PARTICIPATORY PLANNING AND DECISION SUPPORT FOR ECOSYSTEM BASED FISHERIES MANAGEMENT AT THE WEST COAST OF SCOTLAND

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ABSTRACT

Mixed fisheries and the marine ecosystems that sustain them are complex entities and involve multiple and potentially conflicting management objectives and stakeholder interests. The presence of multiple trade-offs complicates the identification of strategies that satisfy various policy requirements while being acceptable to affected stakeholder groups. This creates a demand for tools and processes that support learning, cooperation and planning. We report on the application of decision support methodology used in combination with a co-creation approach to scenario based planning for the demersal fisheries of the West coast of Scotland. These fisheries face significant challenges, such as the depletion of key stocks and increased predation by seals. In collaboration with stakeholders we identified generic management alternatives and indicators to evaluate their performance in a structured evaluation using Multi Criteria Analysis. We identify the potential and limitations of this approach and suggest how it can contribute to Ecosystem Based Fisheries Management. This approach does not provide tactical management advice, but stimulates learning and creates an opportunity for stakeholders to search for strategic and policy relevant solutions in an EBFM context.

Key words: Co-creation, EBFM, Ecopath with Ecosim, decision support, Multi-Criteria Analysis.

1. Introduction

Mixed fisheries and the marine ecosystems that sustain them are complex and involve multiple and potentially conflicting management objectives and stakeholder interests. With a single stock approach to fisheries management these conflicts may remain unarticulated and thus outside the management focus. Dolan et al. (2016) describe how ecosystem management aspects are considered within a continuum from focussing on single-species to systemic and multi-sector perspectives. They place the notion of Ecosystem Based Fisheries Management (EBFM) within a hierarchy of ecosystem management concepts as involving "…a system-level perspective on fisheries in an ecosystem". In EBFM, the conflicting goals of harvesters of prey species and harvesters of predator species become explicit as trade-offs. The presence of multiple trade-offs complicates the identification of management strategies that satisfy policy requirements while being acceptable to stakeholder groups. A key challenge for EBFM is to present trade-offs and to arrive at compromises between multiple concerns in a transparent manner while avoiding information overload.

The European Union is committed to progress towards an ecosystem approach for the management of fisheries and the marine environment. Two main policies include this commitment, namely the Common Fisheries Policy (CFP; EC 2013) and the Marine Strategy Framework Directive (MSFD; EC 2008). In recent years a number of ecosystem models have been established for fisheries in European areas (Hyder et al., 2015), but their role in supporting the implementation of EBFM seems limited due to several barriers. These include: Institutional mismatch, difficulties in obtaining reliable data to parameterise ecosystem models (e.g., diet composition), uncertainty due to the large number of ecological processes modelled, difficulties with finding legitimate and efficient ways to accommodate stakeholders in planning and decision-making, and difficulties with integrating biological, economic and social information in a common framework (Christensen and Walters, 2004, 2005; Ramirez-Monsalve et al. 2016a,b; Ounanian et al., 2012; Benson and Stephenson, 2018).

We aim to contribute to progress with implementing EBFM through a case study in a European setting, namely the demersal fisheries off the west coast of Scotland. The case study forms a part of a large European research project, MareFrame[1], which was funded to remove barriers that prevent a more widespread use of EBFM in Europe. Each of the project's seven case studies engaged stakeholders in an iterative and structured planning process, utilizing outputs of ecosystem-models together with decision support methodology.

Multi Criteria Analysis (MCA) was used as the main decision support method in most case studies. In recent decades, MCA has increasingly been used in environmental planning and decision making, because it helps to deal with complex problems (Huang et al., 2011). However, we are unaware of earlier cases where MCA has supported participatory and structured scenario evaluation in the context of EBFM.

[1] http://mareframe-fp7.org/ (last visited 20.06.18).

MareFrame deployed a co-creation approach to generate credible, policy relevant and legitimate knowledge (see Ballesteros et al., this issue). Co-creation is considered particularly relevant for transdisciplinary and problem oriented research. Transdisciplinary research projects involve "…academic researchers from different unrelated disciplines as well as non-academic participants, such as land managers, user groups, NGOs and the general public, to create new knowledge and theory and research a common question" (Tress et al., 2004). The project research team for this case study comprised experts in fisheries modelling, decision support, and fisheries governance. This team cooperated with stakeholder representatives involved with planning and decision making for fisheries and marine conservation.

A central feature of co-creation is to involve stakeholders in a continuous and iterative research process. The process comprises the stages of co-design and co-production, including (co-) dissemination of results (Mauser et al., 2013). The co-design phase identified the main issues in the context of governance and policy and outlined the general research approach, given the available expertise, data and time. Hence, the case study was not framed by the concerns and interests of the stakeholders alone, but also by relevant policies and practical constraints. In the co-production phase a decision support framework, including several relevant resources was developed. The stakeholders tested the framework and provided feedback on its potential for further development and use.

The aim of this work is to report on the approach, the outcomes and the overall experience of a co-creation approach in scenario based planning with MCA. We identify the potential and limitations of this approach, and suggest how it may contribute to advance EBFM in Europe. Ultimately we aim to illustrate how MCA and co-creation may support the operationalisation of EBFM.

2. Material and methods

Following a common planning approach (Gregory, 2012), we defined alternative management scenarios, simulated their likely performance using a foodweb ecosystem model (Ecopath with Ecosim, EwE), and conducted a structured evaluation of the scenarios with MCA. This was carried out in cooperation with stakeholders as organised into five steps, of which the first three can be taken to represent the co-design phase of co-creation, with the subsequent steps respectively representing co-production and co-dissemination:

1. Identify the overall goals and problem scope of the case study
2. Identify objectives and indicators
3. Identify management scenarios
4. Estimate scenario impacts with models
5. Structured evaluation with MCA and feedback

For the purposes of this work, we considered that "scoping" involves identification of the problem matter to be addressed in the planning exercise (1). This is followed by an "operationalisation" process, where policy and practical constrains are taken into consideration when defining and evaluating management alternatives (2-5).

Participating stakeholders were representatives from fish producer organisations, fisheries associations and environmental Non-Governmental Organisations (NGOs). Most stakeholders were participants of the North Western Advisory Council (NWWAC), which has a formal role in providing advice on issues related to the Common Fisheries Policy in the North Western regional sea area, which includes the case study area. The NWWAC was a partner in the MareFrame project and facilitated dissemination and discussion of the case study development. The NWWAC also invited its participants to the case study meetings, which included three workshops and several web-based meetings. In line with CFP requirements (EC 2013), 60% of the seats of the NWWAC are allotted to representatives of the fisheries sector and 40% to representatives of the other interest groups. While a wide range of stakeholders were invited to contribute, fishing industry perspectives were nevertheless much more strongly represented than other perspectives in the case study meetings.

2.1 The case study

The important commercial fisheries of the west of Scotland case study area (ICES Division VIa, hereafter referred to as VIa; see Fig. 1 for an overview of the area) include: prawn (*Nephrops norvegicus*, hereafter referred to as Nephrops); the gadoids cod (*Gadus morhua*), whiting (*Merlangius merlangus*), haddock (*Melanogrammus aeglefinus*), hake (*Merluccius merluccius*), and saithe (*Pollachius virens*); and anglerfish (mainly *Lophius piscatorius*).

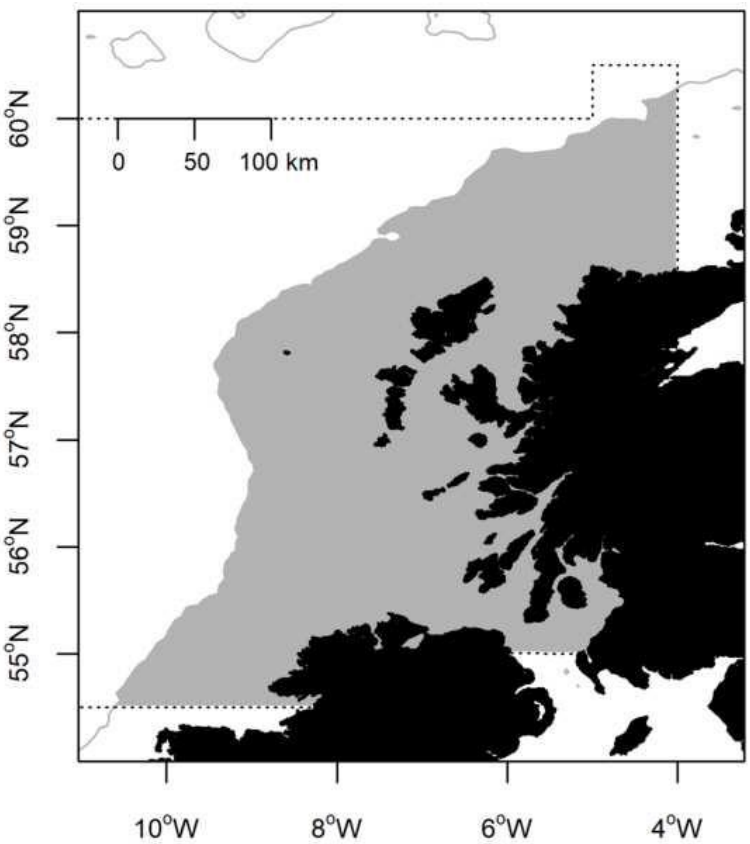

Fig. 1. Map of the west of Scotland case study area showing the model extent shaded in grey. The dotted outline marks the outline of ICES division VIa. The shelf area within division VIa to a depth of 200m was modelled.

UK (Scotland), Ireland and France are the main participants in these fisheries, which are conducted using otter trawlers (ICES, 2012). Trawlers may target a particular species assemblage in particular areas, but invariably catch a mixture of species. The main target fisheries in VIa include an inshore fishery targeting Nephrops (with by-catches of gadoids), a shelf fishery targeting gadoids, and a fishery on the shelf edge, with saithe, anglerfish and hake as important species.

While the fishing mortality (*F*) for shellfish, demersal, and pelagic fish stocks has reduced since the late 1990s in the wider Celtic Sea area (ICES, 2016a), a main problem faced in the demersal fisheries in VIa is that the cod and whiting stocks are depleted as the spawning stocks biomass (SSB) of these stocks have remained close to all-time low levels since the early 2000s (ICES, 2017). *F* for the cod stock remains above F_{MSY} despite an amended recovery plan introduced in 2012 (EC, 2012), which among other things determines Total Allowable Catches (TACs), limits effort, and seeks to incentivize cod avoidance. A voluntary cod avoidance scheme (Holmes et al., 2011) did not achieve intended *F* reductions (Kraak et al., 2013). Since 2012, the TAC for cod has been zero but 1.5% bycatch of live weight of cod is permitted. The catch limits apply to landings, and do not constrain catches as about 60% of the cod catch was on average discarded between 2014 and 2016 (ICES, 2017). As reformed in 2014, the CFP includes an obligation to land all catches of TAC regulated species (EC, 2013). With the landing obligation, cod and whiting stocks could become “choke species” (Baudron and Fernandes, 2015), prompting a premature closure of fisheries for other species. Predation by grey seals (*Halichoerus grypus*) may impede cod recovery, in particular if the seals increasingly target cod individuals when the abundance of cod is low (Cook et al., 2015, Cook and Trijoulet, 2016). The grey seal population is estimated to have more than doubled between 1985 and 2005 but has stabilised since then (SCOS, 2015).

2.2 Estimation of scenario impacts

Scenario impacts were estimated with an ecosystem model and a sub-model to estimate economic indicators. The ecosystem model used was an Ecopath with Ecosim (EwE) (Christensen and Walters 2004; Colléter et al., 2015; Heymans et al., 2016). EwE is a foodweb ecosystem model encompassing the whole trophic food chain from plankton to apex predators (e.g., mammals and seabirds). Groups (i.e., single species or groups of species) are modelled as biomass pools without length or age structure. The use of EwE in a fisheries management context instead of other ecosystem or multispecies models available has both advantages and drawbacks (Christensen and Walters 2004; Heymans et al., 2016). The lack of a length or age structure is a main drawback, which prevents modelling of the impact of alternative selectivities and of issues related to undersized discards. A main advantage is that the model generates insights on the structure and health of the whole ecosystem, which cannot be provided by multispecies models where fewer species and trophic levels are represented in greater details. EwE therefore offers the possibility to calculate ecosystem indicators where the whole foodweb is taken into account (e.g., biodiversity, foodweb evenness, etc.). The literature contains several examples where EwE was successfully applied to investigate fishing management strategies in complex multispecies system (e.g., Stäbler et al., 2016). Appendix A provides details for the EwE model applied to the case study area.

We used revenue and profit as indicators to assess the economic performance of the fishery in each scenario. For each fleet, revenues over the simulation period (2014-2033) were estimated as the landings (Kg) multiplied by the first sale price (£/Kg). We obtained price values from

2008 to 2014 from the Scientific, Technical and Economic Committee for Fisheries of the European Commission (STECF) and used the median prices for the study (Appendix B). Profits for each fleet over the simulation period were calculated as revenues minus costs. To estimate costs over the 2014-2033 period, costs coefficients were calculated using historical data from 2008 to 2014 to relate costs to fishing mortality following Quaas et al. (2012):

(1)

$$Cost\ coefficient_{species} = {}^{Cost_{demersal\ trawl,species}}\!\big/_{Fishing\ mortality_{species}}$$

The resulting costs coefficients are presented in Appendix C. Profits over the simulation period were then calculated as follows using these cost coefficients together with the landings returned by the model:

(2)

$$Profit_{species} = \left(landings_{species} * price_{species}\right) - (cost\ coefficient_{species} * Fishing\ mortality_{species})$$

2.3 Multi-criteria analysis

MCA (Janssen, 2001; Kowalski et al., 2009, Sheppard and Meitner, 2005) was used to support a structured evaluation of alternative management scenarios. MCA software with functionality similar to that described by Mustajoki et al. (2004) was developed within the MareFrame project and is freely available along with the specific MCA model we report on.[2]

[2] The specific MCA model can be assessed and interacted with at the following site: https://mareframe.github.io/dsf/dev/MCA2/DST.html?model=scotland_weighted (accessed 18.06.18). Other generic and specific decision support tools are available at associated webpages.

A main outcome of MCA is a summary score for each scenario, ranking their relative performance. The robustness of the ranking can be explored by a (one-way) sensitivity analysis, by which one parameter is varied at the time. The sensitivity analysis allows for a graphical evaluation of the impact of estimation uncertainty for the indicator values and of changes in the decision weights attributed to sub-objectives and indicators (Mustajoki et al. 2004). The latter is important since it may be difficult to set the decision weights.

2.4 Scope, objectives and indicators

The problem scope for the case study was defined in a workshop with stakeholders held in May 2014 to explore the potential for recovery of the cod and whiting stocks, and to investigate the impact of seal predation. Cod and whiting stocks traditionally have a high economic and cultural significance in Scotland, and the risk of these stocks becoming “choke species” amplifies their importance. Further, the case study identified an approach for Maximum Economic Yield (MEY) for the fisheries concerned. The overall goal of the proposed management alternative was identified as: “achieving sustainable and viable fisheries”.

To be of relevance, a proposal developed by stakeholders must demonstrate consistency with established policy objectives. The CFP and the MSFD are focal for EBFM (Ramírez-Monsalve et al, 2016a) in VIa. In addition, the fisheries and the marine environment in VIa come under the Habitats Directive (EC, 1992), the Birds Directive (EC, 2009), and the Water Directive (EC, 2000).

A key requirement of the CFP is to restore the Spawning Stock Biomass (SSB) of commercial fish stocks to levels consistent with Maximum Sustainable Yield (MSY) by 2020 and/or to maintain them at such levels. The MSFD requires that indicators and thresholds are defined to represent Good Environmental Status (GES) in relation to 11 descriptors. Indicators and thresholds are currently most advanced with respect to descriptor 3, which largely may be seen to represent the CFP requirements of having healthy commercial fish stocks. Three other descriptors were judged to be of potential relevance for this case study. These are descriptor 1 (biodiversity), 4 (integrity of foodwebs) and 6 (integrity of seafloor habitats). Descriptor 6 was not addressed because the model framework was not set up to address spatial aspects. In addition to biological and environmental objectives, the CFP and the MSFD seek to achieve social and economic sustainability for the use of marine resources, notably fisheries, but no specific objectives have been defined for fisheries in VIa for these components.

The assessment and comparison of the management scenarios were carried out using three categories of indicators (i) biomass of key demersal stocks; (ii) ecosystem indicators relevant to assess GES, (iii) economic indicators to assess economic viability and profitability.

The key demersal stocks included cod, whiting, haddock, hake, saithe, and Nephrops. The applied EwE model returns SSB for the three former stocks and Total Stock Biomass (TSB) for Nephrops. The model also returns TSB of a group of similar species, of which hake comprises >80% (see Baudron and Fernandes, 2015), and which henceforth will be regarded as hake for the purposes of this work. Similarly, the model returns TSB of closely related species of which saithe comprises >95% (Bailey et al., 2011), and which here will be regarded as saithe.

Four indicators were chosen as relevant to assess GES: biomass of seals, biomass of seabirds, biomass of prey fish, and an index of “balanced evenness” (see Appendices D and E for details). These indicators were chosen because they could be computed from the biomasses returned by the model and because they are relevant to assess the identified scenarios. The biomass of seals was relevant since we tested a scenario involving a seal cull. As top predators, the biomass of seals depends on what the ecosystem food chain can support.

Similarly, the biomass of seabirds depends on and reflects ecosystem health. Since most seabirds feed on small pelagic fish at a lower trophic level than the species that constitute the prey for seals, seabird biomass offers a different perspective on the food web. The group prey fish was established to encompass small forage-type fish, which support the biomass of piscivorous species of many of which are targeted commercially but also constitutes the diet of many top and intermediate predators. Lastly, the balance evenness index measures the biodiversity of the food web (see Annex D for details). Unlike traditional diversity indices such as the Shannon index, the balance evenness index accounts for the diversity within each trophic level. The main objectives and chosen indicators are presented in Table 1.

Table 1. Objectives defined for the case study (left column); their basis (middle column) and the indicators in the MCA (right column). Details of the MCA indicators are provided in Appendix D.

Objective	**Basis**	**MCA indicators**
To recover the cod and whiting stocks	CFP requirement and stakeholder objective	Cod SSB Whiting SSB
Healthy commercial fish stocks	CFP and MSFD Descriptor 3 Stakeholder objective	Haddock SSB Saithe TSB Hake TSB Nephrops TSB
Maintain foodweb integrity	MSFD Descriptor 4	Balanced evenness Prey fish species TSB Seabird biomass Seal biomass

Economic sustainability	CFP and stakeholder objective	Catch value by fleet (pelagic, demersal and Nephrops) Profit proxy by fleet

2.5 Management scenarios

Generic management scenarios were identified in cooperation with stakeholders to represent candidate approaches to achieve identified objectives to the extent possible. Two scenarios were defined to represent baselines for comparison (Table 2).

Table 2. The explored management scenarios (short name used in MCA in parenthesis), their rationale, and model approach. The scenario marked with (*) involved seal culling and was only included to assess the effect of seal predation on cod and whiting recovery.

Strategy type	Scenario and rationale	Modelling approach
Reference points (baseline)	*Fishing at Maximum Sustainable Yield (F_{MSY})* Baseline for comparing alternatives. Reflects MSY as a main policy goal of CFP. This strategy does not consider aspects of landing obligations (notably choke species problems) and can therefore not be fully implemented in practice due to mixed fisheries interactions.	Set F at (single species) F_{MSY} or best available F_{MSY} proxy for all species.

Economic optimisation	*Fishing at Mixed Maximum Economic Yield (MixMEY)* There is a conflict between the requirements of the Landing Obligation, MSY (partly due to the choke species issues), and the objective of economic sustainability. This conflict is pronounced in a situation of mixed fisheries, where catches of depleted stocks cannot be fully avoided in fisheries for other stocks (Ulrich et al., 2017). F-ranges provide flexibility, increasing the scope for MEY candidates compatible with policy requirements: • Optimize MEY across stocks within the flexibility provided by *F*-ranges. • Relax MSY constraints for Cod and Whiting; MSY constraints for other TAC stocks. • Maintain incentives to avoid cod and whiting catches. • Maintain Nephrops *F* at current level as increasing it would be difficult and would increase risks of catching juvenile cod and whiting).	Identify MEY candidate within *F*-ranges for haddock, saithe, anglerfish and hake. Keep *F* for cod and whiting as low as practically possible without reducing *Fs* for fisheries with these species as bycatch. Reduce *F* for haddock consistent with effort to avoid cod and whiting bycatch. Explore F_{MSY} ranges for other demersal target species. Keep *F* for Nephrops at 2013 level.
Spatial aspects of mixed fisheries	*Spatial Management of the Mixed Fishery (Spatial F)* Promote cod and whiting recovery, giving consideration to the spatial distributions of catch species. This assumes separability between mixed demersal fisheries mainly located on the shelf (cod, haddock and whiting) from those mainly located on the shelf edge (hake, saithe and anglerfish), and that	Explore *F*-ranges while restricting the *F* values applied for each of the following two groups to be within +/-0.05 of each other: (i) cod, haddock and whiting

	different *F*s can therefore be applied to these two groups (shelf and shelf edge). See Annex G for information on the distribution of these species.	(located on the shelf) and (ii) hake, saithe and anglerfish (located on the shelf).
Predator control	*Gadoid recovery* Promote cod and whiting recovery by fishing saithe at upper *F*-range (F=0.42) as saithe has been found to be a significant predator of juvenile cod and whiting. Closure of targeted fisheries for cod and whiting while accepting present level of bycatch simulated by applying F=0.05 (residual *F* currently observed for adult whiting which is no longer actively targeted).	Apply F_{MSY} values for all species except cod, whiting and saithe for which various levels of F are tested.
Predator control	*Gadoid recovery and seal cull** As the previous except for a simulation of an annual cull of grey seals.	As above except *F* for grey seals set at 0.05
Baseline	*Status quo F (SQ)* Alternative baseline: what would happen if present fishing mortalities continue?	*F* at F_{2013} for all groups

2.6 Estimation of scenario impacts

We assessed the performance of the identified management scenarios with the EwE model, applying *F*s corresponding to the scenarios to drive forward simulations for a 30 year period from 2014 to 2033. For the Status quo scenario, we kept *Fs* at their 2013 level. For the F_{MSY} scenario, we applied single stock F_{MSY} reference points defined by ICES from 2014 and onwards. For the other scenarios, we explored ranges of possible F values for each stock. Following a request from the European commission, ICES now provides *F* ranges in addition to the traditional single stock F_{MSY} values. The F_{MSY} ranges are limited by upper and lower boundaries, which are defined such that expected sustainable yield is no more than 5% lower than MSY (ICES, 2016b). The *F*-ranges applied have not been defined for all the stocks relevant to the modelled scenarios, and in some cases we used the best available proxy (e.g. *F*-ranges defined for the same stock in an adjacent area). Appendix F provides details for the F_{MSY} ranges used to model the scenarios. For each stock, we explored the *F* ranges by simulating the upper and lower *F* boundaries and *F* values between these boundaries with 0.05 steps. For each management scenario other than Status quo and F_{MSY}, we simulated each possible combination of *F*s between the stocks, with one simulation corresponding to a single combination. We used the Multisim plugin of the EwE software to perform the simulations.

3. Results: Structured scenario evaluation with MCA

An essential step in the process of using MCA is to develop a hierarchical structure of the problem context, which in turn will enable a systematic evaluation of the identified scenarios. We defined the value tree (Fig. 2) in cooperation with stakeholders to increase the relevance of the MCA.

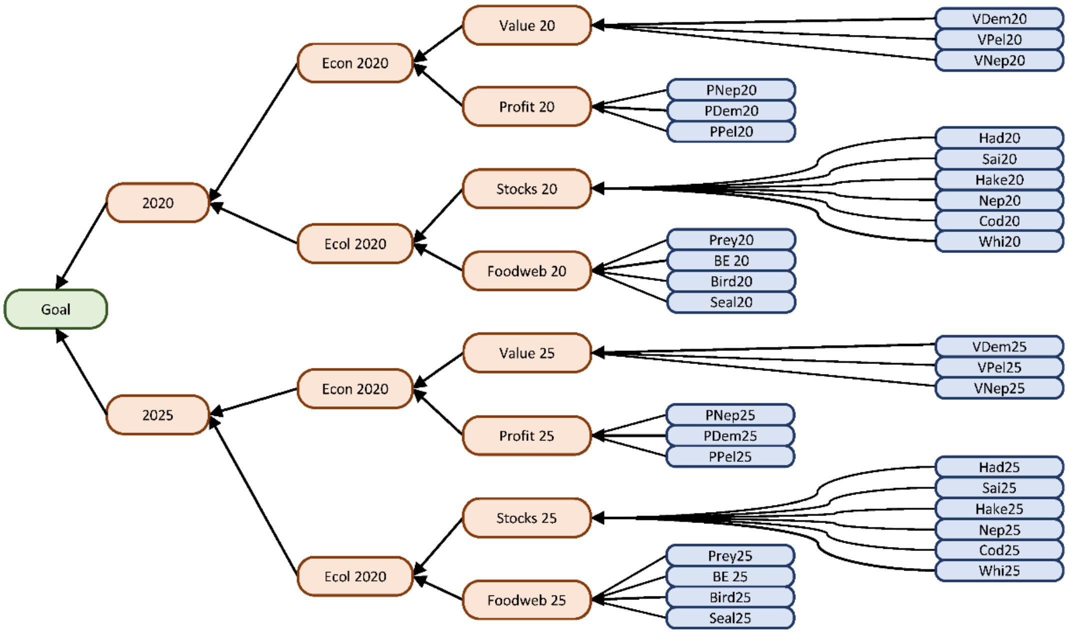

Fig. 2. Structure of the MCA (value tree) used to evaluate the alternatives. The evaluation is based on model estimates for two time points (2020 and 2025) with regard to the 16 indicators presented in table 1.

The value tree is a hierarchical structure and includes two main branches to support deliberations relating to temporal trade-offs.. While the EwE model indicators for each year between 2014 and 2033, we only used estimates of indicators status from 2020 and 2025 in the MCA, calculated as three year averages with the indicated time point at the middle of the interval. The years 2020 and 2025 were chosen by stakeholders to respectively represent short and medium term outcomes. The two branches are symmetrically divided further into sub-branches representing ecological and economic concerns. The economic sub-branch is divided into a branch for profitability and a branch for catch values, and each of the latter is connected to indicator for each of three fleets. The ecological branch is sub-divided to enable a trade-off between commercial stock sustainability and other ecosystem sustainability aspects (termed “foodweb”). The value tree includes separate nodes for the six key commercial stocks. The non-commercial aspects are evaluated through four nodes: availability of important prey fish species ("preyfish"), seals and seabirds and "balanced evenness".

We selected outputs from the scenario modelling with EwE to calculate the MCA indicators (see Appendix D for details). The input data for the MCA (i.e. consequence table) is shown in Appendix H.

Value functions

The value functions describe the relative utility of a given indicator within the available range between the lowest and highest indicator values across the scenarios. The utility values range between 0 and 1, and the shape of the value function defines how utility relates to the indicator value. The utility functions were defined by the stakeholders (the economic indicators) or by the authors (ecological indicators). The definition of value functions and decisions weights (see below) is subjective, but was based on reasoning in order for the MCA to be meaningful. We are not aware of any earlier study that has used MCA in a way that creates a relevant precedence for defining the value functions, which we set as follows:

Economic indicators

The value functions for the economic indicators (catch value and profitability by fleet) were set to increase linearly from the minimum value for the indicator across the scenarios (assigned utility = 0) to the maximum value (utility = 1). This implies that any increase in revenue is equally important within the available range of options.

Stock sustainability

The value functions for the SSB of cod, whiting and haddock were defined in relation to ICES reference points for these stocks, so that the utility SSB would be zero at SSB = 0, increase linearly to 0.5 at B_{lim} and linearly from that point until reaching 1 at B_{pa}, and with no change in utility with SSB values higher than B_{pa} (Fig. 3). For haddock, cod, whiting, saithe and hake ICES has proposed to use B_{pa} as a B_{MSY} trigger point, essentially rendering the B_{pa} a target point for MSY. Since 2013, ICES has not provided separate assessments of haddock in VIa as it is now included in a larger stock area. To define the utility function for haddock SSB we used ICES previous reference points, specific for haddock in VIa (ICES 2013).

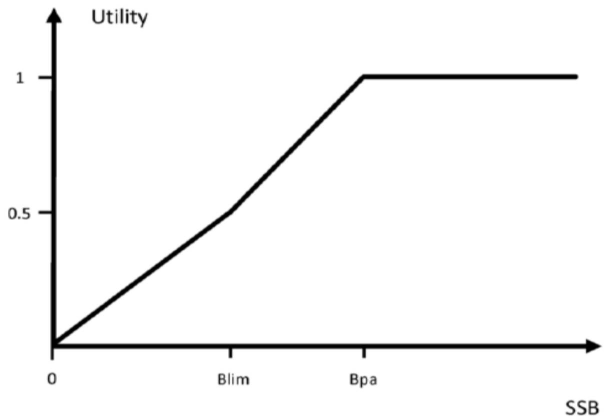

Fig. 3. Utility functions defined for SSB.

We used the average ratio between ICES' SSB estimates for saithe and the TSB estimates for saithe from the EwE model for the years 2004-2013 to rescale ICES reference points. Subsequently we defined the utility functions as described in Fig. 3. The same approach was used for hake. ICES does not provide reference points for SSB for Nephrops in the functional units in area VIa. However, differences between TSB estimates for Nephrops across the scenarios are small. ICES assessments for the years around the year 2013 and later indicate that these stocks are significantly above an MSY level. Accordingly, we set the utility for Nephrops at 1 for all scenarios, assuming that they were at or above levels compatible with ICES notion of B_{pa}.

Foodweb indicators

We set an increasing linear value function for the indicator “Preyfish” to reflect the importance of having prey fish species available for species on higher trophic levels. An increasing linear function was also set for ‘balanced evenness’ and for the biomass of seabirds, reflecting that “more is better” for these indicators within the range of estimated outcomes. The stakeholders defined a dome shaped value for the seal population, preferring that the population does not decline below the current level, and perceiving that a considerably larger seal population would not be desirable as it predates on cod and whiting.

Decision weights

The decision weights were largely set by the stakeholders that participated in the decision support workshop (Table 3), but the time available proved insufficient for thinking carefully through the issues involved. In some instances, the decision weights were therefore redefined by the authors. The participants in the workshop found it difficult to agree on decision weights, reflecting differences in individual preferences. For the purposes of the case study, we encouraged consensus development, bearing in mind that the influence of the Advisory Councils depends on its ability to generate consensus advice (Hatchard and Gray, 2014).

Table 3. Relative decisions weights (presented as ratios) with regard to trade-offs between concerns structured according to the value tree in Fig. 2.

Trade-off	**Relative decision weights**	**Rationales and comments**
Short term (2020) vs. medium term (2025)	3:2	Reflecting the need of getting the industry through a period expected to be particularly economically difficult due to the onset of the landing obligation.

Economic vs. ecological concerns	3:2	Compromise consistent with the statutory composition of the AC regarding industry vs non-industry representatives.
Profit vs. catch value	1:1	At the time of the workshop, an indicator of profitability was not available
Demersal vs. Pelagic vs. Nephrops fleets regarding profit and catch value	2:1:1	In the workshop, stakeholders set the decision weights for the fleets as equal. However, it can be argued that the demersal fleet should be given a higher priority than the pelagic and Nephrops fleets as it is subjected to much higher variability regarding profit and catch value across the scenarios, reflecting a higher sensitivity to economic performance (Appendix J).
Stock sustainability vs. foodweb	3:2	Above argument relating the statutory composition of the AC
Cod vs. whiting vs haddock vs. hake, vs. saithe vs. Nephrops	2:2:1:1:1:1	In a workshop, the stakeholders set decision weights for the six commercial stocks to reflect differences in their economic significance. However, as this branch concerns stock sustainability, while economic concerns are address in a separate branch, the authors decided to redefine these decision weights for the purpose of this analysis. The weights set so that stocks with SSB below B_{lim} in the base year 2013 (cod and whiting) were given double weight compared to the other stocks, which were judged to be above B_{pa}.

Evaluation outcomes

Fig. 4 shows the performance of the management scenarios as summary scores. The highest score indicates the best performing scenario with respect to the identified objectives, given the decision weights and utility functions presented above. Details of how each indicator contributed to the overall performance of each scenario are provided in Appendix I.

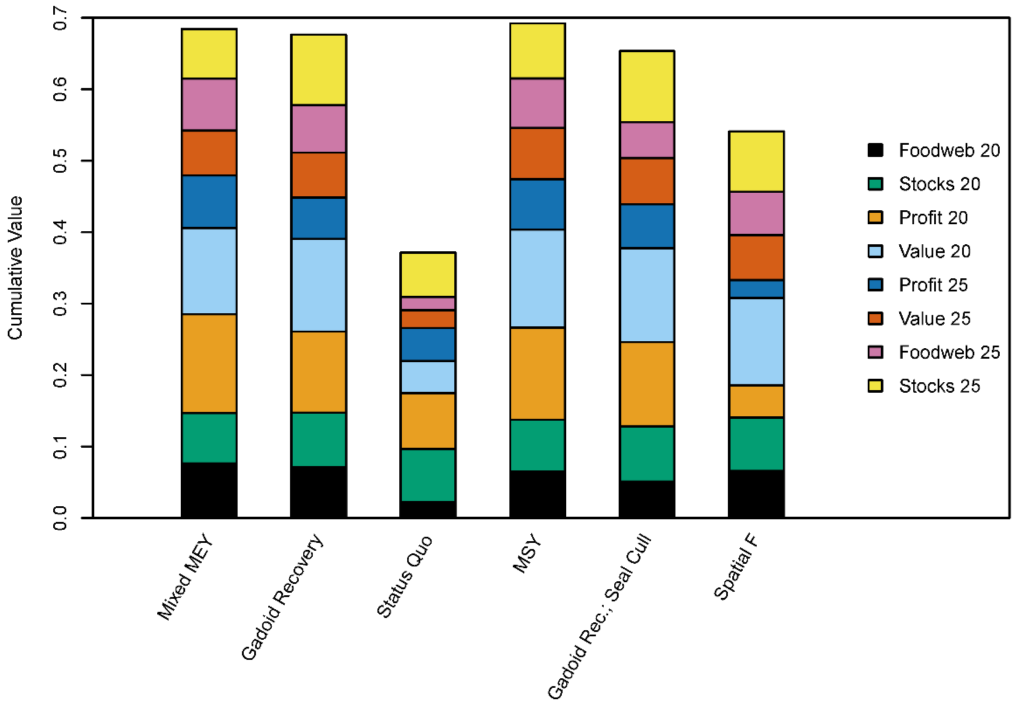

Fig. 4. The figure shows the aggregated score (sum of utility contributions from all indicators) for the identified management alternatives, given the decision weights defined in table 3.

The evaluation indicates that "MSY" would achieve the highest aggregated evaluation score (0.692), closely followed by "Mixed MEY" (0.684), "Gadoid Recovery" (0.677), "Gadoid Recovery with seal cull" (0.653) and then by "Spatial *F* (0.541)" and "Status Quo *F*" (0.372).

The baseline scenario "Status Quo *F*" clearly performed poorly compared to the other scenarios, indicating a potential for improvements through alternative strategies. While "MSY" is consistent with main objectives of the CFP, it is not possible to fully implement in practice due to mixed fisheries interactions and ensuing choke species issues. This also applies to the two "Gadiod recovery" scenarios as the modelling of these relied on F_{MSY} for most species. "Mixed MEY" and "Spatial *F*" were set up and constrained in order to take mixed fisheries issues into account. These scenarios are also subjected to implementation error as they do not represent detailed solutions to the mixed fisheries and choke species issues, and we recognize that the chosen modelling framework is not always suitable for modelling these aspects in detail. However, it seems reasonable that the implementation error was less for Mixed MEY" and "Spatial *F*" than for the scenarios based on F_{MSY}. This suggests that the performance of "MSY" and the gadoid recovery scenarios is inflated compared to "Mixed MEY" and "Spatial *F*". Given that "MSY" and "Mixed MEY" received very similar scores, this indicates that "Mixed MEY" in practice performed best overall. "Mixed MEY" did not perform worse than the other scenarios for any indicator (Appendix I). Although they achieve similar overall scores, there were significant differences between the performance of "Mixed MEY" and "Gadoid Recovery". The former did better regarding profits in the short and medium term, while the latter performed better regarding stocks, in particular in the long term. In turn, "Spatial F" lost out because it performed poorly regarding profitability, in particular for the demersal fleet. This was expected as the scenario involved *F* reductions for stocks caught together with cod and whiting in order to promote recovery of the latter two stocks.

"Gadoid Recovery with seal cull" was only included to explore the impact of grey seal predation as it did not represent an acceptable management scenario in the UK. Predation of grey seal was found to affect the recovery of cod and whiting, although not strongly when compared to the impact of fishing and/or other predator interactions.

No scenario was estimated to lead to rapid recovery of cod and whiting, but the outcomes indicated that recovery of these stocks was possible in the long term through a combination of measures. "Spatial *F*" displayed the greatest cod recovery in the short term and lead to full recovery above B_{pa} as well as the highest cod SSB level across all scenarios in the medium term. Apart from "Spatial *F*", only "Gadoid Recovery" (and "MSY") increased the cod SSB to a level near B_{pa}. The gadiod recovery scenarios lead to the highest increases in whiting SSB, but no scenario involved recovery to B_{pa} for whiting (Appendix H). This is due to the fact that cod predates heavily on whiting in the area. Hence, recovering cod increases predation pressure on whiting and in turn delays its recovery, despite a reduction in *F*. This example illustrates a type of insights which is not available based on single species models without trophic interactions, reflecting how a foodweb model may serve to complement the information basis for EBFM.

Sensitivity analysis

In accordance with the reasoning provided above, and in the interests of simplification, “MSY” was omitted from the sensitivity analysis. The sensitivity analysis indicated that quite small changes in the weights assigned for the temporal aspect changed the ranking of “Mixed MEY” and “Gadoid recovery”, i.e. the two best performing scenarios following “MSY”. The decision weights reflected a slight priority given to short term considerations, and this resulted in an overall preference for “Mixed MEY”. The prioritisation of short term considerations reflects a high discount rate consistent with what has been estimated for other fisheries (Asche, 2001). However, “Gadoid Recovery” would obtain the highest score if stakeholders had assigned equal priority to short and medium term concerns. The other scenarios did not rank highest regardless of the weights assigned for the short and medium term. The ranking of scenarios was, therefore, robust regarding changes in the preference between the ecological and economic objectives in 2020.

The sustainability of cod and whiting stocks were assigned a higher weight than the stocks of haddock, saithe, hake and Nephrops stocks. “Mixed MEY” dominated irrespective of the weight assigned to the cod stock. The ranking of scenarios was robust to stock assessment uncertainty. “Mixed MEY” had the highest overall value (although with a small margin) even if the stock biomass estimate was significantly biased for any of the stocks.

Consequently, and, as explained apart from “MSY”, the sensitivity analysis for all decision weights and predictions indicated that either “Mixed MEY” or “Gadoid Recovery” performed best overall. The preference for these strategies was robust for a wide range of changes in weights assigned to the many sub-objectives and to biases in the predictions for fish stock biomass, profits, the value of landings, and bird and seal abundances.

5. Discussion and conclusions

Identification of scope, objectives and indicators

The scope of the case study was defined in a workshop held early in the project, but it proved necessary to restrict the problem matter later. Stakeholders expressed increasing interest in investigating issues relating to the landing obligation. The researchers perceived that this would risk diverting focus from the project goal to address EBFM, and that the modelling framework chosen would be inappropriate for studying the landing obligation. A compromise was found, and the experience shows the importance of clarifying and managing mutual expectations and needs from start to finish. The limitations with regard to participation of NWWAC members (in particular concerning the representation of other interests than commercial fisheries) underline that outcomes of the case study do not represent a NWWAC position. The case study was explored in terms of a methodology with a potential to support the development and structured evaluation of such a position.

The selection of indicators was challenging as they had to be relevant for evaluating the defined objectives, they had to be easily understood, and possible to estimate (see e.g. Rochet and Rice, 2005; Jennings, 2005, Link, 2005). We did not identify ecosystem indicators with all desired properties and included some improvised indicators. In addition, our approach to estimate the economic indicators, revenue and profit, necessary to compare the performance of management scenarios was simplistic.

Identification of alternative management scenarios

The formulation of operational alternatives was a challenge, in part due to the restrictions regarding what could be estimated by the chosen model. The notion of *F*-ranges presented itself as an opportunity at a late stage of the project, reflecting benefits of an iterative approach to scenario formulation.

Estimation of scenario impacts

While the EwE model was well suited for exploring the impact of predation by seal and piscivorous fish on cod and whiting recovery, it was less suited to investigate the short term impact of the landing obligation. As is often the case for complex ecosystem models, the EwE model does not in itself provide for a formal uncertainty analysis. Models of intermediate complexity such as GADGET provide uncertainty analysis of the estimates for the fish species it considers, but then they include fewer components. In our case study, the lack of uncertainty estimates is to some extent compensated for by the sensitivity analysis in the MCA.

Some stakeholders were sceptical to specific scenario projections. For instance, stakeholders argued that it would not be practically feasible to increase *F* for Nephrops significantly as entailed in some scenarios in the first version of the MCA. This prompted a change of scenario formulations for Nephrops, reflecting the importance of an iterative process and of utilising stakeholders' local ecological knowledge to improve the reliability of outcomes.

Moreover, many stakeholders seemed somewhat sceptical to the use of a broad ecosystem model, questioning the reliability of its detailed outputs. Such scepticism is sound, and stimulates critical examination of the outputs. Yet, model simulations of complex issues on a medium time scale cannot generate predictions with the level of certainty that characterizes traditional single stock projections. As suggested by Degnbol (2005), an ecosystem approach will require that expectations of predictability are lowered, which in turn necessitates change in the way model outcomes are perceived to support planning. Stakeholders and researchers will need to embrace such changes, and the co-creation approach represents one way to stimulate learning, dialogue and creativity with regard to making use of models with high uncertainty and soft predictability. We do not consider this a barrier to future use of ecosystem models as most stakeholders, especially those with a background in fisheries, experience variations in the ecosystem and hence readily understand that model estimates are uncertain.

Structured evaluation with MCA and feedback

The MCA methodology complements the co-creation approach because its main framing elements (e.g. scope, criteria, objectives, problem structure and alternatives) are explicit inputs that can be “opened up” for deliberation (Stirling, 2006). If the role of stakeholders is limited to set decision weights, the MCA would at once be “closing down” a wider policy discourse (Saarikoski et al. 2013). To promote relevance and buy-in, the co-creation approach fosters involvement of stakeholders in a sequential process of “closing” each of the framing elements in order to establish and use the MCA. The co-creation approach, however, does not invite unconstrained deliberation as it insists on policy relevance. Stakeholders were well aware of and accept the policies that apply to the fisheries in question, and thereby in the position to set relevant objectives to be included in the MCA.

The definition of the value tree in MCA lent itself well to a participatory approach, and it was straightforward to reach agreement on a suitable structure. In contrast, stakeholders did not perceive the setting of decision weights and value functions to be intuitive. In testing the MCA approach, we encouraged the stakeholders to reach consensus, having in mind that the NWWAC generally seeks to achieve consensus in order increase the legitimacy and impact of its advice. However, the participants in the workshop stated a preference for an approach based on individual MCAs. It should also be noted that stakeholders may be reluctant to clarify their priorities in public, as this may compromise subsequent negotiation positions (Pope et al., 2019). As long as they build on the same value tree and set of scenarios, individual MCAs can be aggregated into a common result (Mustajoki, 2004). MCAs can also be used by decision makers to provide information on how different stakeholder groups evaluate the issues at hand.

The setting of decision weights is subjective, and appeared to be perceived as abstract and somewhat uncomfortable. Nevertheless, such priorities are also made implicitly when decisions are made unaided by decision support methods. An advantage of MCA is that it requires careful deliberation about priorities in relation to specific trade-offs. The explication of priorities stimulates the articulation of rationales, enhances transparency, and allows for repeatability.

A generic strategy that aims to optimize economic yield within the applicable F_{MSY} ranges was found to represent a promising approach as it makes it possible to take predator-prey relationships (and potentially also harvest technical interactions) into account. Such considerations will require that the main trade-offs are presented, considered and evaluated, for instance with MCA. However, the specific outcomes of this work cannot be taken to represent the views of the stakeholders with which we have cooperated as time and resources did not permit us to evaluate the final versions of the scenarios presented here. The evaluation and the sensitivity analysis suggested that either “Mixed MEY” or ”Gadoid recovery” performed best overall. These two strategies are performing well for a wide range of changes in decision weights and estimates of indicator status. Further efforts to validate the predictions for these two strategies are nevertheless warranted. Also, it would be worthwhile to examine the trade-offs these two management strategies will imply for different stakeholder groupings in more detail.

The reformed CFP has established a framework for regionalized management. A proposal for a multiannual plan for demersal species in western waters is currently considered for adoptation by the Council and the European parliament (EC, 2018). As part of the process of developing the proposal, a public hearing was conducted by the Commission to gather inputs on the plan (DGMARE, 2015). The NWWAC expressed dissatisfaction with the approach of this hearing, finding it insufficiently detailed. If appropriately extended, validated and improved, the tools and processes developed and tested in this case study could potentially provide support for advisory councils and/or groups of member states to explore and document their position on generic management options. The notion of F_{MSY} ranges represents a key element of the proposed multiannual plan (EC, 2018). If adopted, the plan will establish management flexibility to address mixed fisheries issues in the way suggested with the “Mixed MEY” and “Spatial *F*” scenarios.

The fact that the UK has decided to leave the EU, however, raises uncertainty about the management framework that will apply to demersal fisheries off the west coast of Scotland.

Scoping and re-scoping problems and potential solutions is an essential aspect of EBFM (Dickey-Collas, 2014). Combining a co-creation method with scenario based planning, using MCA and ecosystem model simulations, the approach presented appears to have a potential for supporting such a scoping process. We are not aware of published studies that have used MCA in the evaluation of management scenarios for EBFM strategies (but see other articles in this issue for a similar approach). Compatible with any model generating relevant scenario information, the MCA is flexible and incurs low costs. In cooperation with stakeholders, we have shown possible ways to reason about value trees, utility functions and decision weights, but the application of MCA in the domain of EBFM largely remains uncharted land and requires further development and tests in order to be consolidated and used.

Conclusions

MCA and ecosystem model simulations can be combined to support a participatory approach to scenario based planning in EBFM. The approach does not provide actionable management advice, but stimulates learning and creates an opportunity for stakeholders to search for strategic and policy relevant solutions and to position themselves in an EBFM context.

Expectations regarding model precision have to be adjusted when the scope of the management focus is expanded from a single species to complex ecosystems. This should be approached in a way that supports communication and understanding regarding uncertainty in the planning processes.

The MCA facilitated a structured, transparent and repeatable evaluation of trade-offs, based on explicit priorities, but it was difficult for stakeholders to reach agreements on how set utility functions and decision weights. This requires careful deliberation and time and may be complicated due to a reluctance to reveal negotiation positions (Pope et al., 2019). The application of MCA in the domain of EBFM will require consolidation in order to be used in practice.

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Supplementary materials

Contents

Appendix A: Details of the EwE model

An EwE model for the west of Scotland was first established by Haggan & Pitcher (2005) and subsequently updated by Bailey et al. (2011), Alexander et al. (2015), and Serpetti et al. (2017)). The latter version is employed in this study. EwE is a mass-balance foodweb model that can include a large number of species or species [functional] groups modelled as biomass pools. It is useful to investigate trophic flows, quantify prey-predators interactions, and assess ecosystem health due to the large number of trophic levels modelled (i.e. from producers to top predators). EwE comprises two components: Ecopath, a mass-balance model accounting for energy transfers in the ecosystem, which depicts a 'snapshot' of the ecosystem in a given year; and Ecosim, the dynamic component that enables temporal simulations based on Ecopath (Walters et al., 1997). Ecopath is defined by two main equations: (i) the first one describes the equality of production terms for each functional group in the ecosystem between the biological production, and the sum of: predation mortality, fisheries catches, biomass accumulation, net migration and other (i.e. unexplained) sources of mortality; (ii) the second equation describes, for each functional group, the energy balance between consumption and the sum of production, respiration and unassimilated food (Christensen and Pauly, 1992; Polovina, 1984). Ecosim uses a time-dependent differential equation based on Ecopath. Ecosim enables temporal dynamic simulations of fisheries by varying the exploitation rates applied to each group (and subsequently biomasses and catches) whilst the Ecopath parameters (e.g. diet composition) remain constant and equal to the start year (i.e. 'snapshot' year modelled in Ecopath).

The specific area that the EwE model corresponds to ~110,000 km2 of the continental shelf of VIa depth delineated by the 200 m depth contour (see Fig. 1 in the main text). The model comprises 41 functional groups which span ~5 trophic levels and include all the major commercial fish and shellfish species, their main prey (i.e. small fish and plankton groups) and predators (large fish, seabirds and mammals), as well as five fishing fleets. The cod, haddock and whiting groups are split between immature (age 0 and 1) and mature (age 2 and above) components (termed stanzas in EwE). The start year of the model on which Ecopath is based was 1985 while the dynamic component Ecosim was calibrated from 1985 to 2013 (see Serpetti et al. (2017) for details).

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Appendix B: Market price for commercial fish and shellfish species

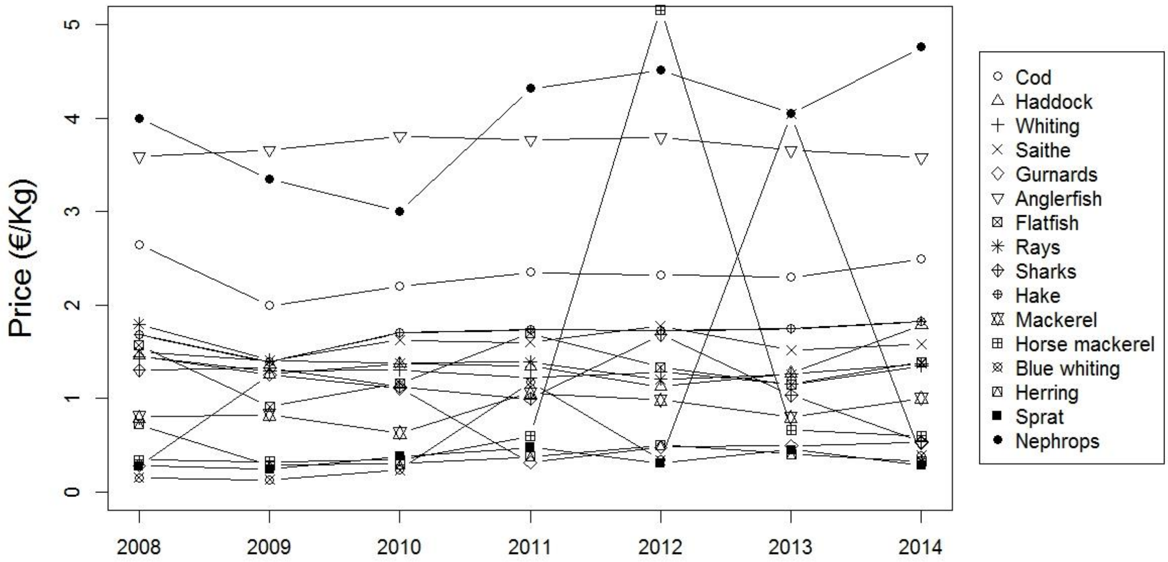

Figure B.1. Market price (€ per kilogram) for commercially caught fish and shellfish species obtained from the STECF database (https://stecf.jrc.ec.europa.eu/).

Appendix C: Cost coefficients

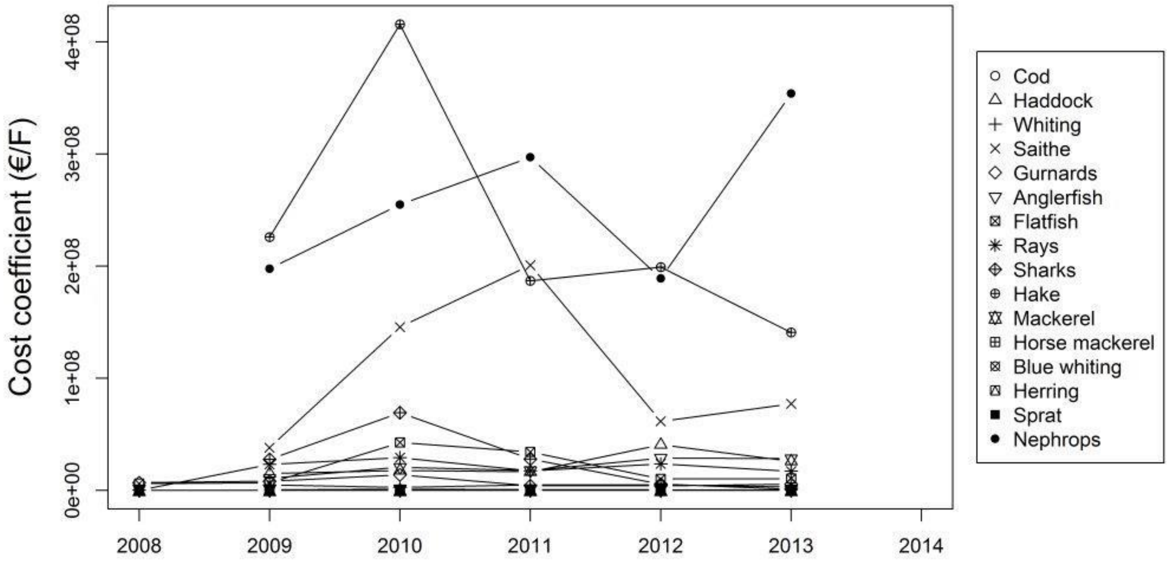

Fig. C.1. Historical time series of cost coefficients (€ as a function of fishing mortality) for commercially caught fish and shellfish species, calculated following the methods from Quaas et al. (2012).

Appendix D: Details of MCA indicators

Table D.1. Table presenting details of MCA indicators.

MCA indicator	Abbreviation in MCA	Definition	Unit
Cod SSB	Cod20; Cod25	SSB of cod in VIa in 2020 and 2025	1000t
Whiting SSB	Whi20; Whi25	SSB of whiting in VIa in 2020 and 2025	1000t
Haddock SSB	Had20; Had25	SSB of haddock in VIa in 2020 and 2025	1000t
Saithe TSB	Sai20; Sai25	TSB of saithe in VIa in 2020 and 2025	1000t
Hake TSB	Hake20; Hake25	TSB of hake in VIa in 2020 and 2025	1000t
Nephrops TSB	Nep20; Nep25	TSB of Nephrops in VIa in 2020 and 2025	1000t
Balanced evenness	BE_20; BE_25	Index described in Annex D.	Number
Prey fish species	Prey20;Prey25	Sum of TSB of the following species/functional groups: Blue whiting (*Micromesistius poutassou*), Norway pout (*Trisopterus esmarkii*), sprat (*Sprattus sprattus*), sandeel (various species), herring (*Clupea harengus*).	1000t 1000t
Seabird biomass	Bird20; Bird25	TSB of seabird species included in the model	1000t
Seal biomass	Seal20; Seal25	TSB of seal species (Grey seal *Halichoerus grypus* and Harbor seal (*Phoca vitulina*).	
Catch value (by fleet)	VPel20, VPel25 VDem20,VDem25 VNep20;VNep25	Catch value for by fleet (pelagic, demersal and nephrops)	Euro
Profit (by fleet)	PPel20,PPel25; PDem20;PDem25 PNep20;PNep25	Profit proxy (by fleet)	Euro

Appendix E: The Balanced Evenness index (Food Web Evenness index)

Calculation of the balanced evenness index (henceforth referred to as the Food Web Evenness index, FEW) is a two-step process. First the expected biomass (B_{ie}) of each species (or functional group, trophospecies, depending on the aggregation level of the model) i is calculated, then an inverted dissimilarity index (Bray-Curtis, *BC*, or Canberra metric, *C*) is used to measure how close the observed biomasses of species are to their expected biomasses.

To calculate expected biomasses, we define a state of 'food web evenness' as decreasing biomasses with increasing trophic levels and equal biomasses within trophic levels. For example, if we assume that biomasses at consecutive integer trophic levels differ by a factor of 10, and total biomass at the second trophic level is B^*, then expected biomass on the third trophic level is $0.1B^*$, and on the fourth trophic level $0.01B^*$. If there are no further trophic levels, then total biomass in the community equals $(1 + 0.1 + 0.01) \cdot B^*$. Biomasses within a trophic level are expected to be equal, thus, if there are four species at trophic level 2, they are all expected to have biomasses equal to $B^*/4$.

We can generalize these relationships as follows: B_{ie} values are calculated based on the total expected biomass at the lowest ('reference') trophic level, B^*, which is estimated as a certain fraction of the observed total biomass in the community Tot_B:

$$B_{ie} = \frac{B^* \cdot \varepsilon^{-(TL_i - TL^*)}}{n_i}, \qquad (1)$$

$$B^* = \frac{Tot_B}{\sum_k \varepsilon^{-(TL_k - TL^*)}} \qquad (2)$$

where $\varepsilon > 1$ is the biomass ratio of consecutive integer trophic levels (10 in the above example). It is the multiplicative inverse of transfer efficiency defined as the ratio of production at consecutive trophic levels. TL_i is the trophic level of i, TL^* is the reference trophic level, n_i is the number of species at the same trophic level as i, Tot_B is total biomass in the community and k is the total number of all (not only integer) trophic levels.

The vector of B_{ie} values can then be compared against observed biomasses (B_{io}) in a community using Bray-Curtis dissimilarity:

$$BC = (\sum_i |B_{ie} - B_{io}|) / \sum_i (B_{ie} + B_{io}) . \qquad (3)$$

The Bray-Curtis dissimilarity index is more suitable to track changes in more abundant species (Krebs, 1999), as it calculates the change in biomass in each group divided by the sum of biomass in the two compared communities. However, for many applications it is more relevant to give equal weight to less abundant higher trophic level species. In these cases the Canberra Metric (Lance and Williams, 1967) measure could be used. This one calculates change in biomass relative to the sum of observed and expected biomass, i.e. relative change compared to group biomass:

$$C = \frac{1}{s} \cdot \sum_i \frac{|B_{ie} - B_{io}|}{B_{ie} + B_{io}}, \qquad (4)$$

where s is the number of species in the community.

Finally, to calculate *FWE* we invert *BC* (FWE_{BC}=1-*BC*) or *C* (FWE_C=1-*C*), so higher index values express higher evenness.

An advantage of the *FWE* index is that it is independent of the total biomass in the system, in the sense that if community A has two times the total biomass of community B, but the biomass fraction of each species in the two communities are the same, *FWE* index values for communities A and B are going to be the same. Thus, *FWE* only tracks relative changes in species biomasses, i.e., in the compositional diversity of the community (it's scale invariant *sensu* Tuomisto, 2012).

It has to be noted that the ‘biomass pyramid’ concept does not hold for the biomass relationships at the very bottom of aquatic foodwebs due to high productivity of phytoplankton and microzooplankton. Thus, for aquatic systems it is sensible to only include multicellular organisms such as macrozooplankton or higher trophic level species when calculating this index.

References

Krebs, C.J., 1999. Estimating community parameters, in: Ecological Methodology. Addison-Welsey Education Publishers, Inc., pp. 373–409.

Lance G.N., Williams W.T. (1967): Mixed-data classificatory programs, I.) Agglomerative Systems. Australian Computer Journal 1:15-20.

Tuomisto, H., 2012. An updated consumer’s guide to evenness and related indices. Oikos 121, 1203–1218. doi:10.1111/j.1600-0706.2011.19897.x

Appendix F: F_{MSY} values and F_{MSY} ranges

Table F.1. Table presenting the single point F_{MSY} values and the F_{MSY} ranges used in the modelling of the scenarios.

Fishery	**Species**	$F_{status\ quo}$	F_{MSY}	**Reference**	F_{MSY} lower	F_{MSY} upper	**Reference**
Demersal	**Cod**	0.60	0.17	ICES, 2016b	0.11	0.25	ICES, 2016a
	Whiting	0.06	0.18	ICES, 2016b	0.15	0.18	ICES, 2016a
	Haddock	0.17	0.19	ICES, 2016c	0.18	0.19	ICES, 2016c
	Saithe	0.07	0.36	ICES, 2016c	0.20	0.42	ICES, 2015
	Hake	0.04	0.28	ICES, 2016d	0.18	0.45	ICES, 2016a
	Anglerfish	0.14	0.31	ICES, 2016d	0.18*	0.41*	ICES, 2016a
Pelagic	**Herring**	0.21	0.16	ICES, 2016e			
	Mackerel	0.13	0.22	ICES, 2016f			
	Horse mackerel	0.30	0.09	ICES, 2016f			
	Blue whiting	0.11	0.30	ICES, 2016f			
Crustaceans	**Nephrops**	0.08	0.109	ICES, 2016b			

*Since no F_{MSY} range values are defined for Anglersfish in ICES area VIa the F_{MSY} range values for ICES areas IIXc and IXa were used instead as the best available proxy.

References

ICES 2016a. EU request to ICES to provide FMSY ranges for selected stocks in ICES subareas 5 to 10. Version 4, 11 July 2016. ICES Advice 2016, Book 5

ICES 2016b. Report of the Working Group for the Celtic Seas Ecoregion (WGCSE). ICES CM 2016/ACOM:13

ICES 2016c. Report of the Working Group on the Assessment of Demersal Stocks in the North Sea and Skagerrak (WGNSSK). ICES CM 2016/ ACOM:14

ICES 2016d. Report of the Working Group for the Bay of Biscay and the Iberian waters Ecoregion (WGBIE). ICES CM/ACOM:12

ICES 2016e. Report of the Herring Assessment Working Group for the Area South of 62˚N (HAWG). ICES CM 2016/ACOM:07

ICES 2016f. Report of the Working Group on Widely Distributed Stocks (WGWIDE).
ICES CM 2016/ACOM:16

ICES 2015. EU request to ICES to provide FMSY ranges for selected North Sea and Baltic Sea stocks. ICES Advice 2015, Book 6. Version 6, 30-6-2016.

Appendix G: Spatial distribution of key demersal species

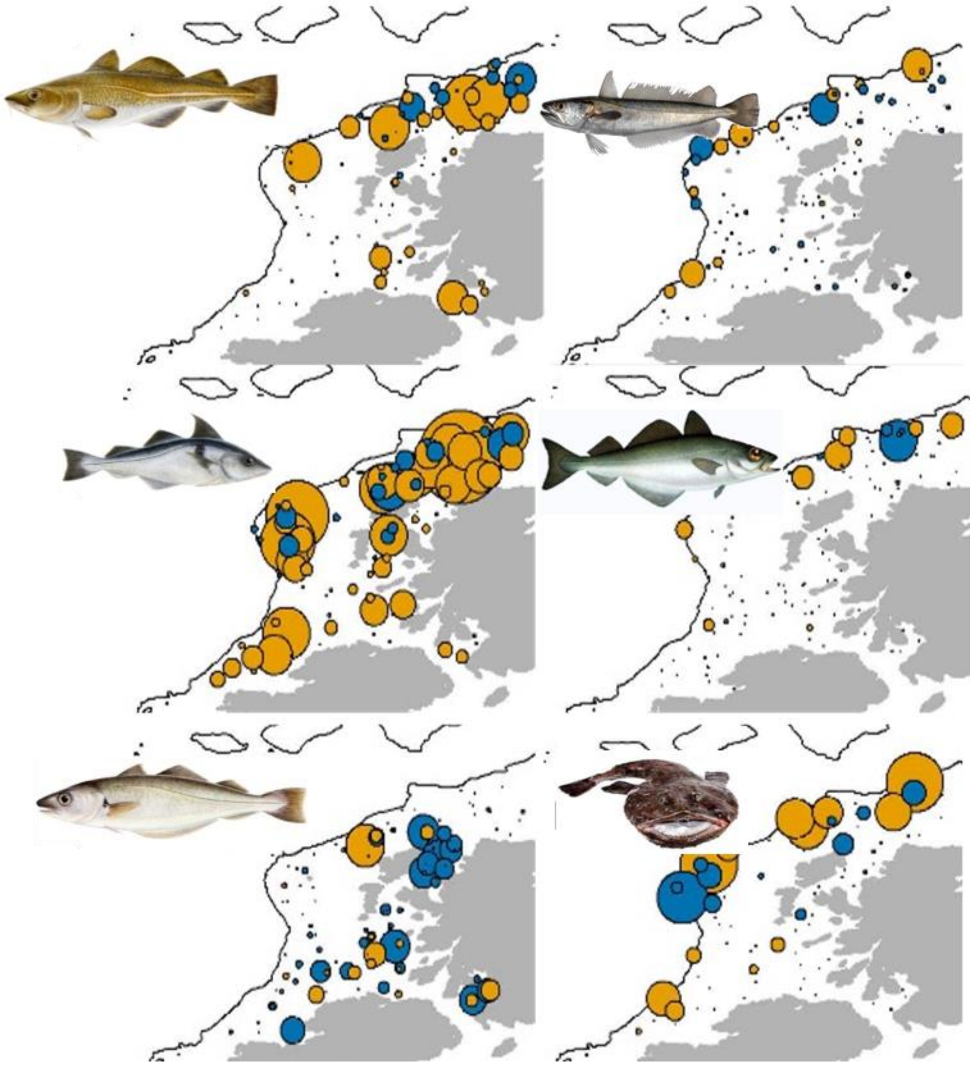

Fig. G.1. Spatial distribution of bottom-trawl survey swept-area density observations from Quarter 1 (yellow) and Quarter 3 (blue) International Bottom-Trawl Surveys for cod (*top left*), hake (*top right*), haddock (*middle left*), saithe (*middle right*), whiting (*bottom left*), and anglerfish (*bottom right*).

Appendix H: MCA scenario data

Table H.1. The table shows the forecasted estimates for indicators for each scenario in the years 2020 and 2025. The abbreviations of indicator names used in the MCA are shown in parentheses. Column headings refer to a) Scenarios names as used in the MCA: MixMEY (Mixed MEY), Gadiod Recovery (GRec), Status Quo (SQ), Maximum Sustainable Yield (MSY), Gadiod recovery and seal cull (GRecSC), Spatial F (SpatialF); b) The minimum and maximum limits for each indicator in the MCA is respectively set to the lowest and highest value across the scenarios. These limits are needs to be defined in the MCA and represent the window of opportunity for each indicator, across the modelled scenarios.

2020	**MixMEY**	**GRec**	**SQ**	**MSY**	**GRecSC**	**SpatialF**	**Min**	**Max**	**Units**
Catch value									
Demersal (VDem20)	198	220	93,1	220	224	206	93,1	224	MEUR
Pelagic (VPel20)	356	355	278	361	355	367	278	367	MEUR
Nephrops (VNep20)	50,8	49,8	57	50,6	49,6	49,2	49,2	57	MEUR
Profitability									
Demersal (PDem20)	108	95,3	68	101	99,4	45,1	45,1	108	MEUR
Pelagic (PPel20)	248	247	196	251	247	256	196	256	MEUR
Nephrops (PNep20)	30,5	29,5	36,7	30,2	29,3	28,9	28,9	36,7	MEUR
Stocks									
Haddock (Had20)	64,5	57,9	48,4	52,5	57,7	86,9	48,4	86,9	1000t
Saithe (Sai20)	150	118	351	145	122	117	117	351	1000t
Hake (Hake20)	214	168	344	168	171	116	116	344	1000t
Nephrops (Nep20)	136	133	152	135	133	131	131	152	1000t
Cod (Cod20)	3,88	5,59	0,182	5,37	5,8	9,26	0,182	9,26	1000t
Whi (Whi20)	5,88	15,5	4,07	6,06	15,5	10,1	4,07	15,5	1000t
Foodweb									
Prey fish (Prey20)	1096	1146	832	1116	1145	1101	832	1146	Mt
Balenced Evenness (BE_20)	0,5063	0,5217	0,4923	0,5154	0,5234	0,5289	0,4923	0,5289	#
Seabirds Bird20	2,81	2,09	1,83	2,08	2,1	2,04	1,83	2,81	1000t
Seals (Seal20)	7,59	7,19	8,92	7,28	6,13	6,84	6,13	8,92	1000t
2025									
Catch value									
VDem25	204	228	97,2	226	235	202	97,2	235	MEUR
VPel25	328	316	286	334	316	342	286	342	MEUR
VNep25	46,2	44,3	58,2	45,5	44	43,1	43,1	58,2	MEUR

Profitability									
PDem25	114	104	72,1	107	110	41,3	41,3	114	MEUR
PPel25	229	221	202	233	221	239	202	239	MEUR
PNep25	25,9	24	37,9	25,2	23,7	22,8	22,8	37,9	MEUR
Stocks									
Had25	79,2	70,1	57,1	63	69,2	98,3	57,1	98,3	1000t
Sai25	145	111	350	143	117	108	108	350	1000t
Hake25	202	157	358	157	161	108	108	358	1000t
Nep25	123	118	156	122	118	115	115	156	1000t
Cod25	11,9	19,7	0,0394	16,9	20,5	31,3	0,0394	31,3	1000t
Whi25	10,1	38,5	4,43	10,1	37,7	18,8	4,43	38,5	1000t
Foodweb									
Prey25	1199	1233	803	1233	1228	1174	803	1233	Mt
BE_25	0,5204	0,5262	0,4767	0,5317	0,526	0,529	0,4767	0,5317	#
Bird25	2,85	2,43	1,76	2,44	2,43	2,31	1,76	2,85	1000t
Seal25	7,09	6,65	9,24	6,75	5,32	6,36	5,32	9,24	1000t

Appendix I: Detailed MCA evaluation results

Table I.1. Table presenting contributions by each indicator to the overall scenario evaluation score in the MCA.

Year	Indicator	MixMEY	Grec	SQ	MSY	GRecSC	SpatialF
2020	Demersal catch value	0,072	0,087	0,000	0,087	0,090	0,078
2020	Pelagic catch value	0,039	0,039	0,000	0,042	0,039	0,045
2020	Nephrops catch value	0,009	0,004	0,045	0,008	0,002	0,000
2020	Demersal profitability	0,090	0,072	0,033	0,080	0,078	0,000
2020	Pelagic profitability	0,039	0,038	0,000	0,041	0,038	0,045
2020	Nephrops profitability	0,009	0,004	0,045	0,008	0,002	0,000
2020	Cod SSB	0,005	0,007	0,000	0,007	0,008	0,012
2020	Whiting SSB	0,003	0,009	0,002	0,003	0,009	0,006
2020	Haddock SSB	0,018	0,018	0,018	0,018	0,018	0,018
2020	Hake TSB	0,018	0,018	0,018	0,018	0,018	0,014
2020	Saithe TSB	0,008	0,007	0,018	0,008	0,007	0,007
2020	Neprops TSB	0,018	0,018	0,018	0,018	0,018	0,018
2020	Balanced Evenness	0,009	0,019	0,000	0,015	0,020	0,024
2020	Prey fish biomass	0,020	0,024	0,000	0,022	0,024	0,021
2020	Seabird biomass	0,024	0,006	0,000	0,006	0,007	0,005
2020	Seal biomass	0,023	0,021	0,022	0,022	0,000	0,016
2025	Demersal catch value	0,039	0,048	0,000	0,047	0,050	0,038
2025	Pelagic catch value	0,019	0,013	0,000	0,021	0,013	0,025
2025	Nephrops catch value	0,005	0,002	0,025	0,004	0,002	0,000
2025	Demersal profitability	0,050	0,043	0,021	0,045	0,047	0,000
2025	Pelagic profitability	0,018	0,013	0,000	0,021	0,013	0,025
2025	Nephrops profitability	0,005	0,002	0,025	0,004	0,002	0,000
2025	Cod SSB	0,013	0,026	0,000	0,020	0,027	0,030
2025	Whiting SSB	0,005	0,023	0,002	0,005	0,022	0,009
2025	Haddock SSB	0,015	0,015	0,015	0,015	0,015	0,015
2025	Hake TSB	0,015	0,015	0,015	0,015	0,015	0,010
2025	Saithe TSB	0,007	0,005	0,015	0,007	0,005	0,005
2025	Neprops TSB	0,015	0,015	0,015	0,015	0,015	0,015
2025	Balanced Evenness	0,016	0,018	0,000	0,020	0,018	0,019
2025	Prey fish biomass	0,018	0,020	0,000	0,020	0,020	0,017
2025	Seabird biomass	0,020	0,012	0,000	0,013	0,012	0,010
2025	Seal biomass	0,019	0,016	0,019	0,017	0,000	0,014
	Total score	0,684	0,677	0,372	0,692	0,653	0,541

Appendix J. Variability of economic indicators

Table J.1. Table presenting the variability of economic indicators across the scenarios for the fleets expressed as coefficient of variation.

	2020		2025	
Fleet	Catch value	Profitability	Catch value	Profitability
Demersal	0,259	0,283	0,259	0,315
Pelagic	0,096	0,092	0,061	0,058
Nephrops	0,057	0,095	0,121	0,213

Appendix K: Selected plots from the sensitivity analysis

The sensitivity analysis can be conducted graphically in the interactive MCA program at the web-location: https://mareframe.github.io/dsf/dev/MCA2/DST.html?model=scotland_weighted

To get the same visual outputs as those described below, the user should exclude the MSY alternative from the analysis in accordance with the reasoning provided in the main text (double click on" MSY" and click "exclude alternative").

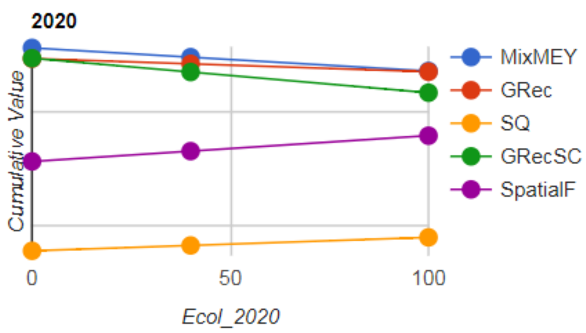

Fig. K.1. Sensitivity of the decision rank for changes in the weight for ecology in 2020. The initial weight is 40. The gadoid recovery strategy will outperform the mixed MEY strategy only if the decision weight assigned to the ecological objectives approaches 100.

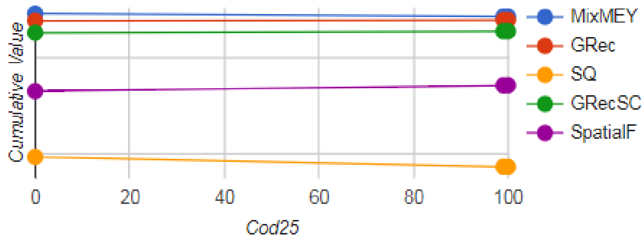

Figure K.2. Sensitivity of the decision rank for changes in the weight for the cod stock in 2025. The assigned weight is 100. Even a reduction to of the cod stock estimate to zero will not alter the top ranking decision.

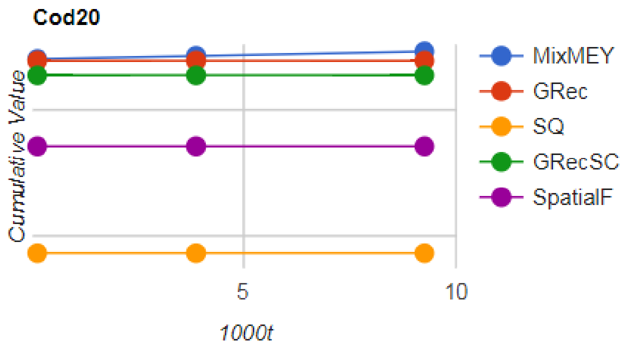

Figure K.3. Impact of uncertainty in the stock projection on the decision rank using the biomass estimate for the cod stock as an example. The prediction by the ecosystem model is 3.88 thousand tonnes in 2020. The mixed MEY strategy will perform best even if the prediction is highly biased. Note that the results are conditional for everything else being fixed in the decision model.