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ORIGINAL PAPER

GIS-based multicriteria evaluation and fuzzy sets to identify priority sites for marine protection

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Abstract There is an increasing momentum within the marine conservation community to develop representative networks of marine protected areas (MPAs) covering up to 30% of global marine habitats. However, marine conservation initiatives are perceived as uncoordinated at most levels of planning and decision-making. These initiatives also face the challenge of being in conflict with ongoing drives for sustained or increased resource extraction. Hence, there is an urgent need to develop large scale theoretical frameworks that explicitly address conflicting objectives that are embedded in the design and development of a global MPA network. Further, the frameworks must be able to guide the implementation of smaller scale initiatives within this global context. This research examines the applicability of an integrated spatial decision support framework based on geographic information systems (GIS), multicriteria evaluation (MCE) and fuzzy sets to objectively identify priority locations for future marine protection. MCE is a well-established optimisation method used extensively in land use resource allocation and decision support, and which has to date been underutilised in marine planning despite its potential to guide such efforts. The framework presented here was implemented in the Pacific Canadian Exclusive Economic Zone (EEZ) using two conflicting objectives - biodiversity conservation and fisheries profit-maximisation. The results indicate that the GIS-based MCE framework supports the objective identification of priority locations for future marine protection. This is achieved by integrating multi-source spatial data, facilitating the simultaneous combination of multiple objectives, explicitly including stakeholder preferences in the decisions, and providing visualisation capabilities to better understand how

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global MPA networks might be developed under conditions of uncertainty and complexity.

Keywords Fuzzy sets · GIS · Marine protected areas · Multicriteria evaluation · Multiple objectives · Protected area siting · Spatial analysis · Spatial decision support

Abbreviations

EEZ	Exclusive Economic Zone
GIS	Geographic Information System
MCE	Multicriteria Evaluation
MPA	Marine Protected Area

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 MPAs by 2012 (United Nations 2002). 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 2003a).

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 2002), 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 or algorithm 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).

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 large scale 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 paper begins with an overview of marine protected areas and resource conservation, then describes the integrated decision support framework, implements the framework in a Western Canada case study, and then discusses its significance for systematic planning of MPAs.

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 (L. J. Wood, personal communication). Furthermore, only a very small proportion of this area is considered to be effectively managed (Kelleher et al. 1995). 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. 2003). The development of fishing technologies over the last 50 years has made fishing 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; Lubchenco et al. 2003; Ward et al. 2001). 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 528 species from the marine biome listed on the IUCN Red List of Threatened Species (IUCN 2003b). 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. 2003), 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 50 to 375 fold increase in protection from the status quo, depending on whether any level of protection, or only no-take MPAs, are considered, respectively. 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 optimisation algorithms and modelling techniques have been developed over the past 25 years to assist in the design of terrestrial protected area 'sets' (Kirkpatrick et al. 1983; McDonnell 2002; Possingham et al. 2000; Pressey et al. 1993; Pressey et al. 1996; Pressey et al. 1997). 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 (Ball 2004; Possingham et al. 2000). 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, e.g. 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 realised (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; Malczewski 1999; Pitz and McKillip 1984). In this paper, 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 realised 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 to 1 (real number scale) or 0 to 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 visualised 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 (Ceballos-Silva and Lopez-Blanco 2003, Janssen and Rietveld 1990); 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, and even less so for MPAs. Extensive literature searches have 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 paper 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. Fourth, it differs from many MARXAN applications in that it addresses both biodiversity conservation and fisheries management objectives explicitly to produce optimised trade off results.

Method

Study site

Canada declared its EEZ under the 1996 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 458,000 km^2 adjacent to the province of British Columbia (Fig. 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

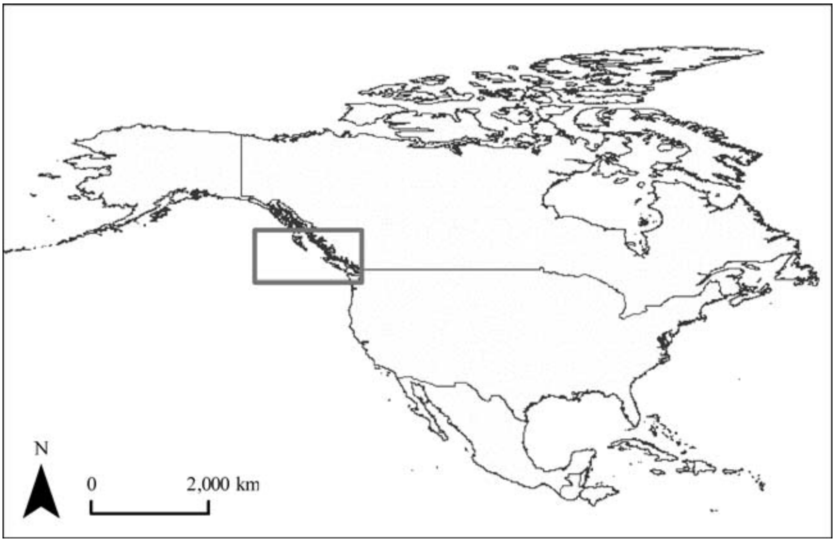

Fig. 1 Study area: the Pacific Canadian Exclusive Economic Zone (EEZ)

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. 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). This situation is analogous to other locations around the world and is representative of the lack of a framework observed for marine conservation globally.

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 258,000 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 prorating 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 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

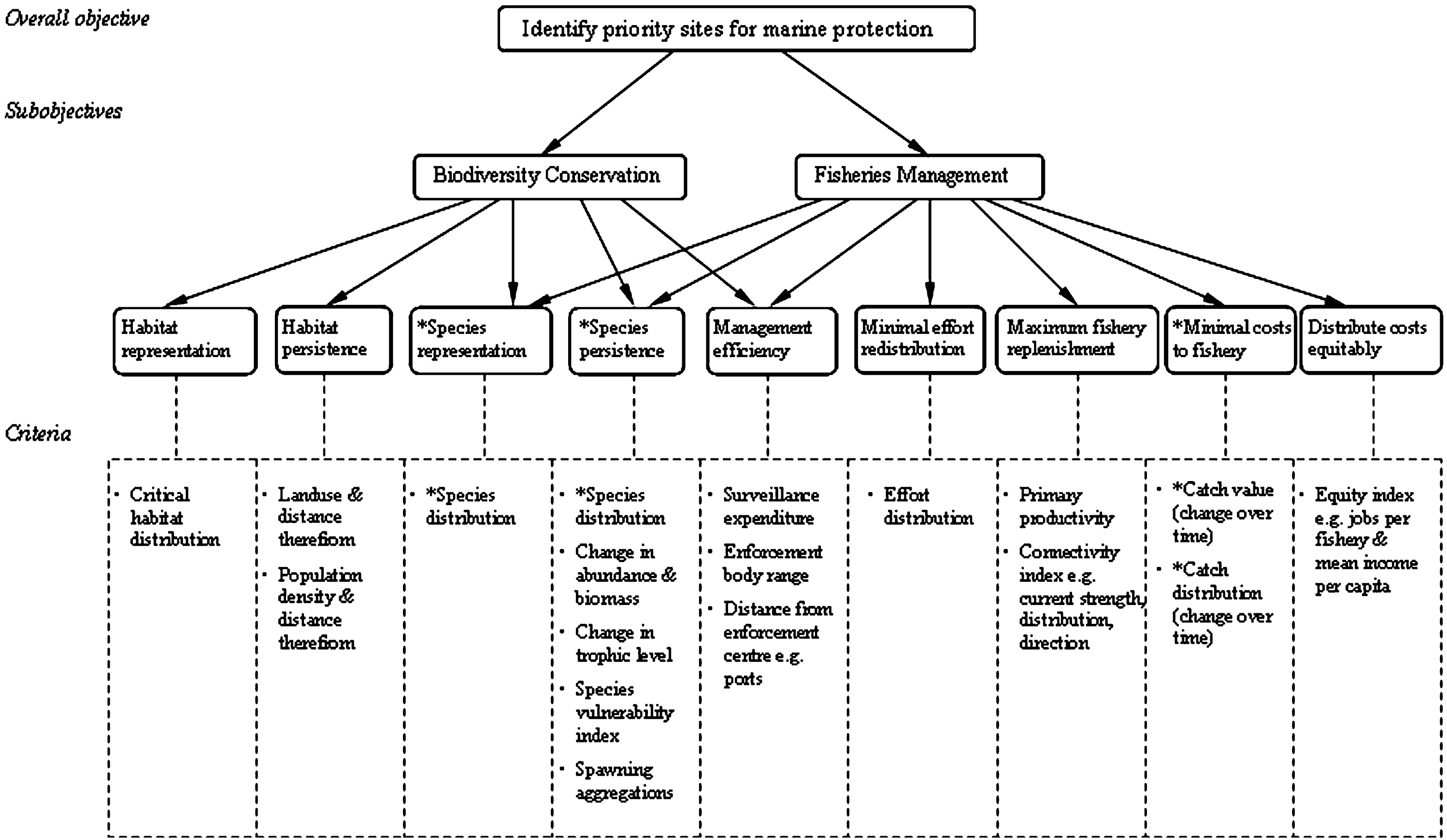

Fig. 2 Hypothetical formulation of the decision problem of identifying priority sites for marine protection using the hierarchy defined by Malczewski (1999). The criteria listed here do not represent an exhaustive list

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 (Fig. 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 methods, 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, 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

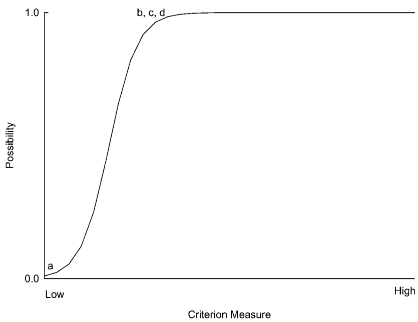

Fig. 3 Monotonically increasing, sigmoidal fuzzy membership function used to standardise MCE criteria layers. a, b c, and d represent inflection points where the membership function rises above zero, approaches one, falls below one and approaches zero again, respectively. A monotonically increasing function rises to one and never falls again. Hence b, c, and d all have the same criterion value of 1

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.

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 Fig. 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, i.e. 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 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 sigmoidal membership function illustrated in Fig. 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 2003b)

Table 1 Table 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

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 is 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, i.e. 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 Fig. 3. Actual weights used for all scenarios are shown in Table 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.

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 (Fig. 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 1 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

Table 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

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. As the ex vessel price values were absolute data, a sensitivity analysis of these 'weights' was not performed.

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 2.

Results and Discussion

Objective 1: Biodiversity Conservation

The results of the MCE for the three scenarios for the biodiversity conservation objective are shown in Fig. 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 (Fig. 4A), both weighting regimes of Scenario 1B (Fig. 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

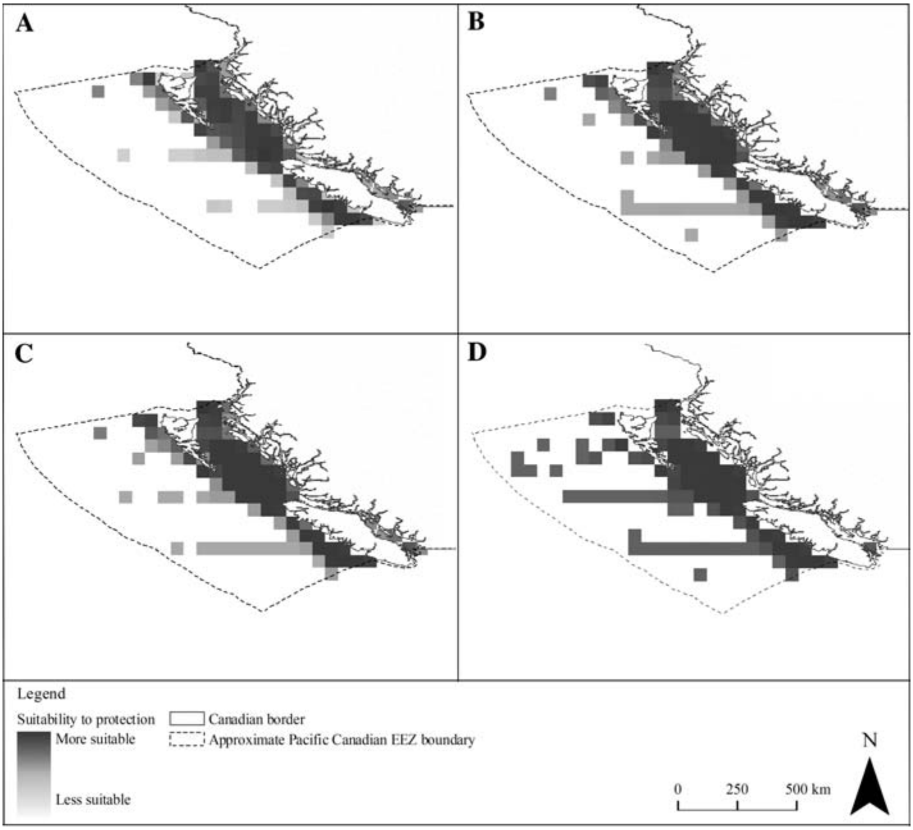

Fig. 4 Outputs of MCE for Objective 1—Biodiversity Conservation. Maps A–D show the 30% most suitable cells for protection from within the Pacific Canadian EEZ identified by the following scenarios: Equal weights for all species; Linear weighting regime for IUCN Red Listed species; Logistic weighting regime for IUCN Red Listed species; Logistic weighting regime for IUCN Red Listed species with *Critically Endangered* category removed

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 (Fig. 4B) are very similar to those of Scenario 1A (Fig. 4A). This is most likely because the weights are not widely spread in this scenario (Table 1)—the highest weight is only around 3 times that of the lowest weight. The logistic weighting regime for Scenario 1B.ii generated an inshore region of uniformly high suitability score (Fig. 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 (Fig. 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 Fig. 5A, and the 30% most valuable cells are shown in Fig. 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 (Figs. 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 (Fig. 4C) for the biodiversity conservation objective and the result for the fisheries management objective (Fig. 5A) as inputs. The results of using the three different weighting schemes outlined in Table 2 are illustrated in Fig. 6.

The results in Fig. 6 show absolute allocations of each cell to one objective or the other, without an indication of suitability. Figure 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 (Fig. 4C). Figure 6B shows the results of weighting fisheries management and biodiversity objectives equally. Quite a large area that is highly suitable to biodiversity

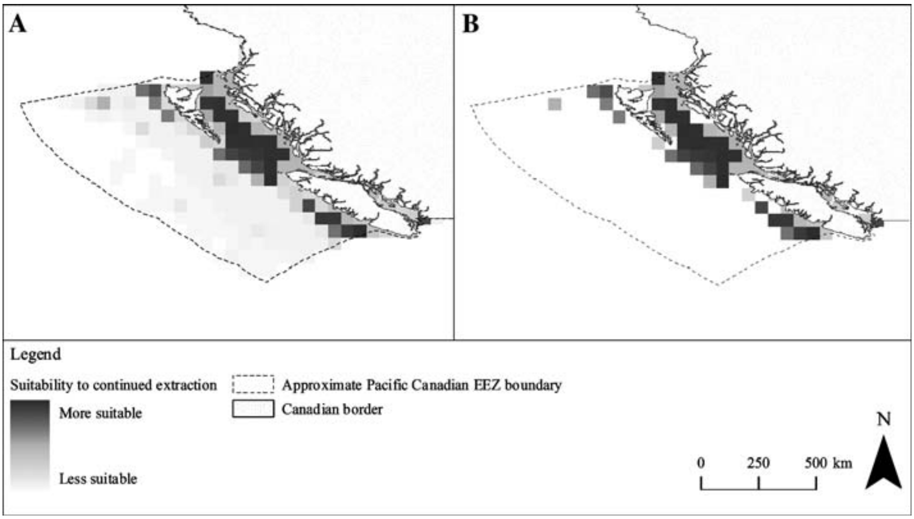

Fig. 5 Outputs of MCE for Objective 2—Fisheries Management. Map A shows the catch value distribution across the entire Pacific Canadian EEZ. Map B shows the 30% most suitable cells for continued resource extraction from within the Pacific Canadian EEZ, as identified by catch value distribution

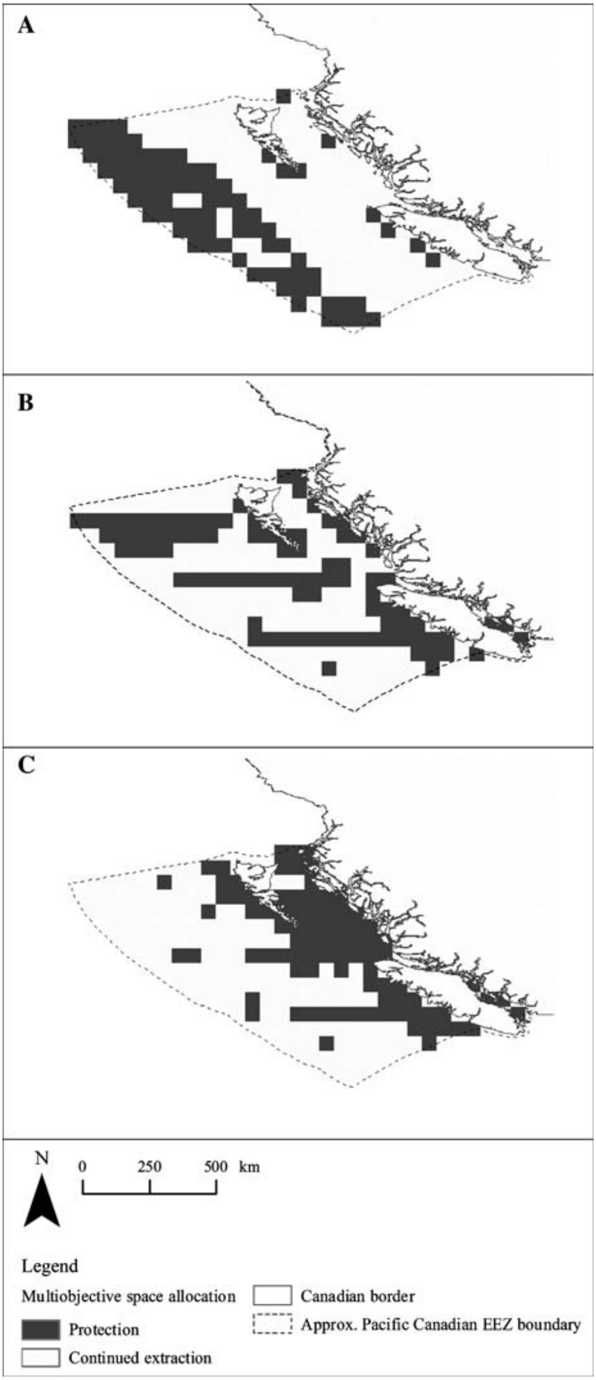

Fig. 6 Outputs of Multiobjective space allocation analysis, with biodiversity conservation and fisheries management as the two, conflicting objectives. Maps A–C show the 30% most suitable cells for protection from within the Pacific Canadian EEZ, as identified by the following objective weighting schemes, listed biodiversity conservation: fisheries management: A = 0.3:0.7; B = 0.5:0.5; C = 0.7:0.3

conservation and valuable to fisheries is selected for biodiversity conservation, and vice versa (compare Fig. 4C with Fig. 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 (Fig. 5A), even though they are of lower suitability for biodiversity conservation. Figure 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 Fig. 4C.

Conclusions

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 visualised. 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 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 paper 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 will be expanded to incorporate as many of the criteria and sub-objectives illustrated in the hierarchical structure of the problem (Fig. 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 Fig. 2 can reasonably be expected to improve the ability of the analysis to capture diversity over multiple scales, which would provide results that are more relevant to the large scale of study. Secondly, this framework will be developed to further extend the spatial scale of the model, specifically to the ocean basin scale. Thirdly, the results obtained from this framework should ideally be compared to those obtained from using other optimisation methods, such as MARXAN.

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