira, J.O. & Gaston, K.J. (2000) Flexibility, efficiency, and accountability: adapting reserve selection algorithsm to more complex conservation problems. *Ecography*, **23**, 565-574.

Rodrigues, A.S.L. & Gaston, K.J. (2002) Optimisation in reserve selection procedures - why not? *Biological Conservation*, **107**(1), 123-129.

Rogers, A.D. (2004) *The biology, ecology and vulnerability of seamount communities*. IUCN, Gland, Switzerland.

Rondinini, C., Wilson, K.A., Boitani, L., Grantham, H. & Possingham, H.P. (2006) Tradeoffs of different types of species occurrence data for use in systematic conservation planning. *Ecology Letters*, **9**(10), 1136-1145.

Sarkar, S., Aggarwal, A., Garson, J., Margules, C.R. & Zeidler, J. (2002) Place prioritization for biodiversity content. *Journal of Biosciences*, **27**(4,S2), 339-346.

Sarkar, S., Pappas, C., Garson, J., Aggarwal, A. & Cameron, S. (2004) Place prioritisation for biodiversity conservation using probabalistic surrogate distribution data. *Diversity & Distributions*, **10**, 125-133.

Sarkar, S., Pressey, R.L., Faith, D.P., Margules, C.R., Fuller, T., Stoms, D.M., Moffett, A., Wilson, K.A., Williams, K.J., Williams, P.H. & Andelman, S. (2006) Biodiversity conservation planning tools: Present status and challenges for the future. *Annual Review of Environment and Resources*, **31**, 123-159.

Sea Around Us (2007) *A global database on marine fisheries and ecosystems*, Fisheries Centre, University British Columbia, Vancouver (British Columbia, Canada). http://www.seaaroundus.org [accessed 5th September 2007].

Shannon, L.V., Crawford, R.J.M., Brundrit, G.B. & Underhill, L.G. (1988) Responses of fish populations in the Benguela ecosystem to environmental change. *Journal du Conseil pour l'Exploration de la Mer*, **45**, 5-12.

Solomon, M., Van Jaarsveld, A.S., Biggs, H.C. & Knight, M.H. (2003) Conservation targets for viable species assemblages? *Biodiversity and Conservation*, **12**(12), 2435-2441.

Soulé, M.E. & Sanjayan, M.A. (1998) Conservation targets: do they help? *Science*, **279**(5359), 2060-2061.

Stevens, T. & Connolly, R.M. (2004) Testing the utility of abiotic surrogates for marine habitat mapping at scales relevant to management. *Biological Conservation*, **119**(3), 351-362.

Stewart, R.R., Ball, I.R. & Possingham, H.P. (2007) The effect of incremental reserve design and changing reservation goals on the long-term efficiency of reserve systems. *Conservation Biology*, **21**(2), 346-354.

Stewart, R.R., Noyce, T., Possingham, H. P. (2003) Opportunity costs of ad hoc marine reserve design decisions: an example from South Australia. *Marine Ecology Progress Series*, **253**, 25-38.

Sumaila, U.R., Zeller, D., Watson, R., Alder, J. & Pauly, D. (2007) Potential costs and benefits of marine reserves in the high seas. *Marine Ecology Progress Series*, **345**, 305-310.

Underhill, L.G. (1994) Optimal and suboptimal reserve selection algorithms. *Biological Conservation*, **70**, 85-87.

United Nations (2002) *Draft plan of implementation of the World Summit on Sustainable Development*. United Nations, Johannesburg, South Africa.

van Jaarsveld, A.S., Freitag, S., Chown, S.L., Muller, C., Koch, S., Hull, H., Bellamy, C., Kruger, M., Endrody-Younga, S., Mansell, M.W. & Scholtz, C.H. (1998) Biodiversity assessment and conservation strategies. *Science*, **279**, 2106-2108.

Vane-Wright, R.I., Humphries, C.J. & Williams, P.H. (1991) What to protect? - Systematics and the agony of choice. *Biological Conservation*, **55**, 235-254.

Wack, P. (1985) Scenarios: shooting the rapids. *Harvard Business Review*, **63**(6), 139-150.

Ward, T., Vanderklift, M.A., Nicholls, A.O. & Kenchington, R. (1999) Selecting marine reserves using habitats and species assemblages as surrogates for biological diversity. *Ecological Applications*, **9**(2), 691-698.

Warman, L.D., Sinclair, A.R.E., Scudder, G.G.E., Klinkenberg, B. & Pressey, R.L. (2004) Sensitivity of systematic reserve selection to decisions about scale, biological data, and targets: case study from southern British Columbia. *Conservation Biology*, **18**(3), 655-666.

Watson, R., Kitchingman, A., Gelchu, A. & Pauly, D. (2004) Mapping global fisheries: sharpening our focus. *Fish and Fisheries*, **5**, 168-177.

WestGrid (2007) *Western Canada Research Grid*, http://www.westgrid.ca/ [accessed 11th September 2007].

Wildavsky, A. (1973) If planning is everything, maybe it's nothing. *Policy Sciences*, **4**(2), 127-153.

Williams, P. (2001) Complementarity. In *Encyclyopedia of Biodiversity* (ed S.A. Levin), pp. 813-829.

Wilson, K.A., Pressey, R.L., Newton, A., Possingham, H. & Weston, C. (2005a) Measuring and incorporating vulnerability into conservation planning. *Environmental Management*, **35**(5), 527-543.

Wilson, K.A., Westphal, M.I., Possingham, H.P. & Elith, J. (2005b) Sensitivity of conservation planning to different approaches to using predicted species distribution data. *Biological Conservation*, **122**(1), 99-112.

Wood, L.J. (submitted) Global marine protection targets: how S.M.A.R.T. are they?

Wood, L.J. & Dragicevic, S. (2007) GIS-based multicriteria evaluation and fuzzy sets to identify priority sites for marine protection. *Biodiversity and Conservation*, **16**(9), 2539-2558.

Wood, L.J., Fish, L., Laughren, J. & Pauly, D. (in press) Assessing progress towards global marine protection targets: shortfalls in information and action. *Oryx*.

Worm, B., Lotze, H.K. & Myers, R.A. (2003) Predator diversity hotspots in the blue ocean. *Proceedings of the National Academy of Sciences*, **100**, 9884-9888.

Zacharias, M.A. & Howes, D.E. (1998) An analysis of marine protected areas in British Columbia, Canada, using a marine ecological classification. *Natural Areas Journal*, **18**(1), 4-13.

7. Conclusions

In this thesis I have provided various contributions to the knowledge and understanding of the current level of protection in the world's oceans, as well as some insights into the practical and theoretical challenges associated with global monitoring and data collection. I have also provided the first spatial visualisations of the extent of marine protection required to meet formally adopted targets. A concluding summary of the findings presented in this thesis, their limitations, as well as some suggestions for future work, is provided below.

For the first half of this thesis I investigated the current extent and distribution of the world's marine protected areas (MPAs). I present the results of the first explicitly marine-focused, global assessment of MPAs, and the first quantitative assessment of the attainability of global marine protection targets, based on an extensive baseline data collection exercise (Chapter 2). Although some consider systematic collection of baseline data to be only a 'filling in' activity (see Longhurst, 1998), the extent and breadth of changes made to the original MPA dataset that I describe in Chapter 2 indicate that this is a misperception of the relationship between new insights and old data. Indeed, prior to this research, global maps of MPAs had never been produced. The use of a Geographic Information System (GIS) to visualise MPAs on maps has enabled knowledge about the extent of protection to be communicated in a more intuitive way. This can help to engage the public and political representatives in debates about if, how and where to increase the level of protection (Pauly *et al.*, 2003). Similarly, the process of comparing targets to their expected achievement dates, and, in the case of the Convention on Biological Diversity (CBD) targets, the display on maps of countries' level of protection compared to the targets, may help to mobilise support for the policy shifts and increased resources needed to increase the current level of marine protection.

The processes involved in developing a global MPA database posed profound practical and theoretical challenges associated with global monitoring of MPAs, notably because the MPA concept, within the concept of 'protection', has deep roots in pragmatics and ethics. In particular, the challenge is to develop from individual perceptions a global definition of the minimum level of regulation of human activity necessary for an area to be considered 'protected' (Chapter 3). These issues are still being debated and, consequently, the operational definition of MPA used for the work presented in Chapter 2 should not be assumed to reflect the full spectrum of spatial regulation of human activity in the oceans. There remain some data gaps, and updating of the database is an ongoing process, so the assessment presented in Chapter 2 will need to be updated as more data come available. Future work on the database could also include expanding it to enable more effective monitoring of additional aspects of the global marine protection targets, including management effectiveness and habitat representation.

To some extent, the results that I present in Chapter 2 validate concerns expressed over the relevance and utility of broad conservation targets (Soulé and Sanjayan, 1998; Solomon *et al.*, 2003; Svancara *et al.*, 2005; Wiersma and Nudds, 2006). However, in Chapter 4, I suggest that the primary role of large-scale targets may be psychological – to motivate behavioural change – rather than ecological. Nevertheless, large-scale targets have historically been relatively unsuccessful in this role. I explored the possibility that this may be attributable to the way in which large-scale targets are formulated, using the SMART framework as a means of assessment. Although a lack of operational definitions of target requirements may be a proximate cause of low measurability and perceived attainability of targets, I suggest that there are more fundamental problems. Firstly, both ecosystems and political systems are multi-tiered and hierarchical in nature, implying a need for a pluralistic, systems approach to marine resource management and conservation (Berkes, 2007), and thereby a hierarchical system of targets to be developed for application at multiple scales in time and space. However, to date, there is a general lack of intermediate-scale targets that translate large-scale targets into locally relevant ones. Indeed, most criticisms of large-scale targets have been based on direct applications of large-scale targets to much more local scale contexts. Secondly, the political will required to achieve the targets does not necessarily reside with those who have the decision-making power to invest the resources necessary for the targets to be met. However, I propose that there seem to be some feedback mechanisms in place, whereby political will and perceived attainability are influenced both by each other as well as by target formulation. Furthermore, the whole process is likely to be undermined by challenging targets that have no clear implementation strategy, as is the case here. I conclude that the process of formulating targets is itself evolving, that the SMART concept may provide useful guidance in this regard, and that it may be premature to abandon the use of large-scale targets altogether.

In the second half of this thesis (Chapters 5 and 6) I therefore investigated the utility of decision support methods to identify global priorities for protection in the world's oceans, which may provide some strategic context to inform and/or facilitate the decision-making process to create global networks of MPAs. The multicriteria nature of the decision problem lent itself to an application of multicriteria evaluation (MCE), which was used to assess the suitability of different cells within the Pacific Canadian EEZ based on two (largely conflicting) objectives: biodiversity conservation and fisheries profit-maximisation (Chapter 5). This was one of the first times that MCE had been used in spatial decision-making in the marine environment, and the first time that MCE had been used to identify priority areas for protection in such a large area. The approach proved useful for investigating how different stakeholder preferences might influence the selection of different cells. However, the cell-level comparison required an aggregate measure of each criterion (in this case, species richness and ex-vessel value of the catch). The species-richness basis of the selection process did not guarantee representation of all species under consideration within the selected areas, which is a primary goal of systematic conservation planning (Margules and Pressey, 2000; Sarkar *et al.*, 2006), as well as the global targets.

In Chapter 6 I used an heuristic place prioritisation algorithm encoded in the ResNet software package (Garson *et al.*, 2007) to investigate various minimum set problems for species representation targets that reflect the global marine protection targets, using distribution range maps for 923 commercially caught (and hence, generally abundant) fish and invertebrate species, and 115 marine mammal species in a global grid of ~180,000 cells measuring 0.5° latitude/longitude. This was the first time that the species distribution ranges of marine species have been used in a globally synthetic way, and is by far the largest application of a place prioritisation algorithm to date. It is also the first time that the concept of a global network of MPAs has been visualised on a map (although this is not to say that the solutions presented are true 'networks'; see Chapter 2). As such the results I present in Chapter 6 are both unique and novel, and bring a new level of context and tangibility to the magnitude of the change required for the targets to be met on time.

The solutions I present in Chapter 6 show locations in the world's oceans that, based on available data, represent global priority areas where regional-scale protected area network design exercises might be undertaken. As such they may be understood to represent the first step in a global planning process to implement global networks of MPAs. However, none of the solutions identified in Chapter 6 are claimed to be definitive MPA network configurations: they are hypothetical global priorities developed based on existing global targets and known spatial patterns of biodiversity. Furthermore, and partly because these solutions are the first of their kind, they are subject to various limitations that provide scope for further work. These are discussed below.

Although the species distributions used represent a broad taxonomic range, there are data gaps, especially in high latitudes. In addition, solutions varied in size and spatial location depending on the surrogate set that was used, which re-confirms that choice of surrogates can substantially influence the results generated by place prioritisation algorithms (van Jaarsveld *et al.*, 1998; Reyers *et al.*, 2000). These analyses should thus be repeated as new species distribution data come available, both for additional species as well as refinements to existing distributions. In addition, since only one type of place prioritisation algorithm was used here, it would be interesting to repeat this study using a range of different algorithms, including heuristics using different rules (e.g., complementarity only), as well as optimal and metaheuristic algorithms.

Various related issues relating to spatial and temporal scale could be investigated as a follow up to this study. For example, the extent to which changes in species distributions resulting from climate change (Soto, 2002; Perry *et al.*, 2005; Parmesan, 2006; Simmonds and Isaac, 2007) affect the location of priority areas for MPAs could be investigated by comparing solutions produced using species distributions modelled under different climate scenarios. Given the variation in solutions (and in some cases, lack of congruence between them) produced by algorithms applied at different scales and resolutions (Pressey *et al.*, 1993; Erasmus *et al.*, 1999; Hopkinson *et al.*, 2000; Warman *et al.*, 2004), it is also important that protected area network design exercises are carried out in smaller areas and with higher resolution, to complement the results presented here. The implementation of MPA networks is unlikely to occur instantaneously, instead developing incrementally through space and time (Meir *et al.*, 2004; Stewart *et al.*, 2007). The interactions between, and impacts of, different spatial and temporal scales of MPA implementation on the characteristics and representation levels achieved (at various scales) could be explored by, for example, comparing solutions generated with areas of local, regional or global importance locked into them.

Finally, the scenarios investigated in Chapter 6 were formulated in order to identify priority areas based solely on their biodiversity content, but multicriteria decision-making techniques could be used to investigate the socio-political feasibility of implementing them. However, at the global scale, the development of meaningful weightings of different criteria is problematic, because the constituency from which the weighting should be derived is effectively the world population, which cannot realistically be censused. A more feasible approach might be to use formal scenario analysis (Peterson *et al.*, 2003; Swart *et al.*, 2004), where hypothetical scenarios of alternative futures (which reflect different policy preferences) are devised, and theoretical weightings developed based on those the policy preferences.

References

Berkes, F. (2007) Community-based conservation in a globalized world. *Proceedings of the National Academy of Sciences*, **104**(39), 15188-15193.

Erasmus, B.F.N., Freitag, S., Gaston, K.J., Erasmus, B.H. & van Jaarsveld, A.S. (1999) Scale and conservation planning in the real world. *Proceedings of the Royal Society of London. Series B: Biological Sciences*, **266**(1417), 315-319.

Garson, J., Aggarwal, A. & Sarkar, S. (2007) *ResNet Manual Ver 1.2*.

Hopkinson, P., Evans, J. & Gregory, R.D. (2000) National-scale conservation assessments at an appropriate resolution. *Diversity and Distributions*, **6**(4), 195-204.

Longhurst, A. (1998) *Ecological geography of the sea*. Oval Press, London, UK.

Margules, C.R. & Pressey, R.L. (2000) Systematic conservation planning. *Nature*, **405**, 243-253.

Meir, E., Andelman, S. & Possingham, H. (2004) Does conservation planning matter in a dynamic and uncertain world? *Ecology Letters*, **7**, 615-622.

Parmesan, C. (2006) Ecological and evolutionary responses to recent climate change. *Annual Review of Ecology and Systematics*, **37**, 637-669.

Pauly, D., Watson, R. & Christensen, V. (2003) Ecological geography as a framework for a transition toward responsible fishing. In *Responsible fisheries in the marine ecosystem.* (eds M. Sinclair & G. Valdimarson), pp. 87-101. CAB International,

Perry, A.L., Low, P.J., Ellis, J.R. & Reynolds, J.D. (2005) Climate change and distribution shifts in marine fishes. *Science*, **308**(5730), 1912-1915.

Peterson, G.D., Cumming, G.S. & Carpenter, S.R. (2003) Scenario planning: a tool for conservation in an uncertain world. *Conservation BIology*, **17**(2), 358-365.

Pressey, R.L., Humphries, C.J., Margules, C.R., Vane-Wright, R.I. & Williams, P.H. (1993) Beyond opportunism: key principles for systematic reserve selection. *Trends in Ecology and Evolution*, **8**(4), 124-128.

Reyers, B., van Jaarsveld, A.S. & Kruger, M. (2000) Complementarity as a biodiversity indicator strategy. *Proceedings: Biological Sciences*, **267**(1442), 505-513.

Sarkar, S., Pressey, R.L., Faith, D.P., Margules, C.R., Fuller, T., Stoms, D.M., Moffett, A., Wilson, K.A., Williams, K.J., Williams, P.H. & Andelman, S. (2006) Biodiversity conservation planning tools: Present status and challenges for the future. *Annual Review of Environment and Resources*, **31**, 123-159.

Simmonds, M.P. & Isaac, S.J. (2007) The impacts of climate change on marine mammals: early signs of significant problems. *Oryx*, **41**(1), 19-26.

Solomon, M., Van Jaarsveld, A.S., Biggs, H.C. & Knight, M.H. (2003) Conservation targets for viable species assemblages? *Biodiversity and Conservation*, **12**(12), 2435-2441.

Soto, C. (2002) The potential impacts of global climate change on marine protected areas. *Reviews in Fish Biology and Fisheries*, **11**(3), 181-195.

Soulé, M.E. & Sanjayan, M.A. (1998) Conservation targets: do they help? *Science*, **279**(5359), 2060-2061.

Stewart, R.R., Ball, I.R. & Possingham, H.P. (2007) The effect of incremental reserve design and changing reservation goals on the long-term effliciency of reserve systems. *Conservation Biology*, **21**(2), 346-354.

Svancara, L.K., Brannon, R., Scott, J.M., Groves, C.R., Noss, R.F. & Pressey, B. (2005) Policy-driven vs. evidence-based conservation: a review of political targets and biological needs. *Bioscience*, **55**, 989-995.

Swart, R.J., Raskin, P. & Robinson, J. (2004) The problem of the future: sustainability science and scenario analysis. *Global Environmental Change*, **14**, 137-146.

van Jaarsveld, A.S., Freitag, S., Chown, S.L., Muller, C., Koch, S., Hull, H., Bellamy, C., Kruger, M., Endrody-Younga, S., Mansell, M.W. & Scholtz, C.H. (1998) Biodiversity assessment and conservation strategies. *Science*, **279**, 2106-2108.

Warman, L.D., Sinclair, A.R.E., Scudder, G.G.E., Klinkenberg, B. & Pressey, R.L. (2004) Sensitivity of systematic reserve selection to decisions about scale, biological data, and targets: case study from southern British Columbia. *Conservation Biology*, **18**(3), 655-666.

Wiersma, Y.F. & Nudds, T.D. (2006) Conservation targets for viable species assemblages in Canada: Are percentage targets appropriate? *Biodiversity and Conservation*, **15**(14), 4555-4567.

8. Appendices

Appendix 1

Table showing the proportion of the world's maritime territories that is protected. Overseas territories and the non-contiguous US states of Alaska and Hawaii are shown separately.

	< 1.6% of maritime territory protected	
*	Alaska	
	Albania	†
	Algeria	†
*	American Samoa	
*	Andaman and Nicobar Islands	†
	Angola	†
*	Anguilla	
	Antigua and Barbuda	†
	Argentina	†
*	Aruba	†
*	Ascension Island	†
*	Azores Islands	†
	Bahamas	†
	Bahrain	†
*	Baker and Howland Islands	
	Bangladesh	†
	Barbados	†
	Belgium	†
	Benin	†
*	Bermuda	†
*	Bouvet Island	†
	Brazil	†
*	British Indian Ocean Territory	†
	Brunei Darussalam	
	Bulgaria	†
	Cambodia	†
	Canada	†
*	Canary Islands	†
	Cape Verde	†
*	Cayman Islands	†
*	Channel Islands	†
	Chile	†
	China	†
*	Christmas Island	†
*	Clipperton Island	†
*	Cocos (Keeling) Islands	†
	Comoros	†
	Congo, Democratic Republic of the	†
	Congo, Republic of the	†
*	Cook Islands	†
	Costa Rica	†
	Cote d'Ivoire	†
	Croatia	†
*	Crozet Islands	†
	Cuba	†
	Cyprus	†

	< 1.6% of maritime territory protected	
*	Desventuradas Islands	†
	Djibouti	†
	Dominica	†
	East Timor	
*	Easter Island	†
	Ecuador	†
	El Salvador	†
	Equatorial Guinea	†
	Eritrea	†
*	Faeroe Is	†
*	Falkland Islands / Malvinas Islands	†
	Fiji	†
	Finland	†
	France	†
*	French Guiana	
*	French Mozambique Channel Islands	†
*	French Polynesia	†
	Gabon	†
	Gambia	†
*	Gaza Strip	†
	Georgia	†
	Ghana	†
*	Gibraltar	†
	Greece	†
	Grenada	†
*	Guadeloupe	†
*	Guam	
	Guatemala	†
	Guinea	†
	Guinea-Bissau	†
	Guyana	†
	Haiti	†
	Honduras	†
	Iceland	†
	India	†
	Indonesia	†
	Iran	†
	Iraq	
	Ireland	†
	Israel	†
	Italy	†
	Jamaica	†
*	Jan Mayen	†
	Japan (Pacific Ocean Coast & Sea of Japan)	†
*	Jarvis Island	
*	Johnston Island	

< 1.6% of maritime territory protected		
*	Juan Fernandez Islands	†
	Kenya	†
*	Kerguelen Island	†
*	Kermadec Islands	†
	Korea, Democratic People's Republic of	†
	Korea, Republic of	†
	Kuwait	†
	Latvia	†
	Lebanon	†
	Liberia	†
	Libyan Arab Jamahiriya	†
*	Lord Howe Island	†
	Madagascar	†
*	Madeira Island	†
	Malaysia	†
	Maldives	†
	Malta	†
*	Marcus Island	†
	Marshall Islands	†
*	Martinique	†
	Mauritius	†
*	Mayotte	†
	Mexico	†
	Micronesia, Federated States of	†
*	Midway Islands	
	Monaco	†
*	Montserrat	†
	Morocco	†
	Myanmar	†
	Namibia	†
	Nauru	†
	Netherlands	†
*	Netherlands Antilles	†
*	New Caledonia	†
	New Zealand	†
	Nicaragua	†
	Nigeria	†
*	Niue	†
*	Norfolk Island	†
*	Northern Mariana Islands	
	Norway	†
*	Ogasawara Islands	†
	Oman	†
	Pakistan	†
	Palau	†
*	Palmyra Atoll	
	Papua New Guinea	†
	Peru	†
	Philippines	†
*	Pitcairn Islands	†
	Poland	†
	Portugal	†

< 1.6% of maritime territory protected		
*	Prince Edward Island	†
*	Puerto Rico	
	Qatar	†
*	Reunion	†
*	Ryukyu and Daitoshoto Islands	†
*	Saint Helena	†
	Saint Kitts and Nevis	†
	Saint Lucia	†
*	Saint Paul and Amsterdam Islands	†
*	Saint Pierre and Miquelon	†
	Saint Vincent and the Grenadines	†
	Samoa	†
	Sao Tome and Principe	†
	Senegal	†
	Serbia Montenegro	†
	Seychelles	†
	Sierra Leone	†
	Singapore	†
	Slovenia	†
	Solomon Islands	†
	Somalia	
	South Africa	†
*	South Georgia and the South Sandwich Is	†
	Spain	†
	Sri Lanka	†
	Sudan	†
	Suriname	†
	Syrian Arab Republic	†
	Taiwan, Province of China	†
	Tanzania, United Republic of	†
	Thailand	†
	Togo	†
*	Tokelau	†
	Tonga	†
*	Trindade and Martin Vaz Island	†
	Trinidad and Tobago	†
*	Tristan da Cunha	†
*	Tromelin Island	†
	Tunisia	†
	Turkey	†
*	Turks and Caicos Islands	†
	Tuvalu	†
	United Arab Emirates	†
	United Kingdom	†
	Uruguay	†
	Vanuatu	†
	Viet Nam	†
*	Virgin Islands (British)	†
*	Virgin Islands (U.S.)	
*	Wake Island	
*	Wallis and Futuna Islands	†
*	Western Sahara	†

< 1.6% of maritime territory protected		
	Yemen	†

> 1.6% of maritime territory protected		
	Australia	†
	Belize	†
	Colombia	†
	Denmark	†
	Egypt	†
	Estonia	†
*	Greenland	†
	Kiribati	†
	Lithuania	†
	Mauritania	†
	Mozambique	†
*	Navassa Island	
	Panama	†
	Romania	†
	Russian Fed (All)	†
	Saudi Arabia (All)	†
	Sweden	†
	Ukraine	†
	United States of America	
	Venezuela	†

> 10% of maritime territory protected		
	Cameroon	†
	Dominican Republic	†
*	Galapagos Islands	†
	Germany	†
*	Hawaii	
*	Heard Island and McDonald Islands	†
	Jordan	†
*	Macquarie Island	†
*	Svalbard Island	†

* overseas territory or non-contiguous US state
† ratified / acceded to CBD