REPORT

Using Vessel Monitoring System Data to Improve Systematic Conservation Planning of a Multiple-Use Marine Protected Area, the Kosterhavet National Park (Sweden)

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Abstract When spatial fishing data is fed into systematic conservation planning processes the cost to a fishery could be ensured to be minimal in the zoning of marine protected areas. We used vessel monitoring system (VMS) data to map the distribution of prawn trawling and calculate fishing intensity for 1-ha grid cells, in the Kosterhavet National Park (Sweden). We then used the software Marxan to generate cost-efficient reserve networks that represented every biotope in the Park. We asked what were the potential gains and losses in terms of fishing effort and species conservation of different planning scenarios. Given a conservation target of 10 % representation of each biotope, the fishery need not lose more than 20 % of its fishing grounds to give way to cost-efficient conservation of benthic diversity. No additional reserved area was needed to achieve conservation targets while minimizing fishing costs. We discuss the benefits of using VMS data for conservation planning.

Keywords VMS data ·
Systematic conservation planning · Marxan ·
Marine protected area

INTRODUCTION

Spatial planning is one of the main tools in place to achieve conservation objectives in protected areas. However, to ensure societal support during planning processes it is vital that neither stakeholders nor conservation practitioners see their interests undermined. Spatial planning involves the practice of zoning (to spatially and temporally designate areas for specific purposes). Zoning serves to reduce two types of conflict: among users competing for the same space, and between users and the environment. The main

challenge facing conservation planning when livelihoods are at stake is that goals be sought in a way that minimizes forgone opportunities for use of natural resources (Margules and Pressey 2000).

To ensure that the capacity of the ecosystem to provide goods and services remains undiminished, in other words, to promote conservation goals, areas are frequently set aside and managed according to measures that enhance biodiversity. Systematic conservation planning (SCP) is an approach to spatial planning that involves finding cost-efficient sets of areas to protect biodiversity. Margules and Pressey (2000) describe SCP as a process comprising, at a minimum, the following stages: (1) compile data on the biodiversity (or surrogates thereof) of the planning region; (2) identify conservation goals for the planning region, preferably in the form of quantitative, operational targets for each species, habitat, etc.; and (3) select a set of conservation areas that collectively meet the representation targets assigned in step 2. The idea is then to minimize cost, where cost can be as simple as the total size of the reserved area. Importantly, in this process cost can also be expressed as loss of revenue resulting from diminished human use, which can be significant in heavily populated areas. These steps are embedded in a broader process accommodating actions for implementing conservation plans on the ground, revisiting and adapting zoning plans based on monitoring data, etc. The approach, which largely hinges on selecting representative sets of conservation areas (step 3), is considered highly effective because (a) it is efficient in using limited resources to achieve conservation goals, (b) it is transparent and flexible in the face of competing uses, and (c) it is accountable in allowing decisions to be critically reviewed (Margules and Pressey 2000).

SCP is increasingly being applied in marine areas around the world, e.g., in California (Klein et al. 2008), the

Florida Keys (Leslie et al. 2003), or the Great Barrier Reef (Great Barrier Reef National Park Authority 2004). Smith et al. (2009) reviewed a number of SCP projects in UK waters and highlighted issues that could help ensure the success of resulting marine protected areas (MPAs).

In marine areas, fishing is one of the most widespread uses, occupying the largest amount of space (Eastwood et al. 2007), and stakes are therefore high. To implement representative conservation networks that avoid negative effects on revenue from fishing it needs to be ensured that the least possible fishing area is reallocated to conservation. Preserving fishing grounds has the additional benefit of avoiding effort displacement, which may cause more harm than benefits to the ecosystem as a whole (Hiddink et al. 2006). But, just where does fishing take place?

The vessel monitoring system (VMS) provides real-time observations of a fishing vessels's position that are temporally continuous, thus allowing the monitoring of a fishing fleet. In addition, VMS data provides greater spatial resolution than logbook data, and the information is independent from fishermen declarations (Deng et al. 2005; Bertrand et al. 2007). Through the VMS, vessel positions obtained from a Global Positioning System (GPS) on board commercial vessels are transmitted via satellite in real time to the relevant fisheries monitoring center, typically on a 1- to 2-h interval. The transmitted signal is usually known as a 'ping'. While VMS data does not record whether the vessel is fishing or in transit this information can be confidently inferred from vessel speed (Deng et al. 2005; Gerritsen and Lordan 2011).

The objective of this paper is to demonstrate the benefits of using spatial fishing effort data in the SCP of a multiple-use MPA. In this case study we used VMS data to estimate the distribution of fishing activity in the Kosterhavet National Park area, where information is also available on the biological diversity of the seabed, which has been recently classified and mapped by Gonzalez-Mirelis and Lindegarth (2012). We estimate the gains and losses in terms of fishing and species conservation under three different planning scenarios where information on habitats, fishing effort and existing protected areas were incorporated to a varying degree.

CASE STUDY

The Kosterhavet National Park, off the West coast of Sweden, was designated in 2009 and it affords protection to the biota of a deep, submerged fjord and adjacent archipelago (Fig. 1). A management plan states the goals for the park, among which stands the long-term conservation of the marine ecosystems, habitats, and species occurring naturally in the region, while ensuring the sustainable use of local biological resources. Notably, the park is one where multiple

uses are allowed. Various kinds of commercial and recreational fisheries have a stake in the area, alongside tourism, and to a lesser extent, shipping. There is a well-established commercial fishery for northern prawn (*Pandalus borealis*) using demersal trawls, and Norway lobster (*Nephrops norvegicus*) using creels. In addition there is a seasonally intense fishery for lobster (*Homarus gammarus*) by both recreational and commercial fishers. The area is also important for recreation, with many hundreds of yachts, motor boats and kayaks visiting the area over the summer months. Tourism has increased 50 % over the last decade with around 80 000 visitors, mainly in July and August (Hambrey 2007). In light of this, a pressing need has emerged to lay out and ensure mechanisms of enforcement of a management system within which competing demands inside the multiple-use park are adequately accommodated while not compromising the conservation goals of the park.

Inside the national park the prawn trawl fishery has been well established since the early 1900s, yielding about 150 tonnes of prawn per year, and we will consider only this activity in the remainder of this paper. This fishery is regulated voluntarily as well as by law, under a special agreement between fisheries organizations and authorities at different levels, in effect since 2000. The regulations stipulate that trawling for prawn is only allowed in areas below 60 m depth. In addition, trawl gear is regulated with regard to weight of otter-boards. Sorting grids for the escape of fish bycatch (Isaksen et al. 1992) are mandatory. Education and sharing of knowledge are given special consideration in the agreement. Under its auspices, courses in marine ecology for fishermen, as well as courses in commercial fishing and fishery technology for researchers and officials, have been arranged over the years.

To safeguard outstanding examples of biodiversity the agreement established priority areas in the form of Seabed Protection Areas (SPAs) where trawling was banned, and subsequently also enforced by law in 2000. With the new regulations that came into force upon the designation of the National Park in 2009, anchoring and use of other equipment that could damage the seafloor became prohibited in SPAs. Figure 2 illustrates the zoning in place, showing that areas above 60 m depth and SPAs as closed for prawn trawling, and that 34 % of park area remains open to the prawn trawl fishery.

Given the goal for the park and the array of stakeholders involved, it is appropriate to ask the question: 'is seabed biodiversity adequately represented in no-trawling areas?' The answer of course depends on what is meant by 'adequate', which for this case study translates into at least two different questions: (1) 'how much of each component of biodiversity is being effectively protected (and is that enough)?', and (2) 'is conflict with users and fishing effort displacement minimized?' Regarding the former question,

Fig. 1 Location and bathymetry of the study area, the Kosterhavet National Park in Sweden

Fig. 2 Management zones and fishing activity in the Kosterhavet National Park. “SPAs” are Seabed Protection Areas. “Partial Protection” refers to areas above 60 m depth, where trawling for prawns is not allowed but other activities are. “Trawling area” is everything below 60 m

it was recently demonstrated that the zoning system of the National Park was heavily biased, with whole community types unrepresented in protected (SPAs), or partially protected (above 60 m) zones (Gonzalez-Mirelis and Lindegarth 2012). Moreover, species of conservation priority within Sweden and the OSPAR commission (OSPAR 2006) such as the seapens *Kophobelemnion stelliferum* and *Funiculina quadrangularis*, occurred mainly in trawled areas. The question whether conflict with users is minimal, i.e., whether fishing grounds have been mostly maintained, has never been assessed quantitatively.

MATERIALS AND METHODS

The process of SCP is underpinned by the delineation of ‘representative reserve networks’. Reserve networks, or systems, are collections of sites whose boundaries are drawn with the help of a map, which collectively contain instances of all regional biodiversity (i.e., the entire pool of species for the area considered), hence ‘representative’. For a reserve network to be effective upon implementation it has to be regulated according to its stated purpose of biodiversity conservation, which usually implies discontinuing human use inside sites included in the network. In the case of marine reserves, this involves banning fishing.

Reserve networks can be simulated to enable comparison between planning scenarios and thus provide an objective basis for management considerations and negotiation with stakeholders. For this to be possible representation targets need to be defined at the outset. These should be in theory the proportion of each species’ distribution area that should be contained in reserved sites. A range of biological data can be used as a proxy for biodiversity, from individual species to broad habitat types. These are generally referred to as conservation features. They offer a mechanism to protect biodiversity without needing specific or extensive knowledge of all the component species or their distributions.

In SCP, the building blocks of a reserve system are the individual cells of a regional grid, known as planning units. By overlaying planning units on distribution maps of various conservation features one can easily compute the amount of every feature contained in every planning unit. Once this is known, the problem of selecting the minimum number of planning units which achieves a pre-specified representation target for all conservation features becomes tractable. This has been called the “minimum-set problem” (Possingham et al. 2000) and is expressed as “minimize overall cost, subject to the constraint that all biodiversity targets (e.g., 20 % of each conservation feature) are met”. This problem has one optimal solution, which is very difficult to determine, and many near-optimal solutions. To

compare between them an objective function is used. The objective function is a mathematical function whose value reflects the desirability of a particular reserve system. In its simplest form, the objective function is a combination of the total cost of the reserve system and a penalty for any of the targets that are not met (Game and Grantham 2008). This objective function is designed so that the lower the value the more efficient the system.

Generally, it is desired that the reserve network is as small as possible, so that as much area as possible is available for different uses and thus continues to return revenue to the society. Additionally, larger reserve networks are more costly to manage (for example, in terms of monitoring and surveillance). Therefore, area itself is usually used as a cost in this process. With no other costs included, a network with a low objective function value is therefore considered area-efficient, because it uses as small an area as possible to achieve its targets. However, if additional spatial layers representing the distribution of different costs are added, an estimate of cost can be calculated for each planning unit and this way other costs can be minimized as well. For example, the distribution of fishing effort can represent revenue loss, and therefore the cost to a fishery. This approach will ensure that planning units of high value to the fishery are not selected to be included in a reserve system because they increase the value of the objective function.

The planning region of this case study was defined as the circalittoral area (below 30 m) of the Kosterhavet National Park. This encompasses an area of approximately 300 km². A regular grid comprising 30 893 planning units (1 ha cells) was used for the analyses. We used benthic biotopes as surrogates for biodiversity, that is, conservation features (Fig. 3). Biotopes are defined in terms of the physical habitat and the species that typically occur in them. The list of biotopes included the following: two hard-bottom classes, one deep (offshore circalittoral rock with axinellid sponges) and one generally occurring above 60 m (nearshore circalittoral rock with bryozoan turf); one biogenic reef habitat (*Lophelia pertusa* reefs), four mobile-sediment classes (offshore circalittoral mobile sand with sparse fauna, offshore circalittoral fine sediment with *Nephrops norvegicus*, offshore circalittoral mixed sediment and rock with *Spirontocaris lilljeborgii*–*Lithodes maja*, and nearshore circalittoral sediment with seapens), and two relatively shallow classes (upper nearshore circalittoral sediment with *Pennatula phosphorea* and upper nearshore circalittoral coarse sediment with scallop). They were mapped by means of a distribution model described elsewhere (Gonzalez-Mirelis and Lindegarth 2012). Any of the offshore circalittoral sediment areas (whereas sand or finer) are suitable habitat for *Pandalus borealis*, the target species of the prawn fishery.

Fig. 3 Biotopes of Kosterhavet National Park

Fig. 4 Benthic survey stations used in this study and shaded relief of the area

Prawn Trawl Fleet Fishing Intensity from VMS Data

The Swedish logbook includes gear set position which was used to estimate fleet composition and the landings originating from Kosterhavet National park. About 43 % of the prawn trawl fleet operating in Kosterhavet is equipped with VMSs, which is mandatory for vessels of 15 m length and larger. These vessels are equipped with a VMS transmitter providing information on identity, position, time, course, and speed approximately every hour to a satellite-based information system (Inmarsat-C). Periods of fishing activity were assigned to vessel and gear type by linking the VMS data to the logbook using the vessel identifier and time. The remainder 57 % of the prawn trawl fleet comprises smaller vessels and is not subscribed to VMS regulations. We used VMS data for years 2007, 2009, and 2010. Instantaneous speed was used to determine the occurrence of a fishing event. Inspection of vessel speed distribution suggested a trawling speed between 1.25 and 2.25 knots, and the data were filtered accordingly (Gerritsen and Lordan 2011). The density of VMS pings is directly related with time spent fishing. It is also related with total area trawled, although the relationship is more complex depending on fishermen individual behavior.

To estimate fishing intensity VMS pings were summarized by planning unit (1 ha grid cells). We used the maximum number of pings out of the 3 years as an estimate of fishing intensity for a planning unit. This results in an estimate of fishing intensity that is not directly translatable into a standard measure of fishing effort, but which effectively ranks planning units according to how heavily they are utilized by the fishing fleet. Fishing intensity was then used as cost in the simulation of reserve networks (see below).

Besides calculating fishing intensity at planning unit level, a GIS layer representing the overall distribution of fishing was also created. This layer was a simple dissolve of all planning units where *any* fishing occurred, i.e., any number of VMS pings was observed. We refer to this as fishing footprint area (see also Fig. 2).

Conservation Planning Settings

The software used to generate conservation networks was Marxan (Ball et al. 2009). Marxan is a decision support system designed to find near-optimal solutions to the minimum-set problem, using a heuristic algorithm called ‘simulated annealing’ with a spatial configuration constrained by a parameter that modifies overall boundary length.

We ran three scenarios to assess the trade-offs between different solutions:

- I. No costs were applied (i.e., the cost of any planning unit was only its area), and no planning units were forced into the reserve network;
- II. Costs were applied from fishing effort, and no planning units were forced into the reserve network;
- III. Costs were the same as in scenario II but planning units which were intersected by SPAs were locked into the reserve system.

Scenario I is optimized for conservation because no factor other than biotic composition affects the choice of planning units. Scenario II aims to balance conservation goals and fishing goals. Scenario III builds on currently enforced SPAs. Therefore Scenario III returns solutions that are, from this point of view, the easiest to implement.

Conservation targets were set to 10 % of each biotope for all three scenarios (Scott et al. 2001; Secretariat of the Convention on Biological Diversity 2004). We generated 100 solutions for each scenario with a boundary length modifier of 0.3, as this setting returned sites of similar size to existing SPAs, and were therefore considered realistic solutions. We only considered the ‘best’ solution of each run, where the score of the objective function defines ‘best’. The ‘best’ solution is the most efficient solution of the run, i.e., that which meets the conservation targets incurring the lowest possible cost to the fishery, and/or occupying the smallest possible area.

We address the question whether there is a potential for gain in terms of fishing revenue and species conservation, by using the distribution of VMS and biotopes in SCP scenarios, as detailed below.

Impact of Systematic Conservation Planning on the Fishery

We measured the (negative) impact of conservation planning on the fishery in two related ways. First we calculated the intersect area between each one of the three (best) solutions and the fishing footprint layer, and subtracted this from footprint area. This gives us an idea of the percent of fishing grounds that would be lost. In addition, we estimated total effort lost by summing over all planning units the number of VMS pings encompassed by each solution, and subtracted this from the maximum total number of pings.

Impact of Systematic Conservation Planning on Conservation

To measure the (positive) impact of conservation planning on species conservation we combined and collated species data from two benthic survey programs (see Fig. 4). Calculating the number of species covered by a given reserve network from field data is a commonly used method to

Fig. 5 Best Marxan solution for **a** scenario I (no cost), **b** scenario II (fishing costs, all planning units available), and **c** scenario III (fishing costs, SPAs locked in)

assess network effectiveness (Sarkar et al. 2005; Grantham et al. 2010), and is sometimes referred to as species recovery rate. We used both dredge/epibenthic sledge data, and underwater video footage data. Ninety-nine sites were sampled using either a rectangular dredge or an epibenthic sledge depending on the conditions of each site. Thirty-nine sites were sampled by means of a Remotely Operated Vehicle carrying a Sperre HD-SDI Camera. While 90 % of taxa from the bottom sampling data were identified at species level, organisms identified from the imagery were discarded if they could not be identified at least to Genus level. Therefore, both datasets are comprised mostly of taxa classified at least to Genus level.

We computed the number of species covered by each scenario (applying a 50-m buffer so as to make the best of the relatively scattered sampling effort) relative to the total number of species detected inside the National Park by either survey method. Following standard practice, we use this as an indicator of the number of species protected by each conservation network solution. The species data enabled us to estimate the improvement on biodiversity conservation quantitatively between scenarios.

RESULTS

For a total planning area of 308 km², 683 benthic taxa were identified, and 23.3 km² are being actively trawled for

Table 1 Scenario I: no costs; Scenario II: fishing cost, all planning units available; Scenario III: fishing cost, and SPAs locked in

Scenario	Total reserved (km ²)	Species protected (% of total species)	Available for fishing (% of currently fished area)	Area lost (%)	Effort lost (% of total measured effort)
I	52.7	39.8	70.8	−29.2	−32.1
II	51.6	32.1	80.2	−19.8	−19.25
III	67.8	46.6	81.1	−18.9	−17.8

prawns, out of a total 104.7 km² where trawling is allowed. Over the years of the study on average 25 prawn trawlers were operating in the fjord. The proportion of prawn trawler efforts equipped with VMS operating in the area was 49 % per year, responsible for 57 % of the yearly prawn landings.

The three best Marxan solutions are shown in Fig. 5. Network size was similar for scenarios I and II but it was larger for scenario III. The increase in area reserved was accompanied by an increase in number of species protected or species recovery rate (Table 1). Scenarios I and II differ mostly in the amount of fishing effort affected, which is higher in the scenario without costs (I). Scenarios II and III differed only in total area reserved and total number of species protected.

The optimal solution from the point of view of biodiversity conservation (scenario I) would affect 32 % of the

fishing effort. When cost is applied (difference between scenarios I and II), effort affected is reduced to 19.2 %. The same effect is observed in fishing area lost, decreasing from 29.2 % in scenario I to 19.8 % in scenario II.

Scenario III is very inefficient with regards to area, as it requires $\sim 15 \text{ km}^2$ more than any of the other two scenarios. This is of course due to the fact that the reserve system is built around existing SPAs. It simply subtracts the SPAs' total biotope contribution from representation targets, and tries to achieve the remainder. However, given the excess area, the loss of effort is marginal (area available for fishing and effort lost values are similar to those for scenario II).

DISCUSSION

The question that we have ultimately addressed in this paper is 'what is the maximum amount of fishing effort that needs to be affected before a marine park fails to meet representation goals?' The combined use of effective biodiversity surrogates (modeled biotopes) and a detailed spatial measure of fishing effort (VMS) has allowed us to address this question quantitatively and in a transparent way. Given a target to include 10 % of each biotope in the reserve network, the prawn fishery of Kosterhavet stands to lose not more than 19.8 % of its current footprint area if it is to give way to cost-efficient conservation of benthic diversity. This means that for a 10 % representation target, a system of reserves that reallocates more than 19.8 % of

fishing grounds to conservation will be inefficient with regards to the cost for the fishery. To put this into context note that when fishing data are not used (scenario I) the fishery loses 29.2 % of its currently used area. The use of VMS data can warrant an almost 10 % gain of fishing grounds for a fishery while conservation targets can still be achieved.

The results for effort (last column of Table 1) shed light on the non-spatial aspects of this question. The effort gained from incorporating VMS data into the planning process is 12.8 % (32.1–19.25 %). With a function linking VMS ping density and, for example, time spent fishing this information could, at least theoretically, be used to calculate the total gain in revenue that could be achieved.

The largest reserve network is predicted to protect the highest number of species. This is an expected effect of species–area relationships (e.g., Rice and Kelting 1955). Remarkably, area increased only when constraints were placed on which planning units should be incorporated in the conservation network regardless of their biological composition (i.e., SPAs), and not when fishing costs were applied. Therefore, another important conclusion from our study is that no additional area was needed to meet conservation targets while simultaneously preserving fishing grounds. Figure 6 shows how effectiveness of the network increases as fishing area increases too.

In this particular case, the best solution of Scenario III showed the lowest amount of fishing effort affected and highest rate of species protected relative to the other two best solutions. The fact that scenario III is area-inefficient to

Fig. 6 Relative increase in species recovery rate and fishing area with the different Marxan scenarios, where scenario I is without fishing costs included, scenario II is with fishing costs and all planning units available, and scenario III is with fishing costs and SPAs locked in

represent biodiversity but not in relation to fishing effort is because SPAs tried to achieve goals other than representativeness (e.g., uniqueness, etc.) (Píriz 2004; Bruckmeier et al. 2005). This network needed to be complemented with additional area in order to meet representation targets because a number of biotopes were not represented in it. On the other hand, fishing activity remains largely unaffected because SPAs were already unsuitable for fishing and therefore represent *de facto* reserves. If the fact that such a large amount of area is not considered a problem from a managerial point of view, or from other potential costs (e.g., from enforcing regulations) scenario III could provide a desirable solution, as it would address a range of conservation and fishery goals. We must emphasize, however, that the way to address the question of reserve placement, mindful that a fishery does not lose out more than it is acceptable to the fishermen, is by using any one conservation zoning proposal (e.g., a given output from a decision support tool such as Marxan) as starting point for a planning process among stakeholders and conservation officials. This process ensures that decisions made regarding spatial planning will be transparent and justifiable.

Fishing area/effort loss estimates are based on Marxan solutions, which in turn are based on the modeled distribution of biotopes. The biotope distribution model, which had been tested against a subset of the species data used here for measuring the effectiveness of conservation networks, exhibited a 72 % classification accuracy (see Gonzalez-Mirelis and Lindegarth 2012), and therefore, a degree of uncertainty must be attached to the representation levels achieved by each reserve network solution. Particularly, offshore soft sediment was a difficult biotope to classify, and was often misclassified as offshore rock (Gonzalez-Mirelis and Lindegarth 2012). In other words, there might be less offshore rock and more soft sediment, particularly at the bottom of the deep trench of the Koster fjord, than is shown on the map (Fig. 3). As a result, fishing appears to be taking place on biotopes that are known not to be suitable for fishing, and are avoided by the fishermen (i.e., rocky bottoms). Two factors are at play: the certainty with which it is possible to tell the vessel positions representing fishing from those representing transit and the accuracy of the distribution of different biotopes. In a situation where better information was available for both factors there would have been much less overlap between fishing and the offshore rock biotope. Better information on biodiversity, including better models, would therefore enhance the benefits of VMS. As with many spatial planning problems, only if the biological composition of the seabed were known everywhere could fishing areas (and conservation areas) be truly efficiently located.

VMS has the benefit of independently sampling the effort of a given fishing fleet, while an obvious drawback is

that not all vessels are included. In this paper we have assumed that the effort distribution of vessels 15 m or larger indicates also fishing by trawlers smaller than 15 m. During the process that led to the joint management plan (Píriz 2004) it was communicated by the fishermen that the smaller part of the fleet largely explores the same fishing grounds, but that the smaller vessels in addition also utilize smaller trenches and some narrow areas where the larger vessels cannot operate. For the picture to be complete the spatial distribution of this effort needs to be incorporated into the planning process as well, preferably as detailed as possible, e.g., by asking fishermen for their plotter tracks or using independent information from other VMSs like the Automatic Identification System (AIS) that in recent years have become increasingly used.

It is clear that minimizing affected effort has positive effects on the fishery (relative to planning for conservation without information about fishing effort distribution), and we have demonstrated that this need not negatively affect conservation goals, as is commonly thought, if SCP principles are used. However, additional aspects play into this. We have assumed that the density of fishing events will remain the same after reallocation of areas, but of course, fishermen could compensate for a loss of fished area by using other areas more intensely. Redistribution of fishing effort could result in negative effects for biodiversity, as shown by Hiddink et al. (2006). The larger the loss of fishing grounds, the larger the chances that these negative effects will outweigh the positive effects from reducing fishing area. That creating marine reserves has positive effects on the ecosystem is of course the assumption behind this study, and the rationale to establish MPAs to halt biodiversity loss worldwide. Recently, Alemany et al. (2012) presented new empirical data supporting this assumption. With respect to the fishery, redistribution of effort could result in either no extra costs, or increased costs, if fishermen need to use more time or more fuel to maintain their catches.

While closing the minimum amount of area possible has been the central issue in this paper, it is worth also considering the benefits for a fishery of closing any part of its fishing grounds at all. The proposed representation target (unfished areas) of 10 % means that the fishers are still allowed to fish 90 % of each habitat. A number of authors (e.g., Ley et al. 2002; Halpern 2003) have shown that for some types of fishing gear and some species, catches and biomass are larger in areas located in the vicinity of fishing closures. On the other hand, minimizing cost is not the same as maximizing profit. To maximize economic benefits, other measures can be taken besides (or even instead of) minimizing effort affected, for example, placing reserves near access points such as ports so that vessels can benefit from a short cruise time to the reserve's edge (if

spillover effects are expected) or placing reserves in intermediate locations between nodes in a network (Gaines et al. 2010). Another question is whether 10 % representation targets, or a 1:10 protected to non-protected area ratio is enough to ensure the persistence of biodiversity in marine reserves (Gaines et al. 2010).

The use of spatially explicit information like VMS has great potential in marine spatial planning, both from a conservation perspective, and an industry perspective. The more information is transmitted with the VMS signal the more a fishery and the conservation community could gain in the planning of multiple-use MPAs. Primarily, information is needed on whether a vessel is fishing or not. Also, better fleet coverage would be beneficial. Jennings and Lee (2012) demonstrated that detailed VMS data can be used to determine core fishing areas which would retain most of the catch while opening up disproportionately large areas for other activities (e.g., conservation). With additional information attached to the VMS ping such as catch per individual effort it would be possible to better address the problem of spatial planning in completely new ways. For example, Ban and Vincent (2009) explored the possibility of simulating reserves systems designed to maintain a certain amount of catch for all fisheries, at the lowest possible cost for biodiversity, i.e., the problem posed here turned on its head, with conserving fishing grounds as the main objective of the planning exercise. High-resolution fishery data such as VMS, especially when combined with logbooks, can make fishing information match in detail the information on biodiversity that is available from marine surveys or modeling. Therefore, VMS data is optimal for incorporating fishing effort into spatial planning processes. Use of spatially explicit, high-resolution fishing activity data such as VMS data can be invaluable in ensuring that biodiversity conservation planning will not occur at the expense of a fishery, and the fishermen that depend on it.

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