Developing best practice for using Marxan to locate Marine Protected Areas in European waters

Robert J. Smith, Paul D. Eastwood, Yoshitaka Ota, and Stuart I. Rogers

Smith, R. J., Eastwood, P. D., Ota, Y., and Rogers, S. I. 2009. Developing best practice for using Marxan to locate Marine Protected Areas in European waters. – ICES Journal of Marine Science, 66: 188 – 194.

Several recent studies have investigated the use of the conservation planning software Marxan to design Marine Protected Area (MPA) networks in UK waters. The systematic conservation planning approach embodied by Marxan has a number of advantages, but these studies have highlighted the need for guidance and advice on best practice. Here, we discuss two broad topics that we feel should inform future developments in the UK and elsewhere in the European Union. First, several technical issues need to be addressed to ensure the scientific defensibility of any conservation planning project. These include identifying which conservation features should be represented in an MPA system, developing a system for setting representation targets, and identifying which data should be included to minimize conflict with human uses of the sea. Second, it is necessary for researchers to engage at an early stage with those responsible for implementation and recognize that reserve selection should be part of a broader conservation planning process centred on a stakeholder-developed implementation strategy. A more-inclusive approach will make use of technical outputs, such as those generated by Marxan, as part of the process of policy development.

Keywords: biodiversity, fisheries, marine conservation planning, reserve selection.

Received 26 October 2007; accepted 5 June 2008; advance access publication 2 December 2008.

R. J. Smith: Durrell Institute of Conservation and Ecology, University of Kent, Canterbury, Kent CT2 7NR, UK. P. D. Eastwood and S. I. Rogers: Cefas, Pakefield Road, Lowestoft, Suffolk, NR33 0HT, UK. P. D. Eastwood current address: Secretariat of the Pacific Islands Applied Geoscience Commission, Private Mail Bag, GPO, Suva, Fiji Islands. Y. Ota: Department of Anthropology, University of Kent, Canterbury, Kent CT2 7NR, UK. Correspondence to R. J. Smith: tel: +44 1227 823455; fax: +44 1227 827289; e-mail: r.j.smith@kent.ac.uk.

Introduction

Marine biodiversity is under increasing pressure from a range of anthropogenic activities (Eastwood *et al*., 2007). Protected areas (PAs) have long been used to mitigate similar threats in the terrestrial realm, but they have been much less widely adopted in marine biomes, although spatial regulations are commonly used to manage fishery exploitation levels in many regions (Roberts *et al*., 2005). The relative paucity of PAs designed to meet marine nature conservation objectives reflects a number of factors, including tenure issues and a traditional focus on fish-stock conservation. However, the failure of existing fishery-management regimes, based largely on harvest control rules, and concerns over associated biodiversity loss, has led to calls for developing Marine Protected Areas (MPAs), based on a broader set of conservation objectives. In particular, there is an interest in conserving benthic biodiversity (Vincent *et al*., 2004), together with a focus on ecosystem-based management of human activities, including fishing (FAO, 2003), which would, by definition, include more extensive restrictions to protect and conserve a broader range of ecosystem components.

Such trends are reflected in Europe, where there is growing interest in designating MPAs in the waters of Member States. European Union (EU) countries are already obliged to create new MPAs as part of the Habitats and Birds Directives (EU, 1979, 1992), and these obligations may be further reinforced by new European legislation under the forthcoming Maritime Strategy (EC, 2005). These initiatives will also help to implement commitments to marine protection under the Protected Areas Programme of the World Conservation Union (IUCN), which promotes the establishment and effective management of a worldwide network of PAs (IUCN, 2000). In addition, they will help countries bordering the Atlantic Ocean, which are signatories to the Convention for the Protection of the Marine Environment of the Northeast Atlantic (the OSPAR Convention), to fulfil their commitments as part of the OSPAR Biological Diversity and Ecosystems Strategy (OSPAR, 2003).

Creating MPA networks is likely to be challenging, but a large body of research exists on the design of both terrestrial and marine PAs to meet nature conservation objectives (Margules and Pressey, 2000). This essay contains a review of some of the key literature on this topic and provides information on how future MPA networks could be developed for European waters. We begin by reviewing different approaches to PA system design, focusing particularly on the systematic conservation planning approach. Then, we describe the Marxan conservation planning software, which is commonly used to design both terrestrial and marine PA systems, and review three recent studies that assessed the use of this software to develop MPAs in UK waters. Finally, we suggest a number of issues that we feel should be addressed to ensure that future MPA design exercises in EU waters are scientifically defensible and likely to produce outputs that will facilitate implementation.

Approaches to PA system design

Most research on PA network design focuses on the terrestrial realm, where PA systems are generally more extensive. The literature reveals that politics plays a key role in the initial development of these systems, causing many PAs to be established in remote, scenically attractive areas that have little commercial value and are home to socially marginalized groups (Oldfield *et al*., 2004). Besides creating a number of long-term social problems, this approach leaves many species and habitats unrepresented (Pressey, 1994). A number of methods has therefore been developed to address this representation problem. The earliest of these focused on conserving large, wide-ranging "umbrella" species or species-rich hotspot habitats, but these approaches are generally poor at representing biodiversity (Margules and Pressey, 2000; Ozaki *et al*., 2006). Instead, conservation agencies responded to these shortcomings by developing specifically targeted schemes, such as Natura 2000 and key biodiversity area approaches, based on protecting sites that are important for priority species and habitats (Eken *et al*., 2004). However, these schemes are also problematic because they tend to be inefficient (Dimitrakopoulos *et al*., 2004; Jackson *et al*., 2004) and inflexible (Knight *et al*., 2007).

A more effective approach is systematic conservation planning (Margules and Pressey, 2000), which avoids being overly prescriptive but has two basic principles. First, it sets numerical targets for how much of each important biodiversity element should be conserved, making the planning process more transparent, more open to stakeholder involvement, and less likely to be affected by direct or unconscious political interference (Cowling *et al*., 2003). Second, it uses complementarity-based methods for selecting sites, so PAs are selected based on how much they would add to the existing PA system, rather than how much of each feature they contain (Margules and Pressey, 2000). Systematic conservation planning has been widely adopted by conservation practitioners and has consequently generated a significant literature on how PA networks can be designed to reduce their impact on other stakeholders and increase the likelihood of implementation (Knight *et al*., 2006b).

Despite advances in PA network design theory, all earlier approaches continue to be used today. Terrestrial PAs are still located based solely on minimizing political cost, or the presence of umbrella species, or without regard to existing levels of representation. Therefore, given the rising interest in designating PA networks for marine nature conservation, it is likely that the same shortcomings will influence their design (Stewart *et al*., 2003). Fortunately, in the short term, this may not be a problem because many MPA systems are so limited that any new MPA is likely to help conserve underrepresented biodiversity. Moreover, some existing schemes, such as the Special Areas of Conservation (SAC) system or stakeholder-driven MPA creation to protect fish stocks, have considerable political buy-in and should be encouraged (Knight and Cowling, 2007). However, it is widely recognized that these initiatives need to be situated within a broader systematic conservation planning process, which should develop MPA networks that effectively conserve biodiversity in a manner that is most acceptable to different and potentially wide-ranging stakeholder groups.

Designing PA networks with Marxan

A central part of the systematic conservation planning process is identifying priority areas for conserving the required features (Knight *et al*., 2006a). The priority areas are generally identified by dividing the planning region into a number of planning units, the size and dimensions of which are user-defined, calculating the amount of each conservation feature in each planning unit, and selecting portfolios of units that, when combined, meet these targets. Identifying these portfolios can be done manually, but it is generally much more efficient to use software to identify the preliminary set of priority areas. The most widely used conservation planning software is Marxan, which has been used to design marine and terrestrial PA networks in many countries (Ball and Possingham, 2000). In this section, we describe how Marxan operates and outline its data requirements.

Marxan selects planning units to meet the representation targets, but it also considers two other factors. First, each planning unit is assigned a cost, and Marxan acts to minimize the combined planning-unit cost of the portfolio, although it will still select expensive planning units if they are needed to meet the targets. This cost can be a measure of any aspect of the planning unit, such as its area, the risk of being affected by anthropogenic impacts, or the opportunity costs resulting from its protection (Wilson *et al*., 2005; Richardson *et al*., 2006a). Second, Marxan can be set to select adjacent planning units preferentially, rather than a series of unconnected units that might be less ecologically viable and more difficult to manage. Reducing fragmentation levels inevitably results in more planning units being added to the portfolio, so Marxan allows the user to adjust this trade-off by weighting the importance of minimizing the combined external edge of the selected patches by setting a boundary-length-modifier (BLM) value (Ball and Possingham, 2000).

Marxan then uses a mathematical approach called simulated annealing to identify PA portfolios. Marxan identifies a highly efficient portfolio each time it is run; therefore, 100 runs generate 100 different portfolios (Ball and Possingham, 2000). It then produces two output types. The first displays the "best" solution, i.e. the portfolio with the lowest cost. The second counts the number of times each planning unit was chosen across all portfolios. Units that appear in every portfolio are considered irreplaceable, because they are always needed to meet the targets, whereas other units could be swapped with similar units while still meeting the targets.

Marxan and MPA planning: examples from the UK

Marxan is best known for its use in designing MPA networks in tropical and subtropical reef ecosystems (Fernandes *et al*., 2005). However, recent national and European legislation has stimulated several exploratory studies that have used the software for conservation planning in UK temperate marine waters. Three of the most relevant to current policy interests in the UK are summarized below by their rationale, objectives, methods, and key assumptions.

Irish Sea Pilot, 2004

The first major study of the use of Marxan for MPA planning in UK waters was the Irish Sea Pilot (Lieberknecht *et al*., 2004; Vincent *et al*., 2004). The study, which formed part of the UK's Review of Marine Nature Conservation (RMNC), examined the effectiveness of existing mechanisms for protecting marine biodiversity and developed proposals for its improvement. A key component of the RMNC was an assessment of the available conservation legislation and the role of site-based protection for threatened, scarce, or nationally and internationally important species and habitats. Work was undertaken to evaluate whether or not Marxan could be used to identify a network of ecologically coherent MPAs in the Irish Sea, based on six predefined criteria: typicalness, naturalness, biodiversity, size, whether or not the area is critical for a mobile species, and whether or not the area supports a nationally important marine feature. The criteria were incorporated into Marxan through indirect means by modifying input data, target criteria, BLM values, and planning unit costs. The study used recently devised marine landscapes for the Irish Sea, benthic species and habitat types as conservation features, and an index of naturalness as the planning unit cost. This index was calculated from spatially explicit estimates of seabed trawling intensity, derived from air and sea fishery-protection and enforcement operations, with heavily trawled areas scoring low and lightly trawled areas scoring high.

Royal Commission on Environmental Pollution, 2004

Between 2002 and 2003, the Royal Commission on Environmental Pollution (RCEP) investigated the environmental effects of marine fisheries (RCEP, 2004). MPAs were given special attention owing to the implied positive contribution they could make to an ecosystem-based approach to fishery management. To demonstrate some of the methods that can be used to design MPA networks and to extend the work recently completed under the Irish Sea Pilot, the RCEP commissioned a special investigation into MPA planning using Marxan. Recognizing that Marxan is primarily designed for conservation planning and not fishery management, two scenarios were examined: one based on non-fisheries biodiversity criteria and the second based on the same biodiversity criteria but where fisheries-related criteria were also incorporated. Scenarios were constructed separately for the Irish and North Seas. Key data inputs for the biodiversity scenario included marine landscapes for the Irish Sea and seabed data for the North Sea. The biodiversity and fisheries combined scenario included additional data on spawning and nursery grounds for key commercial species (Coull *et al*., 1998), as well as landing values within International Council for the Exploration of the Sea (ICES) statistical rectangles.

Building the evidence base for the UK Marine Bill, 2006

Important changes to the way that UK marine waters will be managed in future were presented recently in a new Marine Bill (Defra, 2006). One measure proposes a network of MPAs to be designated to meet the UK's national and international conservation commitments. To increase understanding of the regulatory impact of this measure on existing UK marine legislation, a study was commissioned to identify possible MPA networks in UK waters (Richardson *et al*., 2006b). The study explored a number of scenarios using Marxan, which included setting representation targets of 20 and 60% for important species and habitats to represent the upper and lower limits for protection set by OSPAR, setting a representation target of 10% for all other marine habitats, and varying the importance of existing PAs by locking them into the network or making their cost zero. The study also ran scenarios that included broad, socio-economic design criteria aimed at avoiding conflict with other human activities. Information on conservation features consisted of point-source sample data for important species and habitats from databases maintained by the Joint Nature Conservation Committee (JNCC), UK marine landscapes recently developed by JNCC (Connor *et al*., 2006), and maps of fish spawning and nursery grounds developed by expert review (Coull *et al*., 1998). Despite varying the criteria and generating 12 different scenarios, the total area encompassed by the resultant networks only ranged between 14 and 20%, suggesting that similar outcomes could be achieved through a variety of different configurations.

Recommendations emanating from these UK studies

These three studies sought in part to test the suitability of using Marxan for designing MPA networks in UK waters and identified a number of critical issues that must be addressed in future applications of the method. First, the available point-source sample data for species and habitats were not evenly distributed, and this sampling bias led to the selection of priority areas at the locations of sampling points. Second, data describing the distribution of human activities must be made available at a fine spatial resolution, especially for mapping fishing patterns, the major pressure on the UK marine environment (Eastwood *et al*., 2007). Given the relative ubiquity of fishing compared with other human activities, data demonstrating fishing pressures and impacts will be central to minimizing conflict with competing sectors and so reducing the overall economic cost of any network configuration (Lynch, 2006; Richardson *et al*., 2006a). Third, there is a need to ensure adequate ecological connectivity between sites. Fourth, all the studies emphasized that their results were not meant as a guide to where MPAs should be located, and underscored the importance of comprehensive stakeholder consultation at all stages of future work.

Towards developing best practice

These studies have highlighted a number of areas of concern that should be addressed before Marxan is used to develop MPA networks in the UK or elsewhere in the EU. Below, we discuss how these issues could be overcome and provide broader recommendations to facilitate the successful adoption of the systematic conservation planning approach.

Increasing the likelihood of effective implementation

Systematic conservation planning is a long-term process that permits full stakeholder participation and aims to develop PA systems in a transparent manner, allowing dialogue between different user groups. However, most of the published work on this approach describes only the conservation assessment process, and the different techniques available for identifying priority areas in an efficient manner. This means that many projects focus too much on designing highly efficient MPA networks and, in doing so, fail to achieve stakeholder buy-in (Knight *et al*., 2006a). As a result, project outcomes are often not effectively implemented. Therefore, any conservation planning project, whether it uses Marxan or another approach to identify priority areas, should focus on the following implementation-based issues.

Identify the broad goals of the MPA network

The first step in any conservation planning exercise is to identify the broad goals that the MPA network aims to achieve. In the UK, these goals could include: (i) to conserve the important features that have been identified as part of existing legislation and the OSPAR Biodiversity Strategy; (ii) to represent all the biodiversity in UK waters, including the habitats, species, and ecological processes that are not currently listed for conservation action; (iii) to conserve historical maritime sites and other sites of scenic or cultural importance; and (iv) to conserve the habitats of commercially important fish species. The configuration and extent of the MPA network could be very different, depending on which of these broad goals are considered important, so it is vital that a consensus on these issues is reached at the beginning of the planning process and that these fit within the existing legislative framework (Jones, 2007).

Collaborate with stakeholders

The importance of stakeholder participation in MPA designation and management projects is widely supported by both fishery-management and conservation case studies (Agardy *et al*., 2003; Mascia *et al*., 2003). This is because stakeholders often have important quantitative and non-quantitative knowledge that can improve MPA designation and zoning systems (Aswani and Lauer, 2006). Marxan allows much of this information to be incorporated into the planning system, and it also provides a transparent process for exploring different scenarios and identifying a range of alternative solutions, allowing different stakeholders to identify favoured options. It should be noted, however, that effective stakeholder involvement is difficult and rarely achieved in conservation assessments (Helvey, 2004). Often, this is because the process is not adequately funded, leading to inadequate data collection (McClosky, 1999) and the exclusion of stakeholders who cannot afford to attend consultation meetings. Furthermore, information exchange between different stakeholders at these meetings can be problematic, especially when scientific information is presented in an overly technical manner or when no allowance is made to support effective interactions between different stakeholder groups.

Develop an implementation framework

Stakeholder involvement is a key to successful conservation planning, but it needs to be built into a broader management system (Dalton, 2005; Jones and Burgess, 2005). Therefore, it should be combined with an implementation framework that: (i) identifies potential conservation opportunities and constraints in the planning region; (ii) works with stakeholders to develop ways of representing these in the assessment; (iii) assigns the responsibility for implementing different tasks to the relevant agencies, assessing capacity levels, and training needs where necessary; and (iv) mainstreams the strategy into implementing organizations (Knight *et al*., 2006b). Stakeholder involvement also helps to identify unexpected opportunities, such as where one user group indirectly supports the creation of MPAs to protect an important resource, and helps broaden the constituency of those developing the MPA network, allowing potential sources of funding to be identified.

Disseminate the outputs of conservation assessments

It is vital that conservation assessment reports are written in a way that can be understood by all stakeholders, but care must also be taken to present the results at an opportune time. This is especially true when presenting maps, because these are powerful visual tools that suggest some level of agreement and permanence, even when they are published with caveats explaining their limitations. Therefore, draft maps can lead casual observers to doubt the value of the whole process (Smith *et al*., 2006) and can cause antagonism when seen by stakeholders who were not involved in developing them, especially if they appear to affect their livelihoods. An alternative is to use the approach taken by the report for the UK Marine Bill, which tabulated the results of the Marxan runs instead of relying on maps as the primary means of communicating the final outcomes (Richardson *et al*., 2006b). Tabulated results provide important and sufficient summary information, without taking the additional, somewhat controversial step of specifying where potential MPAs should be located.

Improving scientific defensibility

Systematic conservation planning should be seen as a continuous process that involves periodic conservation assessments to help inform the MPA designation process (Knight *et al*., 2006b). Because the quality of the data in the planning systems will obviously influence the value of the assessment process, we list below ways to improve the scientific defensibility of future assessments, based partly on recommendations made in the UK case studies described earlier. However, it must be emphasized that even assessments based on limited data can provide useful information (Smith *et al*., 2006) and that the conservation planning process should be seen as an adaptive process, with the location of priority areas changing in response to new opportunities and constraints, as well as the availability of better data (Lombard *et al*., 2007).

Select relevant conservation features

There are several different approaches to selecting relevant conservation features, but many practitioners aim to represent biodiversity by choosing a number of habitat types, species, and ecological processes (Cowling *et al*., 2004). Using habitat types is important because they can often be mapped relatively easily using methods that solve the problems associated with raw sample data (Smith *et al*., 2006) and because they can serve as a surrogate for benthic biodiversity. In addition, a range of species can be included as conservation features if their distribution is sufficiently well known. This is especially important for wide-ranging or range-limited species that might not be automatically conserved by protecting patches of the broad habitat types with which they are associated (Smith *et al*., 2008). When selecting conservation features for EU waters, several candidate species and habitats have already been identified through the Habitats and Birds Directives and the OSPAR Biodiversity Strategy. However, consensus is still needed on whether or not some species, such as cod (*Gadus morhua*), can be conserved through an MPA network, owing to concerns over effort redistribution from spatial closures, and also because they are wide-ranging and their distributions cannot be mapped accurately (Horwood *et al*., 2006). Agreement is also needed about which other biodiversity elements should be included in the analysis and whether or not, for example, each mapped habitat types should be represented in the MPA network.

Choose suitable planning units

Several studies have demonstrated that conservation assessments tend to identify different priority areas, depending on the spatial scale of the planning units used (Shriner *et al*., 2006). Therefore, it is important for assessments to use the scale at which management activities are likely to be implemented. In the UK, this could involve using ICES statistical rectangles, which measure half a degree of latitude by one degree of longitude, as planning units in offshore areas, plus smaller planning units for inshore areas. However, selecting large planning units is a much less efficient method for meeting targets, because it is likely that portions of these rectangles would not be required to achieve the targets (Pressey and Logan, 1998). Therefore, it might be more practical to subdivide the ICES rectangles into smaller planning units when selecting potential MPAs. In light of the data quality available, some areas may require a resolution different from others. For example, the analysis of fisheries-distribution data reveals that trawling data are most accurately represented at a grid-cell resolution of 3 km or less (Mills *et al*., 2007).

Map the conservation features

The value of any conservation assessment depends on the underlying conservation feature data. As the three UK case studies demonstrate, many of the available species data are strongly affected by sampling bias, so using these raw data is likely to produce inefficient results. It is probably better, therefore, to use modelling techniques to map the distribution of those species for which several methods are available (Guisan and Thuiller, 2005; Austin, 2007). Expert judgement can also be used to construct broad-scale range maps, if suitable occurrence data are not available (Knight *et al*., 2006b). Habitat types can probably be mapped with more certainty than species distributions, because some physical maps already exist and these can be combined with available biological data. Mapping ecological processes will have to rely more on expert opinion, because the distribution of some of these features can only be inferred from maps showing physical features. Experts should also be consulted when mapping species and habitat types, and these experts should come from a range of stakeholder groups, with fishers in particular having an input (Lundquist and Granek, 2005).

Set targets

Setting targets lies at the heart of systematic conservation planning, and the extent of any MPA system will depend greatly on these reference points. Expert opinion still plays a major part in this process, but some techniques are now being adopted to improve the scientific defensibility of individual targets. For example, estimates of minimum viable populations can be used to set species targets (Cabeza and Moilanen, 2001), whereas approaches based on species-area curves can help to inform targets for habitat types (Desmet and Cowling, 2004). It should be recognized, however, that a great deal of uncertainty will attend the target-setting process, especially when they are developed in part to conserve fish stocks (Helvey, 2004). Therefore, in some cases, it might be better to set low targets initially and increase them as more ecological and stock-response data are collected. Such a process would need strong support from relevant stakeholders, because increasing targets can lead to a loss of trust, but recent research suggests that this gradual process is relatively efficient because it tends to identify initial areas that would also be selected using higher targets (Stewart *et al*., 2007).

Produce planning unit cost and constraint data

In marine environments subject to a number of human pressures, adequate data describing these activities must be included in a conservation assessment. Data should also be used to identify which planning units are locked in or locked out of any portfolio. For example, maps of existing MPAs could be used to identify planning units that should be included automatically, as could maps of any other sites of conservation importance identified by stakeholders. Areas of great socio-economic activity (Eastwood *et al*., 2007) can be excluded from the MPA network, as long as this does not affect target achievement. Developing the planning unit cost data is more complicated, because it first involves deciding what these values should measure. Some studies have used fisheries catch data to minimize the financial impact of any MPA system (Richardson *et al*., 2006a), although this may not be suitable when trying to measure importance for multiple sectors. Other studies have used data on human activities to develop a "naturalness" index (e.g. Lieberknecht *et al*., 2004), preferentially selecting planning units that are less likely to be prized by extractive industries and are more likely to contain intact benthic ecosystems. Several studies have derived estimates of impact levels from certain activities on selected ecosystem components (reviewed in Eastwood *et al*., 2007); however, we are a long way from being able to quantify the full range of impacts from all activities on all important species and habitats. Therefore, in the short term, it will probably be sufficient to measure pressure levels using the best available spatial data on the location and intensity of all major human activities (Eastwood *et al*., 2007; Mills *et al*., 2007).

Assess connectivity and future fisheries effort redistribution

Marxan identifies priority areas based solely on meeting targets and reducing planning unit and boundary costs. Although it cannot, therefore, automatically ensure that portfolios will help to maintain ecological connectivity, there are several techniques for overcoming this limitation. First, the BLM value can be increased gradually until the resultant portfolios are extensive enough to ensure a high degree of connectivity. Levels of connectivity could be assessed, either by expert judgement or, preferably, using biophysical models capable of estimating trajectories of larvae from spawning to settlement areas for those species included as target conservation features (van der Molen *et al*., 2007). Second, linkages can be added between MPAs, ideally based on Marxan irreplaceability score maps (Smith *et al*., 2008). Third, the planning region can be subdivided and targets set for representing habitat in each division. Fourth, the many species of marine fish that use separate areas for different stages in their life history can be represented so that each stage is treated as a separate conservation feature.

Marxan is also unable to predict how the development of any MPA system is likely to affect the redistribution of human activities, particularly fishing effort. Instead, it can identify a number of different portfolios, so that each of these can be incorporated into a fishing-effort redistribution model to predict future patterns and determine the need and extent of any effort reduction. Marxan is also unable to identify where different types of management intervention take place: each planning unit is assumed to be either completely protected or completely unprotected. One way around this problem is to use the software to identify important areas, then develop a *post hoc* zoning plan (Smith *et al*., 2008). However, in future, it will also be possible to use the Marxan with Zones software (AEDA, 2008), which extends the functionality of Marxan by allowing targets to be set, based on how much of each feature is to be represented in different management zones.

Conclusions

In this paper, we have highlighted a number of issues that we think should be addressed before full-scale systematic conservation planning approaches are used in EU waters. These recommendations might initially seem overly ambitious, but a number of factors need to be considered when discussing their feasibility. First, many EU countries plan to designate SACs before augmenting and developing this system into a more representative MPA network under OSPAR, so many of our suggestions could be adopted as part of this incremental process. Second, a great amount of funding is already available for collecting similar datasets. Therefore, one early stage in any planning process should be the development of a coordinated, interdisciplinary research programme that would help fill these data gaps. Third, there is already sufficient biological data and expertise to undertake a preliminary conservation assessment in most parts of the EU, suggesting that the most pressing issues are related to implementation, not science. Taking these measures forward will require investment, commitment, and cross-sectoral planning and coordination. However, such efforts will be worthwhile if they facilitate the success of these important ventures.

Acknowledgements

This work was funded by the European Union under the Interreg 3A Programme, as part of the Channel Habitat Atlas for Marine Resource Management Project, and the Department of Environment, Food and Rural Affairs, marine environment research programme number AE1148.

References

AEDA. 2008. Making more of Marxan. Decision Point, 17: 8–9.

Agardy, T., Bridgewater, P., Crosby, M. P., Day, J., Dayton, P. K., Kenchington, R., Laffoley, D., *et al.* 2003. Dangerous targets? Unresolved issues and ideological clashes around marine protected areas. Aquatic Conservation: Marine and Freshwater Ecosystems, 13: 353–367.

Aswani, S., and Lauer, M. 2006. Incorporating fishermen's local knowledge and behavior into geographical information systems (GIS) for designing marine protected areas in Oceania. Human Organization, 65: 81–102.

Austin, M. 2007. Species distribution models and ecological theory: a critical assessment and some possible new approaches. Ecological Modelling, 200: 1–19.

Ball, I., and Possingham, H. 2000. Marxan (v1.8.2)—Marine Reserve Design using Spatially Explicit Annealing. University of Queensland, Brisbane, Australia. 69 pp.

Cabeza, M., and Moilanen, A. 2001. Design of reserve networks and the persistence of biodiversity. Trends in Ecology and Evolution, 16: 242–248.

Connor, D. W., Gilliland, P. M., Golding, N., Robinson, P., Todd, D., and Verling, E. 2006. UKSeaMap: the mapping of seabed and water column features of UK seas. JNCC, Peterborough, UK. 105 pp.

Coull, K. A., Johnstone, R., and Rogers, S. I. 1998. Fisheries sensitivity maps in British waters. UKOOA, Aberdeen, UK. 63 pp.

Cowling, R. M., Knight, A. T., Faith, D. P., Ferrier, S., Lombard, A. T., Driver, A., Rouget, M., *et al.* 2004. Nature conservation requires more than a passion for species. Conservation Biology, 18: 1674–1676.

Cowling, R. M., Pressey, R. L., Sims-Castley, R., le Roux, A., Baard, E., Burgers, C. J., and Palmer, G. 2003. The expert or the algorithm? Comparison of priority conservation areas in the Cape Floristic Region identified by park managers and reserve selection software. Biological Conservation, 112: 147–167.

Dalton, T. M. 2005. Beyond biogeography: a framework for involving the public in planning of US marine protected areas. Conservation Biology, 19: 1392–1401.

Defra. 2006. A consultation document for the Department for Environment, Food and Rural Affairs. Defra, London, UK. 315 pp.

Desmet, P., and Cowling, R. 2004. Using the species-area relationship to set baseline targets for conservation. Ecology and Society, 9: 11.

Dimitrakopoulos, P. G., Memtsas, D., and Troumbis, A. Y. 2004. Questioning the effectiveness of the Natura 2000 Special Areas of Conservation strategy: the case of Crete. Global Ecology and Biogeography, 13: 199–207.

Eastwood, P. D., Mills, C. M., Aldridge, J. N., Houghton, C. A., and Rogers, S. I. 2007. Human activities in UK offshore waters: an assessment of direct, physical pressure on the seabed. ICES Journal of Marine Science, 64: 453–463.

EC. 2005. Directive of the European Parliament and of the Council establishing a Framework for Community Action in the field of Marine Environmental Policy. Marine Strategy Framework Directive 9388/2/2007–C6-0261/2007–2005/0211(COD).

Eken, G., Bennun, L., Brooks, T. M., Darwall, W., Fishpool, L. D. C., Foster, M., Knox, D., *et al.* 2004. Key biodiversity areas as site conservation targets. BioScience, 54: 1110–1118.

EU. 1979. Council Directive 79/409/EEC of 2 April 1979 on the conservation of wild birds. Official Journal, L 103: 1.

EU. 1992. Council Directive 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora. Official Journal, L 206: 0007–0050.

FAO. 2003. Fisheries Management. 2. The Ecosystem Approach to Fisheries. FAO Technical Guidelines for Responsible Fisheries. 112 pp.

Fernandes, L., Day, J., Lewis, A., Slegers, S., Kerrigan, B., Breen, D., Cameron, D., *et al.* 2005. Establishing representative no-take areas in the Great Barrier Reef: large-scale implementation of theory on marine protected areas. Conservation Biology, 19: 1733–1744.

Guisan, A., and Thuiller, W. 2005. Predicting species distribution: offering more than simple habitat models. Ecology Letters, 8: 993–1009.

Helvey, M. 2004. Seeking consensus on designing marine protected areas: keeping the fishing community engaged. Coastal Management, 32: 173–190.

Horwood, J., O'Brien, C., and Darby, C. 2006. North Sea cod recovery? ICES Journal of Marine Science, 63: 961–968.

IUCN. 2000. Protected Areas, Benefits beyond Boundaries. International Union for Conservation of Nature and Natural Resources (IUCN), Gland, Switzerland. 19 pp.

Jackson, S. F., Kershaw, M., and Gaston, K. J. 2004. The performance of procedures for selecting conservation areas: waterbirds in the UK. Biological Conservation, 118: 261–270.

Jones, P. J. S. 2007. Arguments for conventional fisheries management and against no-take marine protected areas: only half of the story? Reviews in Fish Biology and Fisheries, 17: 31–43.

Jones, P. J. S., and Burgess, J. 2005. Building partnership capacity for the collaborative management of marine protected areas in the UK: a preliminary analysis. Journal of Environmental Management, 77: 227–243.

Knight, A. T., and Cowling, R. M. 2007. Embracing opportunism in the selection of priority conservation areas. Conservation Biology, 21: 1124–1126.

Knight, A. T., Cowling, R. M., and Campbell, B. M. 2006a. An operational model for implementing conservation action. Conservation Biology, 20: 408–419.

Knight, A. T., Driver, A., Cowling, R. M., Maze, K., Desmet, P. G., Lombard, A. T., Rouget, M., *et al.* 2006b. Designing systematic conservation assessments that promote effective implementation: best practice from South Africa. Conservation Biology, 20: 739–750.

Knight, A. T., Smith, R. J., Cowling, R. M., Desmet, P. G., Faith, D. P., Ferrier, S., Gelderblom, C. M., *et al.* 2007. Improving the key biodiversity areas approach for effective conservation planning. BioScience, 57: 256–261.

Lieberknecht, L. M., Carwardine, J., Connor, D. W., Vincent, M. A., Atkins, S. M., and Lumb, C. M. 2004. The Irish Sea Pilot—Report on the Identification of Nationally Important Marine Areas in the Irish Sea. JNCC, Peterborough, UK. 95 pp.

Lombard, A. T., Reyers, B., Schonegevel, L. Y., Coopers, J., Smith-Adao, L. B., Nel, D. C., Froneman, P. W., *et al.* 2007. Conserving pattern and process in the Southern Ocean: designing a marine protected area for the Prince Edward Islands. Antarctic Science, 19: 39–54.

Lundquist, C. J., and Granek, E. F. 2005. Strategies for successful marine conservation: integrating socioeconomic, political, and scientific factors. Conservation Biology, 19: 1771–1778.

Lynch, T. P. 2006. Incorporation of recreational fishing effort into design of marine protected areas. Conservation Biology, 20: 1466–1476.

Margules, C. R., and Pressey, R. L. 2000. Systematic conservation planning. Nature, 405: 243–253.

Mascia, M. B., Brosius, J. P., Dobson, T. A., Forbes, B. C., Horowitz, L., McKean, M. A., and Turner, N. J. 2003. Conservation and the social sciences. Conservation Biology, 17: 649–650.

McClosky, M. 1999. Local communities and the management of public forests. Ecology Law Quarterly, 25: 624–629.

Mills, C. M., Townsend, S. E., Jennings, S., Eastwood, P. D., and Houghton, C. A. 2007. Estimating high-resolution trawl fishing effort from satellite-based vessel monitoring system data. ICES Journal of Marine Science, 64: 248–255.

Oldfield, T. E. E., Smith, R. J., Harrop, S. R., and Leader-Williams, N. 2004. A gap analysis of terrestrial protected areas in England and its implications for conservation policy. Biological Conservation, 120: 303–309.

OSPAR. 2003. Joint HELCOM/OSPAR Work Programme on Marine Protected Areas. First Joint Ministerial Meeting of the Helsinki and OSPAR Commissions, Agenda item 6, Annex 7.

Ozaki, K., Isono, M., Kawahara, T., Iida, S., Kudo, T., and Fukuyama, K. 2006. A mechanistic approach to evaluation of umbrella species as conservation surrogates. Conservation Biology, 20: 1507–1515.

Pressey, R. L. 1994. *Ad hoc* reservations—forward or backward steps in developing representative reserve systems. Conservation Biology, 8: 662–668.

Pressey, R. L., and Logan, V. S. 1998. Size of selection units for future reserves and its influence on actual vs. targeted representation of features: a case study in western New South Wales. Biological Conservation, 85: 305–319.

RCEP. 2004. Turning the Tide: Addressing the Impact of Fisheries on the Marine Environment. HMSO, London, UK. 480 pp.

Richardson, E. A., Kaiser, M. J., Edwards-Jones, G., and Possingham, H. P. 2006a. Sensitivity of marine-reserve design to the spatial resolution of socioeconomic data. Conservation Biology, 20: 1191–1202.

Richardson, E. A., Kaiser, M. J., Hiddink, J. G., Galanidi, M., and Donald, E. J. 2006b. Developing scenarios for a network of marine protected areas: building the evidence base for the Marine Bill. Defra Research and Development Contract CRO 0348, London. 69 pp.

Roberts, C. M., Hawkins, J. P., and Gell, F. R. 2005. The role of marine reserves in achieving sustainable fisheries. Philosophical Transactions of the Royal Society of London, Series B. Biological Sciences, 360: 123–132.

Shriner, S. A., Wilson, K. R., and Flather, C. H. 2006. Reserve networks based on richness hotspots and representation vary with scale. Ecological Applications, 16: 1660–1673.

Smith, R. J., Easton, J., Nhancale, B. A., Armstrong, A. J., Culverwell, J., Dlamini, S., Goodman, P. S., *et al.* 2008. Designing a transfrontier conservation landscape for the Maputaland centre of endemism using biodiversity, economic and threat data. Biological Conservation, 141: 2127–2138.

Smith, R. J., Goodman, P. S., and Matthews, W. S. 2006. Systematic conservation planning: a review of perceived limitations and an illustration of the benefits, using a case study from Maputaland, South Africa. Oryx, 40: 400–410.

Stewart, R. R., Ball, I. R., and 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: 346–354.

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

van der Molen, J., Rogers, S. I., Ellis, J. R., Fox, C. J., and McCloghrie, P. 2007. Dispersal patterns of the eggs and larvae of spring-spawning fish in the Irish Sea, UK. Journal of Sea Research, 58: 313–330.

Vincent, M., Atkins, S., Lumb, S., Golding, C., Lieberknecht, L., and Webster, M. 2004. Marine nature conservation and sustainable development—the Irish Sea Pilot. JNCC, Peterborough, UK. 172 pp.

Wilson, K., Pressey, R. L., Newton, A., Burgman, M., Possingham, H., and Weston, C. 2005. Measuring and incorporating vulnerability into conservation planning. Environmental Management, 35: 527–543.

doi:10.1093/icesjms/fsn198