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An approach for assessing and ranking fisheries management scenarios in spatially delimited marine areas

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Spatial restrictions to human activities such as bottom trawling are increasingly used to improve the ecological condition of disturbed habitats. Such management interventions typically have socio-economic consequences, which creates a challenge for those making decisions about which activities should be restricted and where restrictions should apply. We present an approach for predicting the effects of fisheries management scenarios in spatially delimited marine areas and ranking them—using a loss function—according to how well they achieve desired outcomes across a set of ecological and socio-economic indicators. This approach is demonstrated by simulating alternative fishing gear restrictions and zoning options within a hypothetical marine protected area (MPA). Relative benthic status (RBS; an indicator of ecological condition) and relative catch value (RCV; an indicator of potential economic cost) were estimated for the baseline environment and 21 potential management scenarios. The rank order depended on which indicator was prioritized (i.e. whether RBS or RCV was given greater weighting in the loss function), with the top-ranked scenarios in each case involving considerably different management measures. The methods presented can be applied anywhere using locally or strategically relevant indicators to help identify spatial fisheries management measures that minimize ecological and socio-economic trade-offs.

Keywords: conservation, ecology, economic, fishing, marine protected area, marine spatial planning, sustainability, trawling.

Introduction

Technological advances and society's growing demand for food, energy, and other resources are increasingly putting pressure on the world's oceans (Korpinen and Andersen, 2016; Halpern *et al.*, 2019). This pressure is driving ecological change in marine ecosystems, in some cases leading to the depletion of biological resources and local extinctions (Jackson *et al.*, 2001; Lotze *et al.*, 2006). As interactions among marine users become more complex and competition for space increases, resolving the nexus between protection and sustainable use of marine resources is becoming an increasingly challenging task (McShane *et al.*, 2011; Lombard *et al.*, 2019). The United Nations rank the health of the world's oceans among the most pressing development challenges (United Nations, 2022), with the stand-alone Sustainable Development Goal for the oceans and coasts (Goal 14, "Life below water") calling on the international community to "Conserve and sustainably use the oceans, seas, and marine resources for sustainable development". One approach proposed to achieve this goal, both from conservation and fisheries management perspectives, is the designation of marine protected areas (MPAs) (Lubchenco *et al.*, 2003; Jennings, 2009; Rassweiler *et al.*, 2012).

MPAs are being designated globally to protect biodiversity and natural resources (Lubchenco and Grorud-Colvert, 2015). While some constitute strict "no-take" areas, many are treated as multi-use areas, with varying degrees of protection, on the basis that living resources replenish themselves and can therefore be sustainably exploited (Schratzberger *et al.*, 2019). Restrictions in such multi-use MPAs may be placed on specific fishing gears (e.g. those that disturb the seabed) in areas where sensitive features of conservation interest are present. These gears may be permitted in areas where such features are absent, while less damaging activities may be permitted throughout an MPA. Any permitted activities may, however, have ecological effects that are also of concern to managers, such as reduced biomass of target and non-target species and associated changes to ecosystem functioning in the case of fisheries (Hiddink *et al.*, 2006, 2017). The magnitude of effects will depend on the nature, intensity, and spatial distribution of the activities, as well as the ecology of the area (Rijnsdorp *et al.*, 2018; Sciberras *et al.*, 2018). A task of multi-use MPA management is therefore to protect features of conservation interest and mitigate the ecological effects of any permitted activities.

The socio-economic consequences of activity restrictions must also be considered when selecting MPA management measures (Schratzberger *et al.*, 2019; McConnaughey *et al.*, 2020). Commercial fisheries exist because of the demand for seafood; the consumer benefits from a source of protein, and the fisher benefits from a source of income. Spatial restrictions to fishing activity will therefore reduce the flow of resources from the protected area to society and affect the fisher's income and the broader economy. Economic losses may be offset in the short term by the displacement of fishing activity (and its ecological effects) into unprotected areas (Stevenson *et al.*, 2013; Vaughan, 2017), while in the long term, fishery yields may be enhanced if overfished commercial species emigrate to

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unprotected areas as their populations inside protected areas grow (Roberts *et al*., 2001; Hilborn *et al*., 2004; Stobart *et al*., 2009). Nevertheless, restricting fishing activity within an MPA has the potential to reduce food provision and the profit it generates, with the magnitude of this effect constrained by the productivity and value of the fishery and how much the activity is reduced. In the absence of knowledge about how local fishing restrictions will ultimately affect broader fishing activity, stock density, and economic yield, a challenge of multi-use MPA management is to select measures that balance ecological benefits and potential socio-economic consequences.

To help achieve this balance, we outline a series of steps that can be followed to predict the ecological and socio-economic effects of fisheries management measures in multi-use MPAs, or indeed any areas where spatially discrete management is being considered. We apply this approach to a hypothetical MPA by simulating alternative management scenarios for bottom trawl fisheries and determining how relative benthic status (RBS; *sensu* Pitcher *et al*., 2017, an indicator of ecological condition) and relative catch value (RCV, an indicator of potential economic cost) would be affected. Notably, a method is presented for ranking management scenarios according to how well they achieve desired outcomes across the ecological and socio-economic indicators considered.

Materials and methods

The methods described here were carried out across two interlinked systems: a PostGIS spatial database server for spatial data management and analysis and R for statistical analysis (R Core Team, 2023). The spatial fisheries data and auxiliary spatial layers were managed in a centralized PosgreSQL/PostGIS relational database server. The use of a PostGIS spatial database allows SQL spatial analysis scripting and therefore reduces the time required to process large spatial datasets in R code. We used R to interact with the PostGIS database, to retrieve results of SQL analysis and carry out further statistical processing, and to produce graphical outputs.

Indicator selection

A major pressure associated with bottom trawling is physical disturbance to seabed habitats and the resulting depletion of benthic biota (Hiddink *et al*., 2017; Sciberras *et al*., 2018). We indicate this effect using RBS (Pitcher *et al*., 2017), which represents benthic invertebrate community biomass as a proportion of the biomass at carrying capacity (i.e. the untrawled state). Socio-economic effects of management measures can be inferred from indicators of employment, income, or gross economic activity. The socio-economic indicator used here, RCV, represents the commercial value of the catch as a proportion of catch value (CV) at the baseline (i.e. in the absence of any management intervention).

Inputs required

To predict the effect of fisheries management measures on the above indicators, the approach we follow requires information on: (1) the location and boundaries of the area of interest (derived from GIS shape files or by entering coordinates directly or manually drawing boundaries within a geographic area), (2) the existing fishing activity within the area (based on data from vessel monitoring systems (VMS), automatic identification systems (AIS), or fishers' logbooks), (3) the commercial value of the catch within the area (based on landings values reported within logbooks or sales notes), (4) the type and distribution of seabed habitats within the area (based on survey data or modelled habitat maps), (5) the response and recovery of benthic faunal biomass when exposed to fishing activity (taken from meta-analyses or site-specific studies, where available), and (6) possible management scenarios for the area (i.e. which fishing activities are restricted and the zones where these restrictions apply). For (6), management scenarios can be guided by information on the distributions of ecological features (e.g. species of conservation interest) or physical features (e.g. shipwrecks, wind turbines, oil platforms, etc.) within the area. The inputs used for our case study involving a hypothetical MPA are described below. All inputs other than the MPA boundaries and management measures (inputs 1 and 6) are based on real-world (non-hypothetical) data.

Inputs used for our case study

Area of interest

The boundaries of a hypothetical MPA were drawn in the offshore waters of the northwest European shelf (the exact location is not provided to protect information about the commercial catch in the area). The site covers approximately 2750 km^2 of subtidal soft sediment. The commercial catch in the area consists primarily of the Norway lobster (*Nephrops norvegicus*), followed by haddock (*Melanogrammus aeglefinus*), cod (*Gadus morhua*), and saithe (*Pollachius virens*). Sea pens, including aggregations of a fragile and rare sea pen *Funiculina quadrangularis*, are also present. The sources of information on habitat type, catch composition, and *F. quadrangularis* are described for inputs 3, 4, and 6 below. The site was divided into 3 km^2 grid cells to assess spatial variation in fishing activity and its associated ecological and socio-economic effects. Fishing activity is typically randomly distributed at this spatial scale (Rijnsdorp *et al*., 1998).

Fishing activity

The activity of United Kingdom (UK) fishing vessels within the study region from 2009 to 2017 was determined using VMS and logbook records held by the UK Marine Management Organization (MMO). VMS records provided information on the geographic position, time of day, and vessel speed when the signal was transmitted. Logbooks indicated the type of fishing gear used by each vessel and, therefore, at each VMS point. Vessel lengths were extracted from the logbooks and used to estimate gear width from published relationships between these variables for European vessels (Eigaard *et al*., 2016).

Catch value

To estimate CV, data on catch weight by species and its related sale price were obtained from the logbook records of UK vessels using the study region during 2009–17. These data, which are reported at the scale of ICES statistical rectangles (0.5×0.5 degrees; 3225 km^2 for the study region), were apportioned among VMS points within an ICES rectangle according to the time spent fishing at each point (a single VMS point typically represents approximately two hours of fishing). For example, if a VMS point accounted for 10% of time spent fishing within an ICES rectangle during a trip, then it would be given 10% of the catch reported for this ICES rectangle.

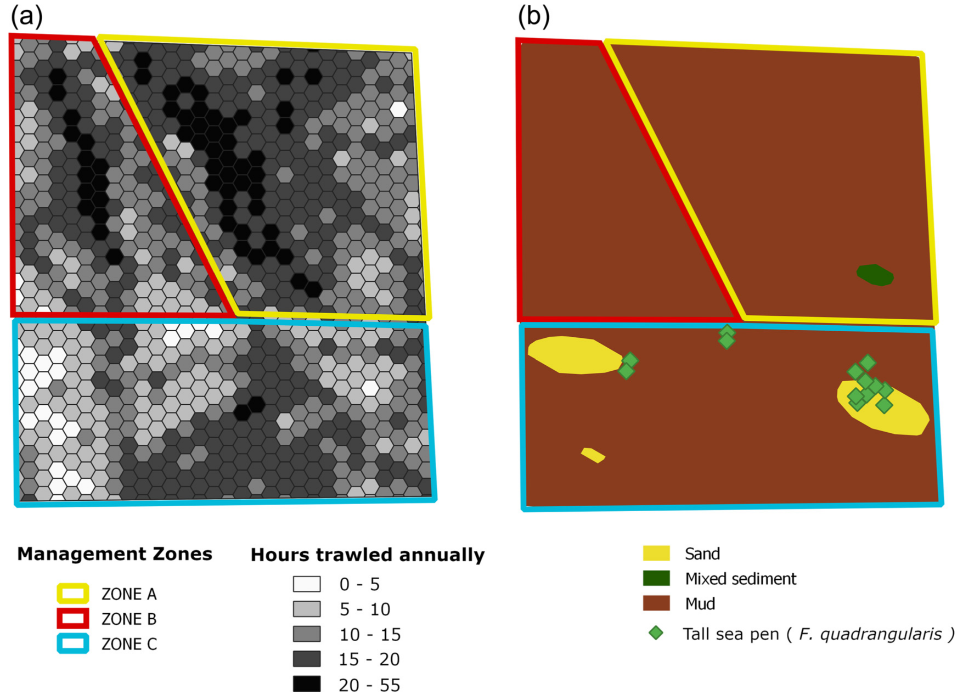

Figure 1. The three management zones of a hypothetical MPA in relation to (a) the distribution of fishing activity (hours per km^2 per year) among 3 km^2 grid cells and (b) seabed habitat type and records of a fragile and rare sea pen (*F. quadrangularis*).

This was done using the *VMStools* package in R (Hintzen *et al.*, 2012).

Habitat

Information on habitat distributions within the study region was obtained from the European Marine Observation and Data Network (EMODnet) broad-scale seabed habitat map (EUSeaMap; www.emodnet-seabedhabitats.eu). Habitat type affects the depth that fishing gear penetrates the seabed and, in turn, influences the fraction of benthic biomass depleted by a pass of the gear. This information was therefore incorporated into estimates of faunal depletion (see input 5 below).

Depletion and recovery of benthos

To estimate RBS (Pitcher *et al.*, 2017), depletion (d; the fraction of benthic invertebrate community biomass removed by a pass of a gear), and recovery rate (r; the intrinsic rate of increase in biomass) must be specified. Mean sediment penetration depth of each gear group [otter trawls (*OT*), beam trawls, and dredges] within each habitat type and the corresponding d were obtained from a global assessment of the effect of bottom trawling on benthic macroinvertebrates (Hiddink *et al.*, 2017). The recovery rate used was $r = 0.82\ y^{-1}$, corresponding to the mean biomass recovery rate for benthic macroinvertebrate communities previously undisturbed by trawling, taken from the same study (Hiddink *et al.*, 2017). While this is likely to be a conservative estimate of r in fishing grounds, as bottom trawling selects for species with fast life histories (Hiddink *et al.*, 2019), we consider it reasonable to incorporate the recovery rates of slower-growing species that are most affected by trawling and are often the focus of conservation efforts.

Management scenarios

The hypothetical MPA was divided into three zones, distinguished by the level of fishing activity (indicated by the density of VMS points; converted to hours spent fishing $km^{-2}\ yr^{-1}$) and the presence of a species of conservation interest, *F. quadrangularis* (Figure 1), to simulate alternative management scenarios. The distribution of *F. quadrangularis* in the area was determined using video imagery data collected during a benthic characterization survey.

The zone in the northeast of the site, *Zone A* (1020 km^2), is characterized by high fishing activity (~6 h $km^{-2}\ yr^{-1}$). *Zone B* is a relatively small area (620 km^2) in the northwest of the site and has a lower level of fishing activity (~5 h $km^{-2}\ yr^{-1}$). *Zone C* (1120 km^2), which makes up the southern section of the site, has the lowest fishing activity (~4 h $km^{-2}\ yr^{-1}$) and is the only zone that is known to contain aggregations of *F. quadrangularis*.

The fishing gears used inside the boundaries of the hypothetical MPA include single-rig *OT* (65% of activity), twin-rig *OT* (20% of activity), and pair trawls (*PT*) (15% of activity). We treat *PT* as if they are "*OT*" when calculating d, as the impact of the weights used for *PT* is comparable to that of the boards and clumps used for single-rig and twin-rig *OT*, respectively (*sensu* Eigaard *et al.*, 2016). However, *PT* are distinct from the other gears in that the net is exceptionally wide (200–300 m), which is possible because it is pulled by a pair of vessels. Using this gear therefore results in a large footprint of seabed disturbance per hour of fishing. Given this distinction, *PT* were separated from *OT* to create gear restriction options.

A total of 21 management scenarios (seven zoning options x three gear restriction options) were simulated, each of which was assessed against the baseline (no management intervention) scenario (Table 1). For simplicity, we did not run scenarios in which different zones were closed to different gears.

Predicting effects of management measures

RBS was estimated for each grid cell using the above inputs and then assessed at the site level by averaging across

Table 1. The different gears (*PT*; *OT*) excluded from zones (*A*, *B*, and *C*) of the hypothetical MPA in each of the 21 management scenarios considered.

	Zone A		*Zone B*		*Zone C*	
Scenario	Pair trawls	Otter trawls	Pair trawls	Otter trawls	Pair trawls	Otter trawls
1	✗					
2		✗				
3	✗	✗				
4			✗			
5				✗		
6			✗	✗		
7					✗	
8						✗
9					✗	✗
10	✗		✗			
11		✗		✗		
12	✗	✗	✗	✗		
13	✗				✗	
14		✗				✗
15	✗	✗			✗	✗
16			✗		✗	
17				✗		✗
18			✗	✗	✗	✗
19	✗		✗		✗	
20		✗		✗		✗
21	✗	✗	✗	✗	✗	✗

The exclusion of a gear is indicated by ✗.

cells. RCV was assessed at the site level only, after first estimating the absolute CV within each grid cell (CV), then averaging CV across grid cells and presenting this value relative to the corresponding baseline value. This was considered preferable to estimating cell-level RCV before averaging across cells, as it accounts for between-cell variation in CV at the baseline and, thus, provides a more accurate assessment of how a management scenario would affect CV over the entire site. As the absolute benthic faunal biomass within grid cells was unknown, no analogous approach was available to assess site-level RBS. A detailed description of the steps taken to predict RBS and RCV for the different management scenarios is provided below. While all gears included in our case study are treated as belonging to the same group with respect to their penetration depth and associated d, we describe methods generalized for use in single and multigear group assessments.

Calculate the trawling footprint within grid cells

Data associated with VMS points (input 2) were aggregated at the grid cell level throughout the delimited area of interest (i.e. the hypothetical MPA; input 1). The trawling footprint (km^2), hereafter referred to as the swept area (SA), was calculated for each gear group in each grid cell,

$$SA_{c,g} = \sum_{i=1}^{n_{c,g}} h_i \cdot s_i \cdot w_i \,, \tag{1}$$

where c is the grid cell, g is the gear group, $n_{c,g}$ is the number of fishing events (VMS points) by gear group g in cell c, h_i is the duration of the ith fishing event [i.e. hours between fishing event i and the directly preceding event (i–1)], s_i is the instantaneous vessel speed (km h^{-1}) of the ith fishing event, and w_i is the width (km) of the gear used during the ith fishing event (Gerritsen *et al.*, 2013).

$SA_{c,g}$ was averaged over the nine years of VMS data (2009–17) to give the mean annual SA (km^2 y^{-1}) for each gear group in each grid cell. This was converted to the swept area ratio (SAR) by dividing by the cell area (km^2). SAR (y^{-1}) indicates the average number of times per year the seabed within a grid cell is disturbed by fishing gear.

Calculate CV and RBS within grid cells

CV was calculated for each grid cell by summing the commercial value of the catch associated with all VMS points within the grid cell during 2009–17 (input 3) and dividing by the number of years of VMS data that were used (i.e. nine).

RBS was calculated using a modified version of that presented in Pitcher *et al.* (2017), wherein we assumed that depletion per trawl pass is a constant fraction of the biomass that remains following previous trawl passes (i.e. depletion as a proportion of carrying capacity is additive on a log scale). First, the proportion of each grid cell covered by each habitat type was determined by overlapping the grid onto the imported habitat map (input 4). SAR and d (input 5) were then used to calculate the total mortality within each grid cell,

$$M_c = \sum_{g=1}^{n_{1,c,g}} \sum_{h=1}^{n_{2,c,h}} SAR_{c,g} \cdot \; -\ln\left(1 - d_{c,g,h}\right) \; \cdot A_{c,h}, \tag{2}$$

where $n_{1,c,g}$ is the number of gear groups in cell c, $n_{2,c,h}$ is the number of habitats in cell c, $SAR_{c,g}$ is the SAR for gear group g in cell c (assumed to apply to all habitats within the cell), $d_{c,g,h}$ is the depletion caused by gear group g when towed over habitat h in cell c, and $A_{c,h}$ is the proportion of cell c that is covered by habitat h. M_c was then converted to the total biomass lost (L) from cell c per year as a proportion of the carrying capacity,

$$L_c = 1 - \exp\left(-M_c\right). \tag{3}$$

As in Pitcher *et al.* (2017), the recovery rate was adjusted by converting r (input 5) to R to account for the assumed random distribution of fishing activity within grid cells (Ellis *et al.*, 2014),

$$R_c = \sum_{g=1}^{n_{1,c,g}} \sum_{h=1}^{n_{2,c,h}} \frac{(r \cdot d_{c,g,h})}{-\ln\ (1 - d_{c,g,h})}. \quad (4)$$

Finally, RBS was calculated for each grid cell from L_c and R_c,

$$RBS_c = 1 - \frac{L_c}{R_c}. \quad (5)$$

Assess site-level ecological and socio-economic effects

RBS was averaged over all grid cells within the hypothetical MPA to indicate the ecological condition (from 0 to 1) at the site level under the baseline scenario. CV was averaged over all grid cells to give the baseline mean annual value of catch harvested within the site, which was used as the reference against which site-level socio-economic effects of management scenarios were assessed.

For each management scenario (i.e. the gear restriction and zoning options specified in input 6), ecological effects were predicted by recalculating RBS at the grid cell level and then the site level after removing data for fishing activity targeted by the interventions. CV was recalculated in the same way to predict socio-economic effects in each grid cell. Then, to predict socio-economic effects at the site level, RCV was calculated,

$$RCV_s = \sum\nolimits_{i=1}^{n_c} CV_{c,s} / \sum\nolimits_{i=1}^{n_c} CV_{c,\ s0}, \quad (6)$$

where $CV_{c,s}$ is the mean annual value of the catch that would be harvested within grid cell c under management scenario s and $CV_{c,s0}$ is the mean annual value of the catch harvested within grid cell c in the baseline scenario ($s0$). As with RBS, RCV ranges from 0 to 1 (whereas CV ranges from 0 to the highest mean annual CV within any grid cell in the baseline scenario). It should be noted that because RBS and RCV are assessed only within the area of interest, the predictions do not incorporate any effects of management measures that may transcend this area (e.g. displacement). It is also assumed that any removed fishing activity is not replaced by new activity using any gears that remain permitted within the area of interest.

Ranking management scenarios

We formulated an approach for ranking management scenarios according to their predicted site-level ecological and socio-economic effects using a loss function. For each indicator, the values at which there is a minimal and maximal "loss" are specified. These values act as thresholds beyond which all possible outcomes for the indicator are considered highly desirable and highly undesirable, respectively. The relative importance of achieving a desired outcome for each indicator is also specified by assigning a weighting factor to their loss. The weighted loss across all indicators is then summed and used to rank the scenarios from most to least desirable overall. A formal description of the loss function is provided in Supplementary Text S1 in the online supplementary material.

For our case study, we set thresholds for "highly desirable" and "highly undesirable" outcomes at 0.90 and 0.50 for RBS and 0.75 and 0.25 for RCV (Figures 2a, b). We then used the approach described above to rank the 22 scenarios (one baseline scenario and 21 management scenarios) under each of the following prioritizations: (1) ecological benefit is the priority (80/20 weighting in favour of RBS), (2) mitigating potential socio-economic consequences is the priority (80/20 weighting in favour of RCV), and (3) equal importance is given to ecological and socio-economic effects (50/50 weighting for RBS and RCV). The surface of the loss function was drawn using a colour gradient to show the overall desirability of potential outcomes for RBS and RCV under prioritizations 1, 2, and 3 (Figures 2c–e). For all 22 scenarios, site-level RCV and RBS were plotted onto Figures 2c–e and the top-ranked scenarios for prioritizations 1, 2, and 3 were marked to allow visual comparison to the baseline. RBS and CV of the baseline and top-ranked scenarios were also illustrated at the grid cell level using maps (Figure 3) and density plots (Figure 4) to provide higher-resolution spatial context to the site-level outputs.

Results

RBS was spatially heterogenous within the hypothetical MPA under baseline conditions (i.e. when no fishing gears are restricted in any of the three zones; Figure 3). The number of grid cells with each RBS value (0–1) followed a unimodal distribution that peaked at 0.73 (Figure 4a, grey line). At the site level, RBS was 0.73 in the baseline scenario (i.e. Scenario 0 in Table 2), meaning that the average benthic biomass within the MPA was predicted to be 73% of what it would be in the absence of fishing activity (see Scenario 21 in Table 2). Site-level RBS for all other management scenarios was therefore bounded between 0.73 and 1.00. CV showed approximately the opposite spatial pattern as RBS (Figure 3b; Figure 4b, grey line). As the site-level socio-economic indicator RCV is measured in relation to the baseline, values ranged from 1.00 when no restrictions were applied to 0.00 when all zones were closed to all gears (Table 2).

The 21 management scenarios and the baseline scenario are ranked in Table 2 based on the loss function representing the desirability of different possible outcomes for RBS and RCV (the smaller the loss, the more desirable; Figures 2a, b). Rankings are presented with: (1) RBS prioritized over RCV (80/20 weighting; see "Ecological" column), (2) RCV prioritized over RBS (80/20 weighting; see "Socio-economic" column), and (3) equal importance given to RBS and RCV (50/50 weighting; see "Equal" column).

Ecological priority

The top-ranked scenario when ecological benefit was prioritized (Figure 2c) was the closure of *Zones B* and *C* to all trawlers (see "Ecological" column in Table 2). RBS is predicted to be maximized and CV minimized throughout these two zones (Figures 3c, d), causing grid cells with an RBS of 1 and a CV of 0 to become predominant within the MPA (Figure 4, purple lines). With *Zone A* unaffected in this scenario (Figures 3c, d), the modes observed for RBS and CV under baseline conditions remain apparent but much less pronounced (i.e. far fewer cells have these values; Figure 4, purple *vs*. grey lines).

At the site level, implementing this scenario is predicted to increase RBS from 0.73 to 0.87 (Table 2), indicating an increase in benthic community biomass from 73 to 87% of the carrying capacity. This value falls slightly short of the thresh-

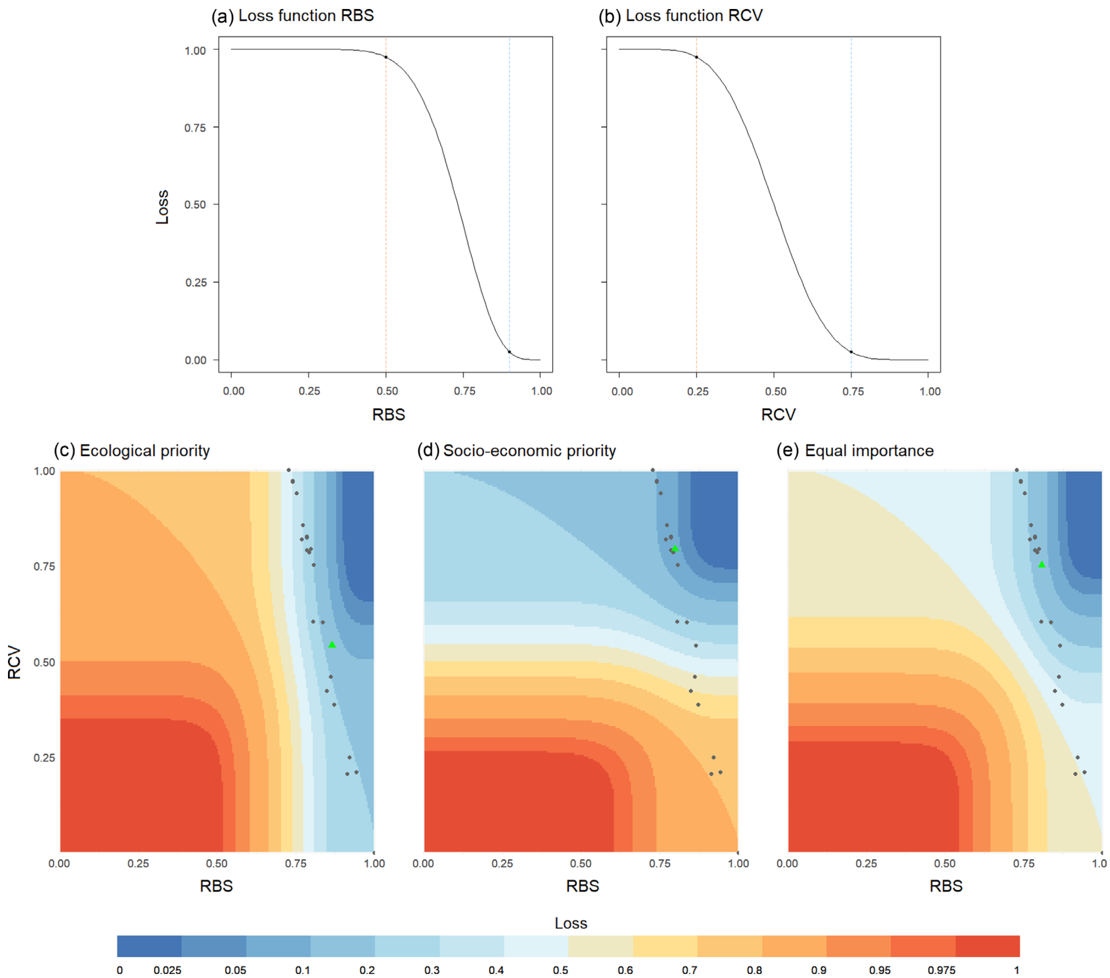

Figure 2. The loss function for (a) RBS and (b) RCV, and the combined loss for RBS and RCV when (c) the ecological outcome was prioritized (weighted 80/20 in favour of RBS), (d) the socio-economic outcome was prioritized (weighted 80/20 in favour of RCV), and (e) equal weighting was given to ecological and socio-economic outcomes. RBS and RCV for the baseline and the 21 management scenarios are plotted in (c–e). The top-ranked scenario is marked by a green triangle.

old for "highly desirable", set at 0.90. RCV, on the other hand, is predicted to decline from 1.00 to 0.54 (Table 2), meaning a predicted 46% reduction in the value of catch harvested within the MPA. This decline does not take RCV beyond the threshold of 0.25 at which the outcome is considered "highly undesirable", but RCV is substantially lower than the threshold for "highly desirable" at 0.75.

Although impacts on features of conservation interest were not incorporated into the rankings for our case study, implementing this scenario would have a secondary benefit of protecting the fragile and rare sea pen *F. quadrangularis*, which has only been recorded in *Zone C* and was used to delineate its boundary (Figure 1b). Moreover, this scenario does not restrict the use of any gears within the area where fishing activity is highest (*Zone A*; Figure 1a).

Socio-economic priority

The top-ranked scenario when mitigating potential socio-economic consequences was prioritized (Figure 2d) was the closure of all three zones to pair trawlers only (see "Socio-economic" column in Table 2). This is predicted to cause RBS within a strip of grid cells in the centre of the MPA to increase from low (0.0–0.4) to moderate levels (0.4–0.8) and become less distinct from the surrounding grid cells (Figure 3e). Areas with predominantly high RBS (0.8–1.0) are also predicted to expand, particularly in the northwest of the MPA (Figure 3e). These changes make the unimodal cell frequency distribution for RBS become more pronounced and shift to a higher value than was observed at the baseline (Figure 4a, brown line). The predicted effect of this scenario on CV is approximately the opposite of the effect on RBS at the grid cell level (Figure 3f; Figure 4b, brown line).

At the site level, an increase in RBS from 0.73 to 0.80 is predicted for this scenario (Table 2), which equates to half the increase predicted for the top-ranked scenario when ecological benefit was prioritized. RCV, on the other hand, is predicted to decline to 0.79 (Table 2), less than half the reduction predicted for the former scenario and above the threshold for "highly desirable" set at 0.75. The limited effect of removing pair trawlers throughout the site reflects the small proportion of fishing activity contributed by this gear (15%).

Unlike the top-ranked scenario when ecological benefit was prioritized, this scenario would not completely exclude fishing activity from *Zone C*, suggesting that protection of the species of conservation interest that occupies this zone may be limited. However, as areas of this zone with

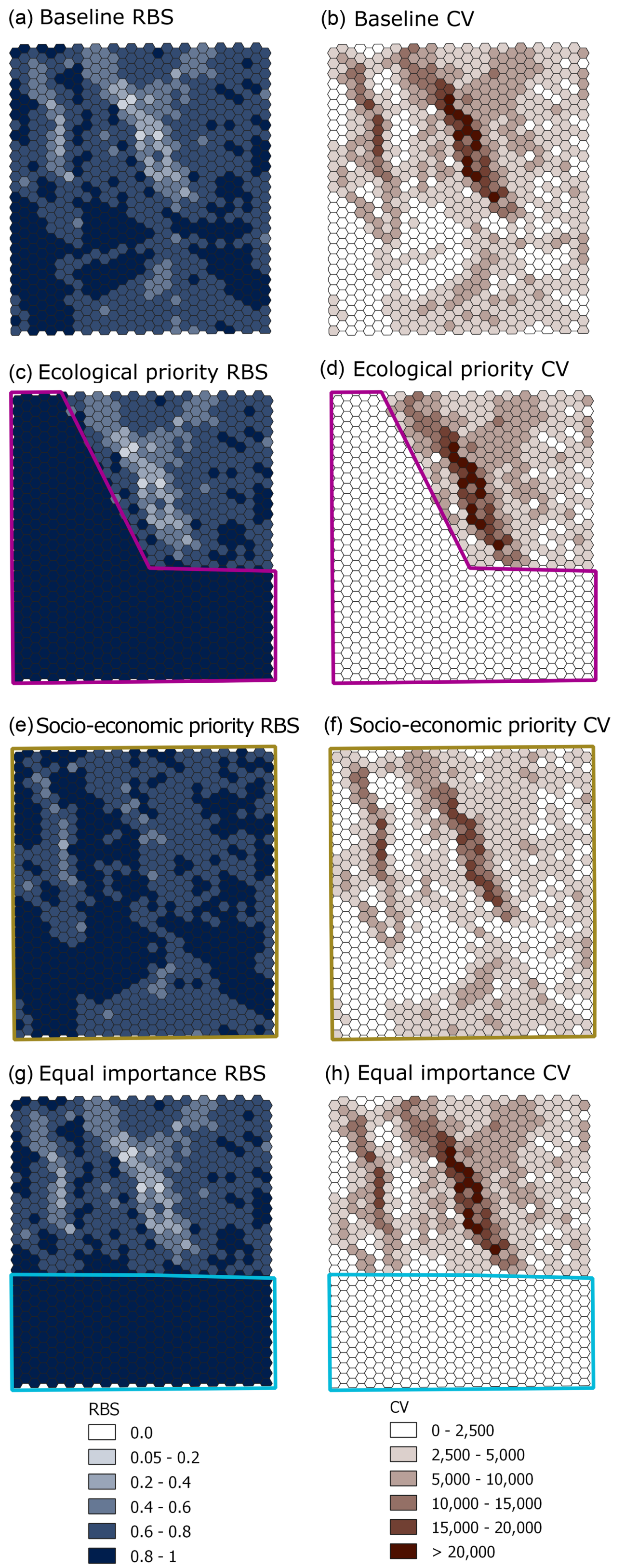

Figure 3. Predicted RBS (RBS; left-hand panels) and CV (CV; right-hand panels; converted to arbitrary units) in 3 km^2 grid cells within a hypothetical MPA. Scenarios include: (a–b) the baseline (no management intervention) scenario, (c–d) the top-ranked scenario when the ecological outcome was prioritized (*Zones B* and *C* closed to all trawlers), (e–f) the top-ranked scenario when the socio-economic outcome was prioritized (all zones closed to pair trawlers), and (g–h) the top-ranked scenario when equal importance was given to ecological and socio-economic outcomes (*Zone C* closed to all trawlers). See Table 2 for rankings.

high RBS (0.8–1.0) are predicted to expand (Figure 3e), notably where aggregations of *F. quadrangularis* have been observed (see Figure 1b), this scenario would possibly benefit this feature to some extent despite not guaranteeing its full protection.

Equal importance of ecological and socio-economic effects

When equal weighting was given to ecological and socio-economic effects (Figure 2e), the top-ranked scenario was the closure of *Zone* C to all trawlers (see "Equal" column in Table 2). This is predicted to maximize RBS and cause CV to decline to zero throughout *Zone C*, with no changes from baseline conditions in *Zones A* and *B* (Figure 3g, h). The predicted effect of this scenario on the cell frequency distributions for RBS and CV is similar to that of the top-ranked scenario when ecological benefit was prioritized, though changes from the baseline are expected to be smaller (Figure 4).

Unsurprisingly, the predicted site-level effects are intermediate to those of the top-ranked scenarios when ecological and socio-economic effects were prioritized, with RBS increasing from 0.73 to 0.81 and RCV decreasing from 1.00 to 0.75 (Table 2). Despite involving similar measures to the former scenario (*Zones B* and *C* to closed to all trawlers), the predicted site-level effects on RBS and RCV are closer to those of the latter scenario (all zones closed to pair trawlers). The Predicted RBS falls short of the threshold of 0.90 at which the outcome is considered "highly desirable", but the corresponding threshold for RCV, set at 0.75, is achieved.

Like the top-ranked scenario when ecological benefit was prioritized, implementing this scenario would fully protect the area where the species of conservation interest have been observed (*Zone C*) (see Figure 1b). There would be no effect in the two areas with relatively high fishing activity (*Zones A* and *B*).

Discussion

We have presented an approach for predicting the effects of spatially discrete fisheries management scenarios and ranking them according to the desirability of their predicted ecological and socio-economic outcomes, as specified by the user (e.g. an MPA or fisheries manager). The top-ranked scenarios are those that minimize trade-offs between conflicting preferences—or, where possible, maximize mutual benefits—for the indicators considered. Rankings can be weighted according to the importance placed by the user on achieving desired outcomes for each indicator. These outputs, especially when used alongside other relevant information (e.g. the spatial distribution of species of conservation interest), provide a means for identifying potentially suitable management measures from a range of possible scenarios within a spatially delimited marine area of interest. The code we used to rank management scenarios is provided in the online supplementary material.

In our case study, the management measures with desired ecological benefits tended to have undesired socio-economic consequences, with RBS and RCV strongly negatively correlated (Pearson's $R = -0.98$) across the 22 scenarios considered (Table 2). In such circumstances, high values cannot be achieved simultaneously for ecological and socio-economic

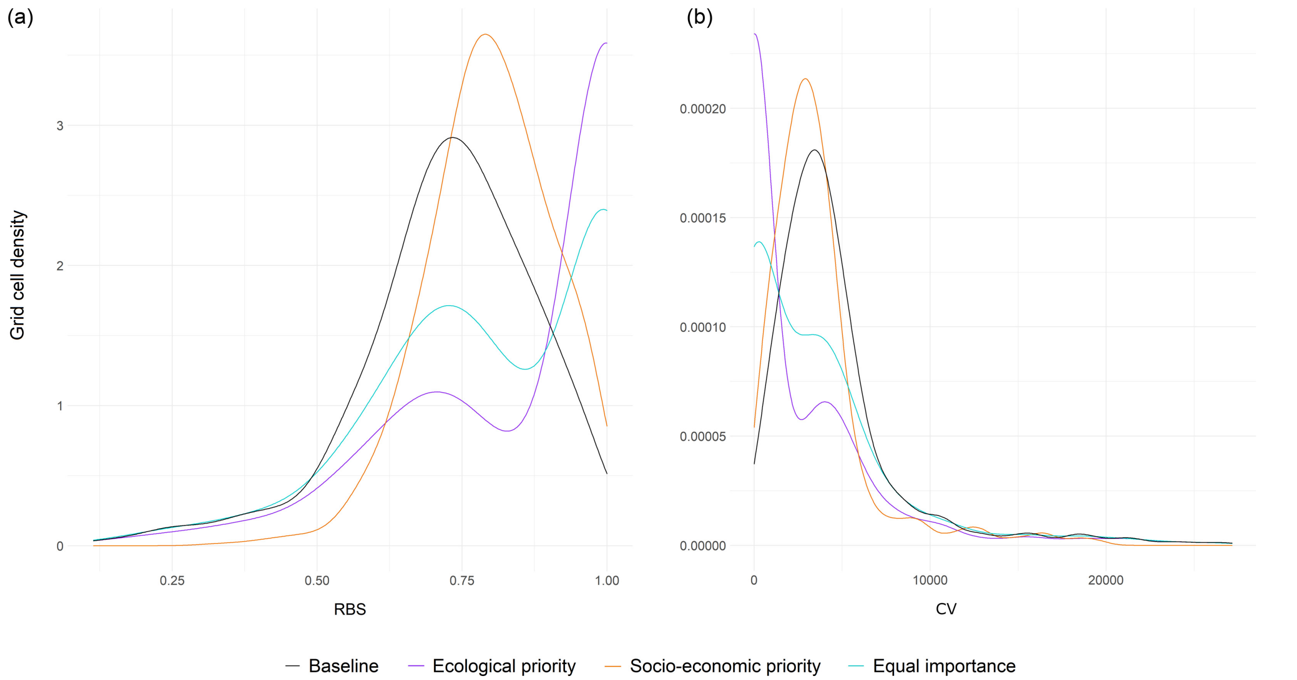

Figure 4. Density of grid cells with each value of a) RBS and b) CV (CV; converted to arbitrary units) in a hypothetical MPA. Distributions are shown for the baseline scenario (no management intervention; grey line), the top-ranked scenario when the ecological outcome was prioritized (*Zones B* and *C* closed to all trawlers; purple line), the top-ranked scenario when the socio-economic outcome was prioritized (all zones closed to pair trawlers; brown line), and the top-ranked scenario when equal importance was given to ecological and socio-economic outcomes (*Zone C* closed to all trawlers; blue line). See Table 2 for rankings.

Table 2. The predicted RBS and RCV for each management scenario, and the ranking of each scenario when the ecological outcome (high RBS) was prioritized, when the socio-economic outcome (high RCV) was prioritized, and when RBS and RCV were considered equally important.

					Ranking		
Scenario	Zone	Gear	RBS	RCV	Ecological	Socio-economic	Equal
0	Baseline (none)		0.73	1.00	22	12	15
1	A	Pair	0.77	0.86	17	7	7
2	A	Otter	0.80	0.60	12	14	12
3	A	All	0.86	0.46	3	16	16
4	B	Pair	0.74	0.97	20	10	13
5	B	Otter	0.77	0.82	18	8	9
6	B	All	0.79	0.79	15	6	6
7	C	Pair	0.74	0.97	21	11	14
8	C	Otter	0.79	0.78	11	4	3
9	C	All	0.81	0.75	4	5	1
10	A and B	Pair	0.79	0.83	14	2	4
11	A and B	Otter	0.85	0.42	13	17	17
12	A and B	All	0.92	0.25	7	19	19
13	A and C	Pair	0.78	0.82	16	3	5
14	A and C	Otter	0.87	0.39	10	18	18
15	A and C	All	0.94	0.21	6	20	20
16	B and C	Pair	0.75	0.94	19	9	10
17	B and C	Otter	0.83	0.60	2	13	8
18	B and C	All	0.87	0.54	1	15	11
19	A, B, and C	Pair	0.80	0.79	8	1	2
20	A, B, and C	Otter	0.91	0.21	9	21	22
21	A, B, and C	All	1.00	0.00	5	22	21

The "Zone" column indicates which zones are managed and the "Gear" column indicates which gears are excluded.

indicators. This highlights the importance of paramaterizing the loss function used to rank management scenarios in a way that accurately reflects desired outcomes, a point emphasized by the fact that adjusting these parameters can produce very different rankings (see Supplementary Text S2 in the online supplementary material). The threshold at which the outcome for an ecological indicator is considered highly desirable could, for example, be set at levels where ecosystem function is unlikely to differ from natural conditions, while for a socio-economic indicator, this threshold may be set at a level where fishery profitability is maintained. Thresholds that define highly undesirable outcomes might be set at the point where a fishery would be commercially inviable or where ecosystem functioning is likely to be severely impaired. In cases where there is not such a strong negative correlation between indicators across management scenarios, or where thresholds for desirability are set at lower levels than in our case study, simultaneously achieving desired ecological and socio-economic outcomes may be possible.

The ecological indicator used in our case study, RBS, relies on published depletion and recovery parameters rather than site-specific ecological data (Pitcher *et al.*, 2017). This characteristic has prompted its use in global assessments of trawl effects (Mazor *et al.*, 2021; Pitcher *et al.*, 2022) and makes it particularly valuable when benthic community data are limited. The socio-economic indicator, RCV, is likely to be useful to anyone interested in the potential effects of management intervention on fishery revenue, though calculating RCV does require information about the landed catch and its origin. Other indicators may also be useful to managers, such as catch weight or composition, which could be considered important from both social (e.g. food provision) and ecological (e.g. species conservation) perspectives. Equally, catch metrics that are calculated relative to time spent fishing or fuel consumption could be informative from both economic (e.g. energy cost) and environmental (e.g. carbon emission) perspectives (Bastardie *et al.*, 2010). The approach presented here can be used to predict effects on any such indicators in the same way we have shown for RBS and RCV.

We selected a small number of management scenarios (three gear restriction options crossed with seven zoning options) for our case study to demonstrate the utility of the approach. However, the assessments could easily be expanded to a broader range of scenarios, such as closing different zones to different gears or applying seasonal fishing restrictions (e.g. Rife *et al.*, 2013). Our approach could also be applied at a broader spatial scale, whereby the effects of closing some MPAs to fishing but not others are predicted. That is, rather than considering different zones within a single site, the assessment could instead consider different sites within a wider MPA network and help to identify suitable no-take areas or "highly protected marine areas" (HPMAs). There is increasing recognition of the conservation benefits of HPMAs, but managers also need to be able to evaluate the ecological and socio-economic trade-offs of such designations (Schratzberger *et al.*, 2019; Benyon *et al.*, 2020). The approach presented here could be useful in this respect.

An important caveat to the outputs from our approach is that they apply only within the delimited area of interest and do not necessarily capture the broader consequences of management measures. Any fishing activity removed from an area may be displaced elsewhere. This could offset the socio-economic effects implied by the outputs, though significant financial losses may remain. For example, closures in the Bering Sea pollock fishery resulted in an estimated loss of ~US$2000 per haul on average (Haynie and Layton, 2010). Over the long term, the removal of fishing activity from one area may cause catches to increase in adjacent areas as unexploited stocks grow and emigrate (Roberts *et al.*, 2001; Stobart *et al.*, 2009). This could theoretically more than compensate for the immediate decline in RCV that occurs in the area where restrictions are enforced, though such "spillover" effects may only be expected for stocks that have previously experienced substantial declines in recruitment due to overfishing (Hilborn *et al.*, 2004). It must also be considered that displacement of trawling means displacement of its ecological effects (Vaughan, 2017), and that these effects could be exacerbated if the new fishing grounds were previously undisturbed (Sciberras *et al.*, 2018). Theoretically, the effects of displacement on ecological indicators such as RBS could be incorporated into our approach by simulating the redistribution of fishing activity into surrounding waters, which would then be included as part of the area within which the effects of management measures are predicted. Forecasting displacement is notoriously challenging (e.g. Vaughan, 2017), but plausible assumptions can be applied, for example, redistributing the removed activity into surrounding waters such that the spatial distribution of activity is maintained (Pons *et al.*, 2022). Doing this would allow the effects of management measures to be assessed in a broader context rather than focusing only on the area of direct management interest, as we have done here.

Even within the delimited area of interest, several assumptions of our assessments require consideration. To predict CV, we assumed that the fishing activity and yield of preceding years are representative of future years. This assumption could be invalidated if exploitation rates are unsustainable, if shifts in species distributions occur (e.g. due to climate change), or if there are changes to market demand for species in the area of interest (Perry *et al.*, 2005; Last *et al.*, 2011; Merino *et al.*, 2012). Moreover, changes to market demand could invalidate our assumption that the same catch will have the same value over time (see Delgado *et al.*, 2003, Chapter 3). Our assessments also assume that gear restrictions will not affect the use of permitted gears within the area of interest. However, one gear may be replaced by another (i.e. "substitution effort"; Vaughan, 2017), thus potentially moderating the predicted changes to ecological and socio-economic indicators. If gear restrictions were determined to be an unsuitable management intervention at a site, then restrictions could equally be based on vessel length, engine capacity, or any other characteristic considered to be an appropriate focus of management. An analogous assumption about consistency in permitted fishing activity would be required but may be more (or less) plausible depending on the local context.

The approach we have outlined can be applied anywhere that spatially discrete fisheries management interventions (e.g. gear restrictions and zoning) are being considered. The only inputs required to produce the outputs we have presented, other than information that can be obtained through literature review, are data on fishing activity, the associated catch, and the seabed habitat types in the area of interest. However, anyone applying this approach must consider the suitability and quality of the inputs they use. Fishing activity data are often unavailable in the form of raw VMS points or at the scale that these points were aggregated for our case study (3 km^2). Having to reduce the spatial resolution of activity data (e.g. to c-squares; 0.05° x 0.05°) or being unable to directly link activity to catch (e.g. because data in either case are unavailable for individual fishing trips) could reduce the accuracy of outputs and, therefore, make them less reliable for informing management decisions. Having incomplete data on fishing activity could have a similar effect. For example, our case study relied on data for UK vessels only, which we assumed to be representative of the whole fleet to illustrate our approach; however, if activity of the remainder of the fleet differed from that of UK vessels, then the data used would not be wholly representative. The information used to parameterize responses to management measures should also be carefully considered. For example, we used depletion (d) values obtained via global meta-analysis, as has typically been applied elsewhere to estimate RBS (e.g. Mazor *et al.*, 2021; Pitcher *et al.*, 2022). If values can instead be obtained at a regional scale, or indeed from studies conducted in the area where management intervention is being considered, then the outputs may more accurately resemble the response of the affected benthic invertebrate community.

We envision that the approach presented here will assist in the decision-making process for spatial fisheries management. Specifically, it can help managers limit the potential socio-economic consequences required to secure ecological benefits within delimited areas of interest (e.g. a multi-use MPA or network of MPAs). Others have developed methods to simulate alternative scenarios for the management of fish stocks and identify the best options, a field broadly referred to as management strategy evaluation (MSE; Punt *et al.*, 2014). It is important to note that our area-based methods are not directly relevant to stock-based management; indeed, the outputs of our approach say little about population-level effects or, indeed, any other effects that may transcend the area of interest. The loss function we used to rank alternative scenarios according to their overall desirability could, however, be applied wherever potential management measures have predictable effects on ecological and socio-economic indicators. Our approach in its current form relies on areas of interest (e.g. MPAs) being delineated and divided into zones before management scenarios are assessed and ranked. This may be informed by various factors, including the spatial distribution of species and habitats of conservation importance, biodiversity and productivity hotspots, and the footprint of fishing activity (Smith *et al.*, 2009; Gaines *et al.*, 2010; Fulton *et al.*, 2015). A possible direction in which the approach could be developed is to simulate gear restrictions over many combinations of grid cells and use the results to delineate zones within an MPA (or MPAs within a wider region) where specific interventions would produce the most desirable outcomes overall. Such an approach would help determine both the spatial configurations of MPAs and how activity within them could be optimally managed.

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Supplementary data

Supplementary material is available at the *ICESJMS* online version of the manuscript.

Conflict of interest

The authors have no conflicts of interest to declare.

Author contributions

Conceptualization: RM, DSC, FG, ET, SW, and MS; method development and coding: RM, MAS, and DSC; data acquisition: RM; writing—original draft: DSC, FG, ET, RM, MAS, and MS; writing—review and editing: DSC and MS.

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Data availability

The raw VMS and logbook data we used to assess fishing activity and CV are held by the MMO and are not publicly available. Bottom trawling data for European seas, derived from VMS points and aggregated at a resolution of 0.05° x 0.05°, can be downloaded from the OSPAR Data and Information Management System (ODIMS) (https://odims.ospar.org/en/maps/map-bottom-fishing-i_-surface-subsurface_khexe/). Data on the distribution of marine habitats in Europe can be downloaded from the European Marine Observation and Data Network (EMODnet) (www.emodnet-seabedhabitats.eu).

Data on the predicted values for RBS and CV at the grid cell level for each of the scenarios (including the baseline) in our case study are provided in the online supplementary material. To maintain anonymity of the hypothetical MPA location and protect information about the commercial catch in this area, the original projected spatial grid for the study site has been transformed into a Cartesian coordinate plane, with the reference grid origin at $x = 0$ and $y = 0$. Grid cells were defined with a regular increment of 0.05 in both the x and y axes to maintain the original spatial grid. CV data are also presented in arbitrary units. The code used to analyse cell-level data and rank the scenarios at the site level is also included in the supplementary material.

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