ICES Journal of Marine Science

ICES CIEM International Council for the Exploration of the Sea Conseil International pour l'Exploration de la Mer

ICES Journal of Marine Science; doi:10.1093/icesjms/fsw028

Assessing fishery and ecological consequences of alternate management options for multispecies fisheries

Robert Brian Thorpe[1]*, Paul James Dolder[1], Stuart Reeves[1], Peter Robinson[1], and Simon Jennings[1,2]

[1]*Centre for Environment, Fisheries and Aquaculture Science, Lowestoft, Suffolk NR33 0HT, UK*
[2]*School of Environmental Sciences, University of East Anglia, Norwich NR4 7TJ, UK*

**Corresponding author: tel: +44 1502 524555; fax: +44 1502 513865; e-mail: robert.thorpe@cefas.co.uk*

Thorpe, R. B., Dolder, P. J., Reeves, S., Robinson, P., and Jennings, S. Assessing fishery and ecological consequences of alternate management options for multispecies fisheries. – ICES Journal of Marine Science, doi: 10.1093/icesjms/fsw028.

Received 24 September 2015; revised 4 February 2016; accepted 7 February 2016.

Demands for management advice on mixed and multispecies fisheries pose many challenges, further complicated by corresponding requests for advice on the environmental impacts of alternate management options. Here, we develop, and apply to North Sea fisheries, a method for collectively assessing the effects of, and interplay between, technical interactions, multispecies interactions, and the environmental effects of fishing. Ecological interactions involving 21 species are characterized with an ensemble of 188 plausible parameterizations of size-based multispecies models, and four fleets (beam trawl, otter trawl, industrial, and pelagic) characterized with catch composition data. We use the method to evaluate biomass and economic yields, alongside the risk of stock depletion and changes in the value of community indicators, for 10 000 alternate fishing scenarios (combinations of rates of fishing mortality *F* and fleet configuration) and present the risk vs. reward trade-offs. Technical and multispecies interactions linked to the beam and otter trawl fleets were predicted to have the strongest effects on fisheries yield and value, risk of stock collapse and fish community indicators. Increasing beam trawl effort led to greater increases in beam trawl yield when otter trawl effort was low. If otter trawl effort was high, increases in beam trawl effort led to reduced overall yield. Given the high value of demersal species, permutations of fleet effort leading to high total yield (generated primarily by pelagic species) were not the same as permutations leading to high catch values. A transition from *F* for 1990 to 2010 to F_{MSY}, but without changes in fleet configuration, reduced risk of stock collapse without affecting long-term weight or value of yield. Our approach directly addresses the need for assessment methods that treat mixed and multispecies issues collectively, address uncertainty, and take account of trade-offs between weight and value of yield, state of stocks and state of the environment.

Keywords: fisheries, indicator, LFI, mixed, multispecies, North Sea, risk, trade-off, uncertainty.

Introduction

Mixed-fisheries catch different species and stocks with the same gears at the same time. Typically, these species and stocks are also interacting ecologically. Analyses to determine stock status, yields, fishing impacts, and management options in mixed-fisheries should therefore address technical interactions between fleets and species as well as multispecies ecological interactions (Murawski, 1991; Vinther *et al.*, 2004; Rindorf *et al.*, 2013). If species and stocks in a mixed-fishery are managed using separate single-species targets, then technical and multispecies interactions usually mean that the targets cannot be met simultaneously, with progress towards a target for one species or stock compromising status or potential yield of others (e.g. Gislason, 1999; Collie and Gislason, 2001; Gray *et al.*, 2008; Lynam and Mackinson, 2015).

Technical interactions are an important issue for managers of the international demersal fisheries of the North Sea, because multiple stocks are exploited in mixed-fisheries. The need for management to address the poor conservation status of North Sea cod while still allowing fishing on other, less depleted stocks caught in the same fisheries, has stimulated several analyses of the consequences of technical interactions (Vinther *et al.*, 2004; Reeves *et al.*, 2009; Ulrich *et al.*, 2011). Such analyses have sought to identify trade-offs and their consequence and to define alternate options for balancing conflicting objects (Hoff *et al.*, 2010; Ulrich *et al.*, 2011; ICES, 2014a). Existing evaluations of mixed-fisheries issues in the North Sea have tended to focus on short timescales; either TAC setting (Vinther *et al.*, 2004) or quantifying impacts on target stocks (Ulrich *et al.*, 2011). For some North Sea demersal stocks, mixed-fishery evaluations now form part of the annual ICES advice (ICES, 2014a,b).

Multispecies interactions influence reference points and assessments of fishing effects (Pope, 1979, 1991; Sparre, 1991). The intensity of multispecies interactions and their stability in space and time are complex and notoriously difficult to predict, especially in communities with high biodiversity. Consequently, existing analyses have often focused on small groups of interacting species for which detailed diet data are available (e.g. Gislason, 1999; Gårdmark *et al.*, 2013; ICES, 2013).

Moves towards the collective treatment of mixed-fishery and multispecies issues are necessary in Europe because they will help to address management questions that have become particularly salient with the recent reform of the Common Fisheries Policy (CFP) and the requirement of the Marine Strategy Framework Directive that the CFP should contribute to achieving "Good Environmental Status" for biodiversity and foodwebs (EU, 2008, 2010, 2013; Rice, 2011). Specifically, advice that accounts for technical interactions between fleets and species as well as multispecies ecological interactions is needed to improve assessments of the effects of alternate fleet fishing strategies on fisheries yield and value, the state of stocks and the state of fish communities. The interplay between the state of target stocks and the state of fish community has become a pressing issue because metrics of the state of fish communities are increasingly identified as indicators of the state of biodiversity and foodwebs (Greenstreet *et al.*, 2012).

In principle, multispecies and mixed-fisheries interactions can be investigated with holistic end-to-end models which consider large numbers of ecosystem components and may include comprehensive descriptions of fleet dynamics (e.g. Christensen and Walters, 2004; Fulton *et al.*, 2011). However, the effects of parameter uncertainty on complex ecosystem models are poorly understood and may be challenging to address owing to practical constraints such as long run-times and very limited data for parameterization or validation. Further, high model complexity is not a prerequisite for addressing many operational management questions and a range of less complex models that further simplify ecosystems and/or deal with subsets of ecosystems can usefully be adopted (e.g. Fulton *et al.*, 2003; Butterworth and Plaganyi, 2004). Size-based models provide one tool for investigating the effects of multispecies interactions and their consequences for fisheries and the fish community. These models predict changes in the size and abundance of interacting species as a function of fishing mortality F (Pope *et al.*, 2006; Hall *et al.*, 2006; Andersen and Rice, 2010; Blanchard *et al.*, 2014). They can be used to estimate reference points in a multispecies context and, because they provide full accounting for species, body size and abundance, and can also be used to estimate metrics that have been proposed or specified as indicators of fishing effects (Hall *et al.*, 2006; Pope *et al.*, 2006), such as the large fish indicator (LFI; Greenstreet *et al.*, 2011), or the slope of the community size spectrum (SSS; Shin *et al.*, 2005).

Previous analyses have shown that estimates of reference points and fishing effects based on multispecies models are strongly affected by parameter uncertainty (Hill *et al.*, 2007; Gaichas *et al.*, 2012; Link *et al.*, 2012). The relatively simple structure and parameterization of size-based models enables evaluation of the effects of parameter uncertainty. For the size-based model of Hall *et al.* (2006), Thorpe *et al.* (2015) reported a method to quantify this by taking an ensemble of models for a full range of parameter combinations, estimated from data and literature, and reducing this to a filtered ensemble of models by screening outputs against data. Effects of parameter uncertainty on model predictions can then be expressed by presenting the distributions of outputs from the filtered ensemble.

Here, we use a size-based fish community model to investigate the effects of technical interactions among North Sea fleets on (i) on stock sizes, yield and value; (ii) trade-offs between the quantity and value of sustainable yield; (iii) risks to biodiversity (depletion of the most sensitive species); and (iv) risks of not achieving targets for foodweb indicators. We also assess whether fish community indicators can guide assessment and management of multispecies fisheries given uncertainty. Our approach is based on a simplified fleet categorization and on those species that currently dominate catches. In this form, the results produced are suited to informing strategic decisions about fleet investment and allocation to provide sustainable long-term yield and to meet environmental targets. However, the approach could readily be modified to accommodate a more complex set of fleets or species and to inform short-term decision making.

Methods

We described the complex of vessels fishing the North Sea in four fleet categories: beam trawlers, industrial trawlers, otter trawlers, and pelagic trawlers. While ICES (2012a,b), for example, used catch data for 88 combinations of nation, vessel size, gear type, and mesh size (as a proxy for target species) to characterize the area's demersal fisheries, we preferred the simpler four fleet classification to increase the accessibility and generality of our results. These four fleets take >90% of the North Sea catch. The catch compositions for the four fleets (Table 1) were determined from data reported by Member States to the EU Scientific Technical and Economic Committee for Fisheries (STECF, 2014). These included landings as well as estimated discards, where available, by gear type. In this paper, we refer to these catch compositions as the "historic" case, which can be broadly characterized as follows. Beam trawlers mainly target flatfish, particularly sole and plaice, but also take a bycatch of other species such as cod and whiting. Industrial trawlers use small mesh trawls to target forage fish such as sandeel and Norway pout for use as fishmeal and fishoil, but may take a small bycatch of whitefish (cod, haddock and whiting). Otter trawlers use demersal trawls to target bottom dwelling fish including cod, haddock, and saithe as well as the crustacean *Nephrops norvegicus*, while also taking some catches of flatfish, such as plaice. Pelagic trawlers mainly target herring and mackerel, with smaller catches of other pelagic species. In addition to the "historic" case, we also considered an "idealized" case where each stock was caught by just one fleet. Idealized here refers to alignment of the fleets with the stock boundaries, to allow us to predict the consequences of removing mixed-fishery effects, but "idealized" should not be interpreted as meaning "ideal" for the fishing industry. The historic case was used to examine the effects of a transition to F_{MSY}, while we compared the "historic" and "idealized" cases to assess how much additional catch and value would be available if fisheries for species or groups of species could be managed independently.

Multispecies interactions were described with a modified version of the length-based multispecies model developed by Hall *et al.* (2006) and subsequently modified by Rochet *et al.* (2011) and Thorpe *et al.* (2015). Briefly, the model represents 21 fish species

Table 1. Relative catch per unit effort by the four fleets as calculated from STECF (2014) data for the period 2003–2013 and for the idealized case, in which each stock is caught by only one fleet.

	Historic fleet (STECF)				Idealized fleet						
Stock	**B**	**I**	**O**	**P**	**B**	**I**	**O**	**P**	$F_{1990-2010}$	F_{MSY}	**Price (£/t)**
Sprat	0.0002	0.4719	0.0002	0.5278	0	1	0	0	1.060	1.30	203
Norway pout	0.0000	0.8891	0.0016	0.1094	0	1	0	0	0.477	0.35	90
Sandeel	0.0000	0.9685	0.0030	0.0312	0	1	0	0	0.573	0.35	188
Poor cod	0.0551	0.0268	0.9140	0.0041	0	0	1	0	0.548	0.72	200
Long rough dab	0.6599	0.0055	0.3399	0.0008	1	0	0	0	0.548	0.60	668
Dab	0.6599	0.0055	0.3399	0.0008	1	0	0	0	0.548	0.41	668
Herring	0.0000	0.1499	0.0013	0.8488	0	0	0	1	0.247	0.25	327
Horse mackerel	0.0008	0.0421	0.0958	0.8614	0	0	0	1	0.627	0.50	498
Lemon sole	0.6599	0.0055	0.3399	0.0008	1	0	0	0	0.548	0.33	2829
Sole	0.9746	0.0017	0.0237	0.0001	1	0	0	0	0.534	0.22	6941
Mackerel	0.0000	0.1051	0.0122	0.8827	0	0	0	1	0.627	0.32	770
Whiting	0.0680	0.0557	0.8695	0.0068	0	0	1	0	0.365	0.21	1017
Witch	0.6599	0.0055	0.3399	0.0008	1	0	0	0	0.548	0.27	975
Gurnard	0.0511	0.0268	0.9140	0.0041	0	0	1	0	0.548	0.27	399
Plaice	0.6599	0.0055	0.3399	0.0008	1	0	0	0	0.548	0.25	1049
Starry ray	0.6599	0.0055	0.3399	0.0008	1	0	0	0	0.548	0.15	736
Haddock	0.0022	0.0078	0.9885	0.0014	0	0	1	0	0.577	0.30	1243
Cuckoo ray	0.6599	0.0055	0.3399	0.0008	1	0	0	0	0.548	0.11	736
Monkfish	0.0551	0.0268	0.9140	0.0041	0	0	1	0	0.548	0.10	2911
Cod	0.0950	0.0167	0.8841	0.0042	0	0	1	0	0.882	0.19	2015
Saithe	0.0001	0.0189	0.9794	0.0026	0	0	1	0	0.377	0.30	924

Fleet codes: B, beam trawl; I, industrial; O, otter and P, pelagic. The 1990–2010 average fishing mortalities are taken from ICES (2012d) and mean prices are for UK vessels landing catches from 2008 to 2012 (see text).

in 32 equal length-classes spanning the full size ranges of all species. Individuals progress through length-classes as they grow then mature at a threshold length at maturity. Reproduction is described with a hockey-stick spawner–recruit relationship, which determines the numbers of recruits entering the smallest size class based on the biomass of mature individuals. Species' dynamics are linked via predation mortality (M2), which varies with predator abundance, size-, and species-preference as defined with a diet matrix. Individuals are also susceptible to residual natural mortality (M1) and F. Parameterization and validation of the model are described in Thorpe *et al.* (2015). Fishing mortality (F) is applied assuming species-specific relative size-selectivity established from fits to F at length data from the IBTS survey (ICES, 2012b,c).

The consequences of parameter uncertainty were assessed by developing 78 125 models, with all combinations of parameters drawn from ranges that spanned parameter values estimated from data and literature. Details of the parameter choices, and their underlying rationale can be found in Thorpe *et al.* (2015), Supplementary Table S1. Models in this unfiltered ensemble were then screened against data to identify plausible models. The screening criteria were (i) all species should persist when there is no fishing and (ii) that mean predicted biomass of assessed species after 30 years simulated fishing at 1990–2010 rates (ICES 2012d) should be within a factor of two of the biomass estimated in ICES (2012d). Application of the screening criteria led to a filtered ensemble of 188 models. We used a factor of two because biomass estimates from the single species assessments are uncertain owing to uncertainties in the assessment data and methods and because environmental trends and stochastic processes influence recruitment dynamics and hence variation in abundance in the real world. However, previous analyses of the effects of changing this factor have shown that such leniency has small impacts on the predictions made with the filtered ensemble (Thorpe *et al.*, 2015).

We took mean F for the period 1990–2010 ($F_{1990-2010}$) as a measure of "historic" fishing rates (F_{HIST} in Thorpe *et al.*, 2015). When species were not assessed, F was assigned by grouping non-assessed stocks with assessed stocks with similar morphology and behaviour. Stocks within each group were assumed to be fished at the same F (Pope *et al.*, 2000). Poor cod, gurnard, and monkfish were grouped with haddock, cod, and whiting. Long rough dab, dab, lemon sole, witch, starry ray, and cuckoo ray were grouped with plaice.

The mean catchability (or realized mortality per unit of effort) for a given stock by a given fleet was calculated from the STECF and ICES data over the period 2003–2013 for 12 stocks where quantitative assessments of historic F were available (cod, haddock, herring, horse mackerel, mackerel, Norway pout, plaice, saithe, sandeel, sole, and whiting). Mean catchability, q, was estimated as:

$$\bar{q}_{fl,s} = \frac{1}{n}\sum_{y=1}^{n}\frac{F_{fl,s,y}}{E_{fl,y}},$$

where $\bar{q}_{fl,s}$ is the mean catchability for fleet fl on stock s over the years $1, \ldots, n$, with $E_{fl,y}$ and $F_{fl,s,y}$ the fishing effort and fishing mortality exerted by the fleet in year y. Fishing mortality for a given fleet was estimated from the catch by that fleet (STECF, 2014) as a proportion of the total catch, given F estimated in the assessment (downloaded from ICES online stock assessment database, last accessed February 2015). This approach characterizes the relative strength of technical interactions for each fishery and stock over a number of years independent of the level of stock biomass and fishing effort. Projected F in response to changes in effort were based on this constant q (since $F = qE$). A linear relationship between F and E was assumed in our analyses, but alternative forms could be considered which include density-dependent effects on catchability and catch (e.g. Thøgersen *et al.*, 2012).

For each stock, a relative catch by fleet was calculated if they fished at 1990–2010 average levels of effort ($E_{1990-2010}$). The total relative catch for each stock was converted into F by assuming that a relative effort of 1 for each fleet (corresponding to average $E_{1990-2010}$) would generate a fishing mortality of $F_{1990-2010}$. The variable F_{MSY} was approximated by using the fleet fishing scenario generating the mortality closest to individual stock estimates of F_{MSY} when averaged across all 21 stocks. Non-assessed stocks were grouped with the assessed ones as previously described. Calculating the mortalities in this manner preserves the relative q by fleets and the overall F, as shown in Table 1.

The fishing pressure exerted by each of the four fleets was expressed as a proportion of an $F_{1990-2010}$ baseline (0, 0.2, 0.4, 0.6, 0.8, 1.0, 1.2, 1.4, 1.7, and 2.0 × $F_{1990-2010}$). All combinations of all levels of fishing were considered giving 10 000 scenarios, each of which was run with the 188 member filtered ensemble. All simulations were run for 50 years and mean outputs from the last 10 years of the simulation were used to calculate stock biomass and community metrics. In every case biomass projections reached an equilibrium before the end of the simulation. Fishery yields were expressed in terms of weight and value. Values were taken as the product of weight and mean price for each species, where price was the mean first sale price for the period 2008–2012, as determined from data for UK vessels landing into ports in the UK and internationally.

The community metrics calculated were the biomass fraction >40 cm (dubbed the LFI) and the slope of the size-spectrum (slope of relationship between log numbers in each log size class and log size, SSS). We concentrated on the LFI because it has been proposed as an appropriate community measure with a suggested target value of 0.3 (Greenstreet *et al.*, 2011) and on the SSS because it was shown to be the indicator that best linked fishing-induced community responses to F in previous analyses (Thorpe *et al.*, 2015). Ensemble means and individual ensemble member results (for the 5 and 95% percentile surfaces) were used to estimate the mean response and associated uncertainty for each scenario. Stocks were deemed to be at risk of collapse if their biomass fell to <10% of unfished biomass. The ensemble mean number of stocks at risk was taken to represent the overall level of risk associated with a given scenario, although other assumptions could readily be accommodated to suit risk aversion and reference points adopted in the management system.

Results

Technical and multispecies interactions had profound effects on yield, catch value, and the effects of fishing on stock status and indicators. We focus on the beam and otter fleets (Figure 1) as these show the most marked changes, but responses for all four fleets are shown in Supplementary Figures S1–S4. Industrial trawling makes a modest contribution to total yield irrespective of the effort exerted by other fleets while highest overall yields are achieved when pelagic and industrial effort are closer to $E_{1990-2010}$ (Supplementary Figures S1–S4). For the beam and otter trawl fleets, increasing beam trawl effort is predicted to result in greater increases in beam trawl yield when otter trawl effort is low. When otter effort is moderate, increasing beam trawling leads to increased yield at first, but then at higher effort, yield starts to decline. When otter effort is already very high, increases in beam trawl effort actually reduce overall yield (Figure 1a). The value of yield varies substantially among species (Table 1) and when biomass yields are converted to value, the beam and otter fleets make a relatively greater contribution to value as effort increases and the pelagic, and particularly the industrial fleets, are relatively less important. Overall, economic value of yield attains a maximum in the region where the beam effort is ∼0.4–0.8 × $E_{1990-2010}$ and otter effort is 0.8–1.2 × $E_{1990-2010}$ (Figure 1b). In comparison, the optimum effort in terms of tonnage is ∼1.4 × $E_{1990-2010}$ for the otter fleet and 0.4 × $E_{1990-2010}$ for the beam fleet (Figure 1a). Thus observed patterns of fishing between 1990 and 2010 tend to maximize monetary value rather than weight of catch.

Owing to technical and multispecies interactions, there may be risks to more sensitive species as effort is increased. If a stock is defined as 'at risk' when biomass falls to <10% of the unfished biomass, then even those levels of fishing effort that result in intermediate levels of biomass yield or value of yield may place some stocks at risk (Figure 1c).

In general, higher yields are linked to higher risk, but there are differences across the fleets. The industrial fleet makes modest contributions to yield and risk. Yield from the beam fishery is more valuable, but is linked to higher risk. Yields from the pelagic fleet have intermediate value but are obtained at low risk. The otter fishery produces the highest yields at the highest risk. In general, as effort increases, yield increases, quickly at first, then more slowly. Then as the effort continues to rise towards $E_{1990-2010}$, yield plateaus while the risk of depletion increases sharply. Further increases in effort start to reduce yield and markedly increase risk. For the otter and beam fleets (Figure 1c), low levels of risk are achieved in an area roughly bounded by the zone where relative effort for otter and beam fleets is <0.6, whereas pelagic and industrial fleets can operate closer to $E_{1990-2010}$ without increasing our measure of risk. While the range of effort combinations where all 21 stocks have a <5% chance of depletion is relatively small, these combinations can still yield over 70% of the mean total yield and over 80% of the mean total gross economic yield (Figure 1a and b). For our measure of risk, $E_{1990-2010}$ (leading to $F_{1990-2010}$; point 1.0, 1.0 on Figure 1c) puts 3 or 4 stocks at risk. If all stocks were fished at F_{MSY} as opposed to $F_{1990-2010}$ (Table 1), corresponding to a reduction in beam and otter effort of ∼50% (the region near 0.5, 0.5 in Figure 1c), risk is reduced to near zero. Reducing F from $F_{1990-2010}$ to F_{MSY} is therefore expected to greatly lessen risk with only a modest loss of yield.

The LFI and SSS respond to changes in effort among fleets (Figure 1d and Supplementary Figure S4: LFI, SSS not shown) in a similar manner. They are much more sensitive to otter and beam trawling effort than to pelagic or industrial effort. We find that the LFI is particularly sensitive to the otter fleet effort at all levels of otter trawling. The LFI also declines with beam trawling effort, though it is most sensitive to changes in beam trawl effort when effort increases from low-to-moderate levels, and becomes largely insensitive as effort increases further. The strong differential sensitivity of the LFI (and SSS) to fleet effort suggests that there is no simple relation between these indicators and average fishing mortality in a multifleet world. However, both indicators decline to low levels as a response to increasing F in otter trawl fisheries, as these fisheries catch the largest individuals and species (Figures 2 and 3).

While high values of the LFI are achieved with no fishing, and there is a general decrease in the LFI with increased effort, a wide spread of indicator values results from changes in effort allocation among fleets. Consequently, an average fishing mortality of 0.2 year^{-1} is predicted to correspond to an LFI of 0.25 for an otter trawl fishery but nearly 0.5 for an industrial/pelagic fishery: taking the LFI from below the recommended target to above the

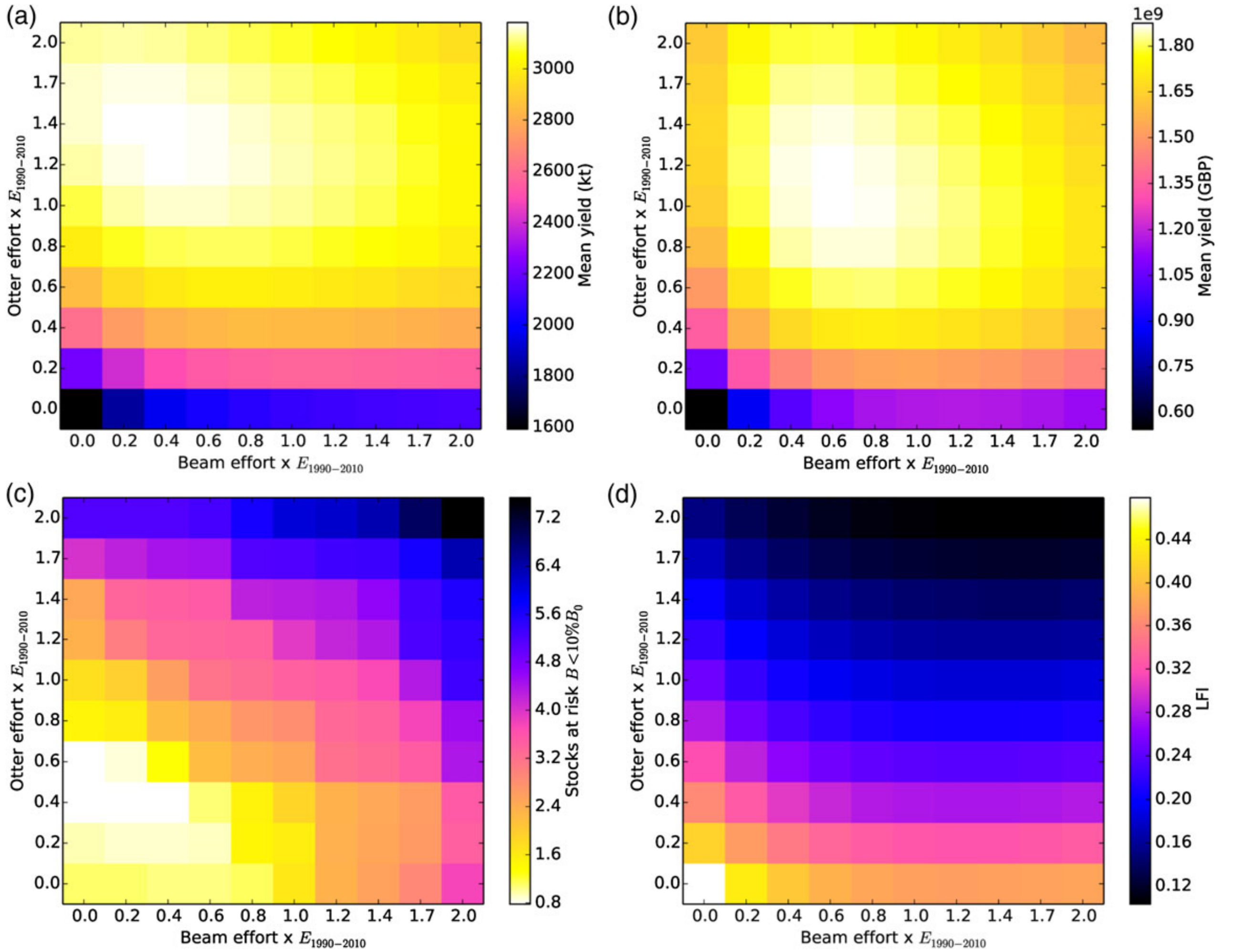

Figure 1. Heat maps showing (a) mean total yield (kg), (b) mean total gross economic yield (£), (c) mean number of stocks at risk of 90% biomass depletion relative to the unfished state, and (d) mean simulated LFI as a function of relative fishing effort in the otter and beam trawl fisheries, assuming that pelagic and industrial fleets are each operating at $E_{1990-2010}$.

recommended target (Figure 2). Further the LFI or SSS can potentially increase as levels of risk increase (Figure 3) depending on the type of fishing activity which is dominant. A modest reduction in otter effort from $E_{1990-2010}$ increases the LFI more than intensive reductions in effort by other three fleets, so a switch away from otter trawling may increase the LFI even if some stocks were put at risk. Further, the simulations suggest that managing the system to keep above an LFI target of 0.3 would probably result in low risk but at the cost of low yield (Figure 4a). A management outcome providing higher yield and low risk would be linked to an LFI of $\sim$0.3, but an LFI of this value is not uniquely associated with this outcome; it could equally indicate moderate yield and high risk.

For illustrative purposes our analyses included a broad range of fleet permutations but, in practice, deviations from the status-quo fleet composition would be limited by commercial, political, and social factors. When we assume that the relative effort of each fleet remains within a factor of two of all other fleets, the reduced set of interactions between the value of yield from the fishery, the fraction of the ensemble meeting the LFI target and the number of stocks at risk outcomes are further constrained (Figure 4c and d). In this subset of scenarios, an increase in fishing effort initially leads to a reduction in LFI but the yield rises without an increase in risk. High yield and low risk are uniquely associated with an ensemble mean LFI of $\sim$0.3. Then, if fishing effort increases further, yield stays roughly constant but risk increases (Figure 4c) and the ensemble mean LFI decreases. Once parameter uncertainty is included, however, the apparently clear signal is blurred (Figure 4d) and there are cases where high-risk fishing can coexist with a higher LFI and low-risk fishing with a lower LFI.

The transition to F_{MSY} from $F_{1990-2010}$ primarily involves a reduction in the beam and otter trawling effort and is predicted to reduce risk without affecting yield (Figure 5). This is confirmed by an analysis that matches annual estimates of F for the period 1970–2014 to the relevant simulations. Risk levels were predicted to have increased from 1970 into the 1990s, before decreasing to low values by 2010 (Figure 6). However, the economic value of the catch remained relatively stable despite the changes in risk.

Finally, we ran simulations with idealized fleets, in which no individual species could be caught by more than one fleet (Table 1). In general terms, idealized fishing has the effect of increasing the risks associated with beam trawling for no extra reward, but improves the yield from the otter fishery while reducing risk (Figure 7a and b). Overall, the optimum combination of mixed

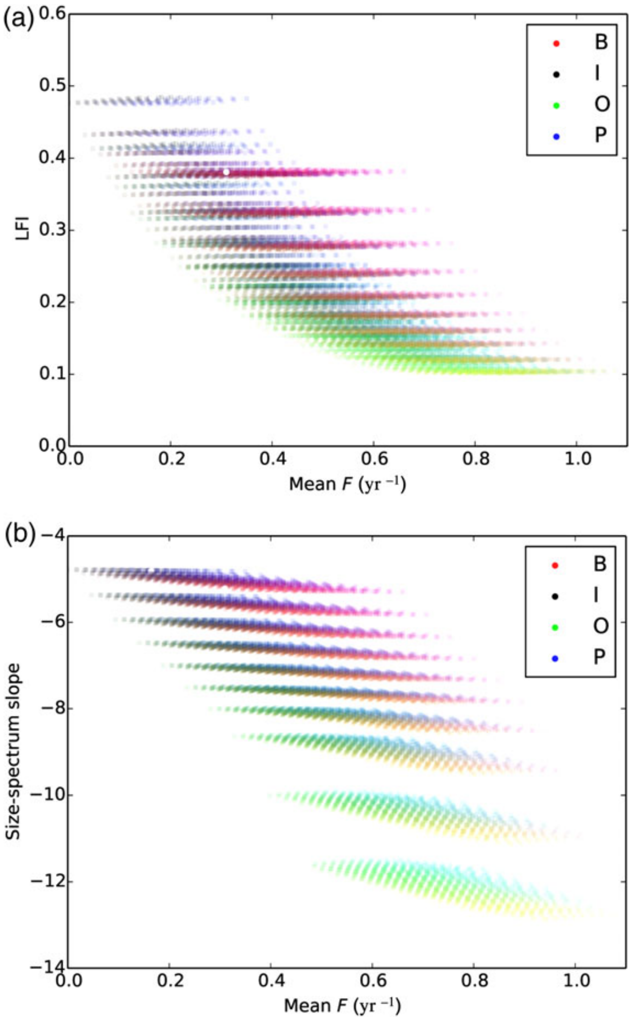

Figure 2. Relationship between mean fishing mortality and ensemble median values of (a) the LFI and (b) the SSS for all combinations of all levels of effort. Fleet codes: B, beam trawl (and red dots); I, industrial (black); O, otter (green); and P, pelagic (blue). Composite colours represent scenarios where more than one fleet is fishing, for example, yellow for beam and otter effort, purple for beam and pelagic effort (see Supplementary Figure S5).

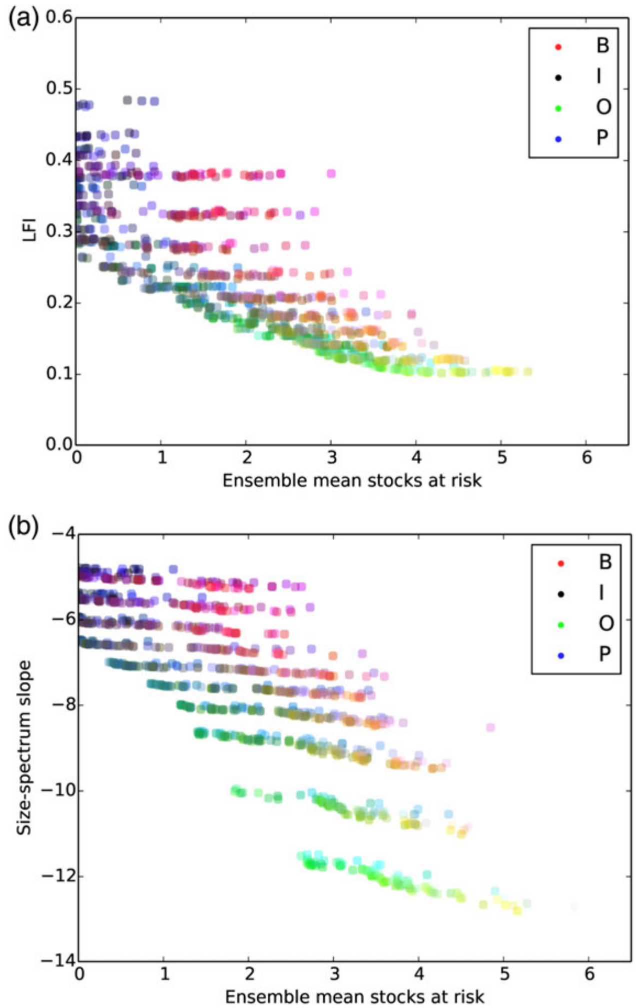

Figure 3. Relationship between the ensemble mean number of stocks at risk and median values for (a) the LFI and (b) the SSS for all combinations of all levels of effort. Fleet codes: B, beam trawl (and red dots); I, industrial (black); O, otter (green) and P, pelagic (blue). Composite colours represent scenarios where more than one fleet is fishing, for example, yellow for beam and otter effort, purple for beam and pelagic effort (see Supplementary Figure S5).

otter/pelagic effort is unchanged by idealized fishing, but the highest yield that can be achieved without risk increases by ~5%.

Discussion

The demand for management advice on mixed and multispecies fisheries creates many challenges for the assessment community. Existing efforts to address these challenges have often focused on 'mixed' and 'multispecies' issues independently. We have shown that these challenges can be addressed simultaneously by including technical and multispecies interactions in the same modelling framework. Thus our approach can be used to predict the possible long-term effects of different fleet fishing strategies on (i) fisheries yield and value, (ii) potential risks to individual stocks, (iii) trade-offs between risks and yield, and (iv) values of fish community indicators. Such predictions can help to inform the development of management plans and identify options that would help to meet targets for both fisheries and environmental management. Our work is timely because a number of different models, which each account for mixed-fishery or ecological interactions to differing extents, are now being used for evaluation of the biological and economic impacts of alternate long-term management plans for North Sea stocks (STECF, 2015).

Results showed that technical and multispecies interactions linked to beam and otter trawl fleets had the strongest effects on fisheries yield and value, risk of stock collapse, and fish community indicators. Consequently, understanding and managing the trade-offs involving these fleets is central to meeting management objectives in North Sea mixed-fisheries. The range of effort combinations where all 21 stocks had a <5% risk of depletion <10% B_0 was relatively small, and these produced 70% of the maximum total value. When all stocks were fished at F_{MSY} the risk to individual stocks was much reduced in relation to historic risks in the North Sea, but with relatively small reductions in yield or value. However, since we present equilibrium results there will still be a transition

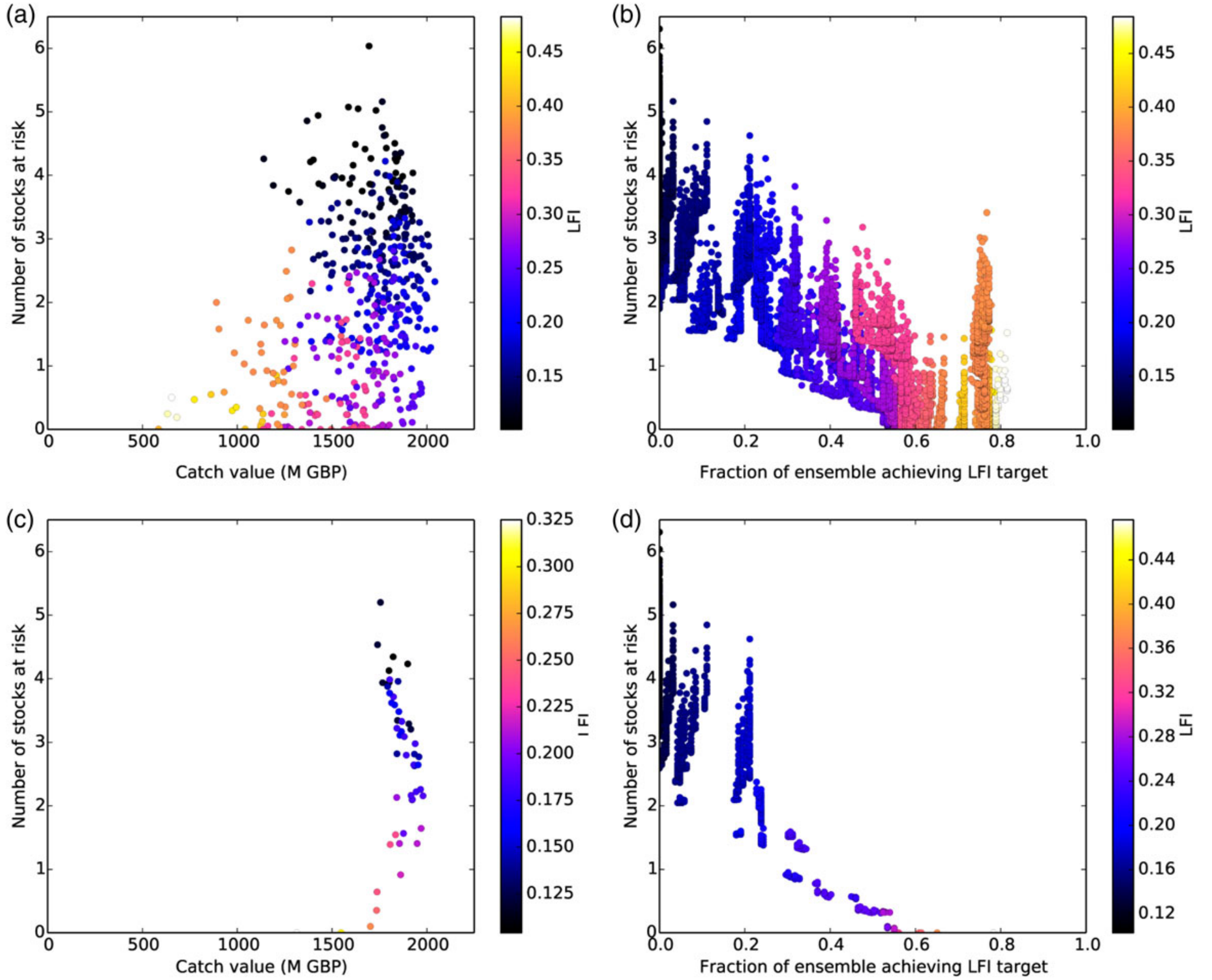

Figure 4. Relationships between (a) the value of yield from the fishery, (b) the fraction of the ensemble meeting a LFI target of 0.3 and the ensemble mean number of stocks at risk, (c) the value of yield from the fishery, and (d) the fraction of the ensemble meeting the LFI target of 0.3 and the ensemble mean number of stocks at risk. Results in (a) and (b) are for all fleet fishing scenarios, while in (c) and (d) consideration is restricted to cases where the relative effort of any one fleet lies within a factor of 2 of the effort in all other fleets.

cost of moving towards F_{MSY} and additional risks from climate and environmental effects on year-to-year stock dynamics. Changes to our results when we assumed idealized fleets were rather small, suggesting that the mix of species taken within fleets led to greater challenges for management than the sharing of species among fleets. This result applies solely to the fleet resolution we considered and would need to be revisited if other resolutions were considered.

In contrast to the approach of Thorpe *et al.* (2015), the present analysis focused on the effects of fleet-specific variations in *F*. This provides more insight into the performance of fish community indicators. In Thorpe *et al.* (2015), the SSS provided greater power than the LFI to detect changes in community-wide *F*. However, for the wider range of fishing scenarios in the present analysis, the two indicators perform similarly with both showing differential sensitivity to the various fishing fleets. Consequently, there will be no simple relationship between overall mortality and indicator values if the contributions of the different fleets to total fishing mortality are changing. One approach to address differential fleet sensitivity might involve the use of fleet-specific indicators. In the absence of such indicators, and given our observations that (i) fishing that poses a high risk to some stocks can coexist with a higher LFI and (ii) lower-risk fishing can coexist with a lower LFI, our results suggest that signals from basic community indicators will not be interpretable without supporting information. This outcome supports the conclusions of Fay *et al.* (2013), who emphasized the importance of expert knowledge of the fishery when interpreting community indicator values.

The value of the LFI was most sensitive to changes in otter trawl effort. This was consistent with modelling results in Spiers *et al.* (2016) who showed an overriding effect of cod, and to a lesser extent saithe, mortality on the LFI. Predicted trends in the LFI following the transition from historic fishing to F_{MSY} were consistent with trends in LFI values estimated from data by Engelhard *et al.* (2015).

The main mechanism for controlling fishing mortality in European fisheries is via single-species TACs. These have well-understood limitations when applied in mixed-fisheries (Ulrich *et al.*, 2011). Policy changes in the most recent iteration of the CFP (Article 15 in EU (2013)), that require the landing and counting against quota of all commercial species catches, are likely to

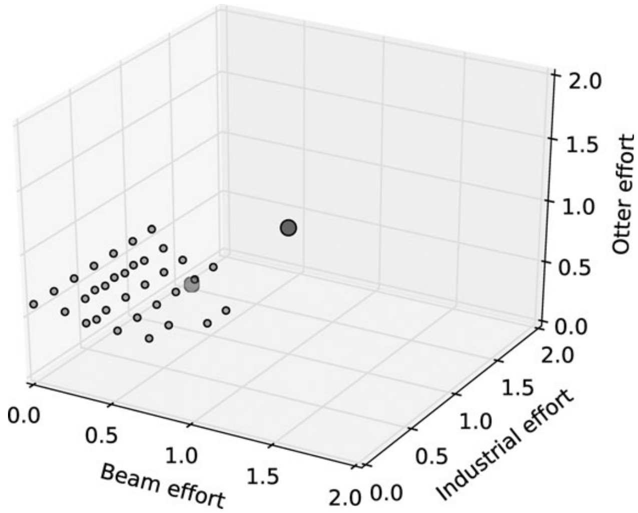

Figure 5. Combinations of fleet effort that result in no stocks being at risk and yields >75% of the total maximum yield. The black point indicates fishing all stocks at F_{HIST} and the grey point at F_{MSY}.

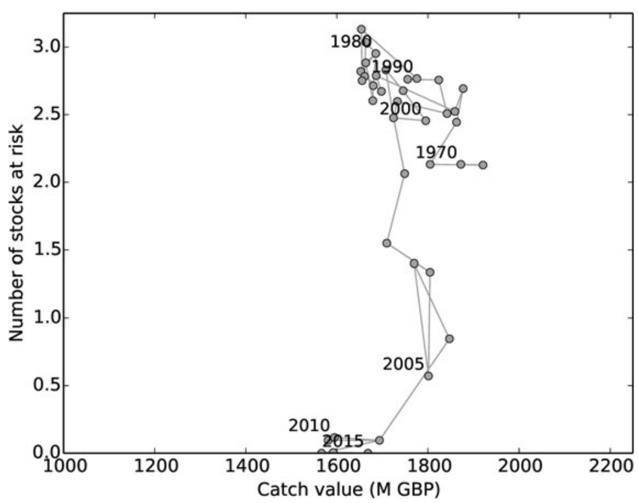

Figure 6. Simulated catch value and level of risk for the period 1970–2015 (2015 figure based on estimated F).

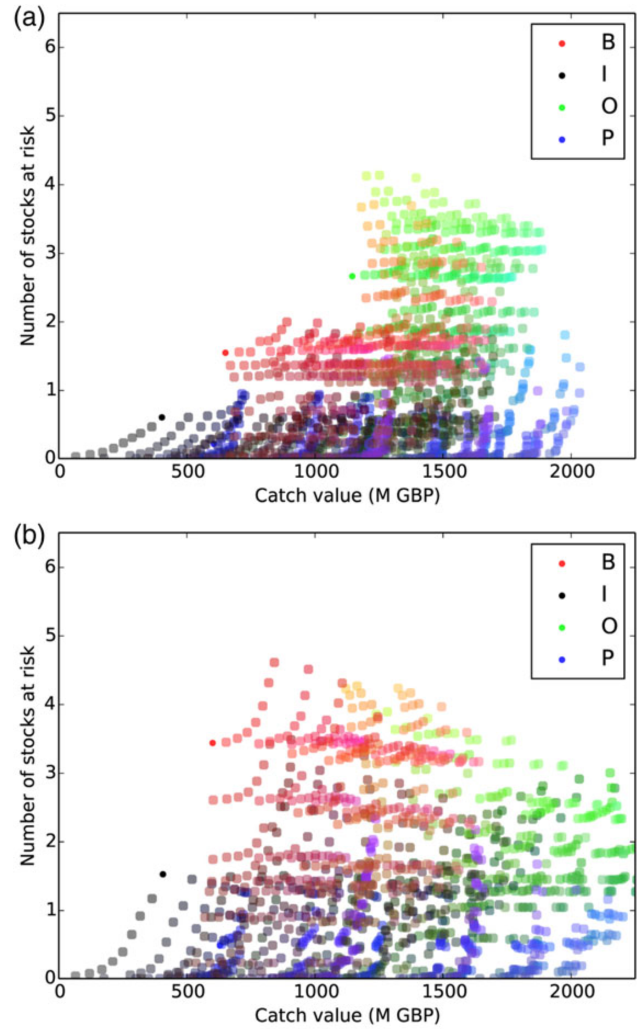

Figure 7. Relationship between the value of yield and the number of stocks at risk for (a) historic and (b) idealized fleets. Fleet codes: B, beam trawl (and red dots); I, industrial (black); O, otter (green); and P, pelagic (blue). Composite colours represent scenarios where more than one fleet is fishing, for example, yellow for beam and otter effort, purple for beam and pelagic effort (see Supplementary Figure S5).

exacerbate management challenges as stocks in poor status limit the fishing opportunities for healthier stocks. The ICES mixed-fishery advice (ICES, 2012a,b) is intended to quantify such effects using a detailed model of fleet/métier activity to apportion catch and thus evaluate short-term TAC management. Our analysis is complementary to the ICES mixed-fishery advice as we consider both multispecies and mixed-fishery interactions over a longer timescale. We assume all catch is landed and explore the long-term relationships between yield, value, and species risk implied in a mixed-fishery and multispecies context. Consequently, our findings highlight trade-offs in the fishery system and management pressures that are likely to result from efforts to balance the social, economic and environmental objectives of the CFP.

Our findings are subject to several caveats, but most could be addressed with further work as our approach is tailored to meet specific and emerging advisory needs. First, the model does not incorporate food-dependent growth, so we may underestimate the risks associated with fishing industrial species that are consumed by species feeding at higher trophic levels. Second, predation by seabirds and mammals that may compete with fishing fleets is ignored. Third, the fleet definitions were highly simplified to support concise and transparent presentation of results, but more highly resolved definitions would often be needed to support management advice. Fourth, we only addressed parameter uncertainty and not uncertainty about the functional forms assumed in the model. Fifth, the most sensitive species taken in these fisheries, such as larger skates and rays, are not included in the model, and risks to these species from the demersal trawl fisheries are expected to be relatively high (e.g. Walker and Hislop, 1998).

In conclusion, the main benefit of our approach is that it directly addresses the growing demand for assessment methods that treat mixed and multispecies fisheries collectively and take account of trade-offs between weight of yield, value of yield, state of stocks, and state of the environment. The approach uses data that are collected routinely to support existing methods of assessment and management, and can support strategic advice on the effects of changing fleet configurations and the management options to meet fisheries and environmental targets. By presenting risks associated with these options, the approach is also well suited to exploring the consequences of changes in risk aversion of managers.

Supplementary data

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

Acknowledgements

This work was funded by the UK Department for Environment, Food, and Rural Affairs (Projects MF1225 and MA016A). We thank Paul Marshal from IFREMER and an anonymous referee for valuable comments that helped us to improve the analyses and presentation.

Funding

Funding to pay the Open Access publication charges for this article was provided by the UK Department for Environment Food and Rural Affairs through Project MF1225.

References

Andersen, K. H., and Rice, J. C. 2010. Direct and indirect community effects of rebuilding plans. ICES Journal of Marine Science, 67: 1980–1988.

Blanchard, J. L., Andersen, K. H., Scott, F., Hintzen, N. T., Piet, G., and Jennings, S. 2014. Evaluating targets and trade-offs among fisheries and conservation objectives using a multispecies size spectrum model. Journal of Applied Ecology, 51: 612–622.

Butterworth, D. S., and Plaganyi, E. E. 2004. A brief introduction to some multi-species/ecosystem modelling approaches in the context of their possible application in the management of South African Fisheries. African Journal of Marine Science, 26: 53–61.

Collie, J. S., and Gislason, H. 2001. Biological reference points for fish stocks in a multispecies context. Canadian Journal of Fisheries and Aquatic Sciences, 58: 2167–2176.

Christensen, V., and Walters, C. J. 2004. Ecopath with Ecosym: methods, capabilities and limitations. Ecol. Model, 172: 109–139.

Engelhard, G. E., Lynam, C. P., Garcia-Carreras, B., Dolder, P. J., and Mackinson, S. 2015. Effort reduction and the large fish indicator: spatial trends reveal positive impacts of recent European fleet reduction schemes. Environment Conservation, 42: 227–236

EU. 2008. Directive 2008/56/EC of the European Parliament and of the Council of 17 June 2008 establishing a framework for community action in the field of marine environmental policy (Marine Strategy Framework Directive). Official Journal of the European Union, L164: 19–40.

EU. 2010. Commission decision of 1 September 2010 on criteria and methodological standards on good environmental status of marine waters. Official Journal of the European Union, L232: 14–24.

EU. 2013. Regulation (EU) No 1380/2013 of the European Parliament and of the Council of 11 December 2013 on the Common Fisheries Policy, amending Council Regulations (EC) No 1954/2003 and (EC) No 1224/2009 and repealing Council Regulations (EC) No 2371/2002 and (EC) No 639/2004 and Council Decision 2004/585/EC. Official Journal of the European Union, L354: 22–61.

Fay, G., Large, S. I., Link, J. S., and Gamble, R. J. 2013. Testing systemic fishing responses with ecosystem indicators. Ecological Modelling, 265: 45–55.

Fulton, E. A., Link, J. S., Kaplan, I. C., Savina-Rolland, M., Johnson, P., Ainsworth, C., Horne, P., *et al.* 2011. Lessons in modelling and management of marine ecosystems: the Atlantis experience. Fish and Fisheries, 12: 171–188.

Fulton, E. A., Smith, A. D. M., and Johnson, C. R. 2003. Effect of complexity on marine ecosystem models. Marine Ecology Progress Series, 253: 1–16.

Gaichas, S. K., Odell, G., Aydin, K. Y., and Francis, R. I. C. C. 2012. Beyond the defaults: functional response parameter space and ecosystem-level fishing thresholds in dynamic food web model simulations. Canadian Journal of Fisheries and Aquatic Sciences, 69: 2077–2094.

Gårdmark, M., Lindegren, M., Neuenfeldt, S., Blenckner, T., Heikinheimo, O., Müller-Karulis, B., Niiranen, S., *et al.* 2013. Biological ensemble modelling to evaluate potential futures of living marine resources. Ecological Applications, 23: 742–754.

Gislason, H. 1999. Single and multispecies reference points for Baltic fish stocks. ICES Journal of Marine Science, 56: 571–583.

Gray, T., Hatchard, J., Daw, T., and Stead, S. 2008. New cod war of words: "cod is god" versus "sod the cod": two opposed discourses on the North Sea Cod Recovery Programme. Fisheries Research, 93: 1–7.

Greenstreet, S. P. R., Rogers, S. I., Rice, J. C., Piet, G. J., Guirey, E. J., Fraser, H. M., and Fryer, R. J. 2011. Development of the EcoQO for the North Sea fish community. ICES Journal of Marine Science, 68: 1–11.

Greenstreet, S. P. R., Rossberg, A. G., Fox, C. J., Le Quesne, W. J. F., Blasdale, T., Boulcott, P., Mitchell, I., *et al.* 2012. Demersal fish biodiversity: species-level indicators and trends-based targets for the Marine Strategy Framework Directive.—ICES Journal of Marine Science, 69: 1789–1801

Hall, S. J., Collie, J. S., Duplisea, D. E., Jennings, S., Bravington, M., and Link, J. 2006. A length-based multispecies model for evaluating community responses to fishing. Canadian Journal of Fisheries and Aquatic Sciences, 63: 1344–1359.

Hill, S. L., Watters, G. M., Punt, A. E., McAllister, M. K., Quéré, C. L., and Turner, J. 2007. Model uncertainty in the ecosystem approach to fisheries. Fish and Fisheries, 8: 315–336.

Hoff, A., Frost, H., Ulrich, C., Damalas, D., Maravelias, C. D., Goti, L., and Santurtún, M. 2010. Economic effort management in multispecies fisheries: the FcubEcon model. ICES Journal of Marine Science, 67: 1802–1810.

ICES. 2012a. Report of the Working Group on Mixed Fisheries Advice for the North Sea (WGMIXFISH), 21–25 May 2012, ICES Headquarters, Copenhagen. ICES CM2012/ACOM:22. 94 pp.

ICES. 2012b. Report of the Working Group on the Assessment of Demersal Stocks in the North Sea and Skagerrak (WGNSSK). ICES CM2012/ACOM:13, ICES Headquarters, Copenhagen.

ICES. 2012c. Manual for the International Bottom Trawl Surveys. Series of ICES Survey Protocols. SiSP 1-IBTS VIII. ICES, Copenhagen. 68 pp.

ICES. 2012d. Report of the ICES Advisory Committee 2012, Book 6: North Sea. ICES, Copenhagen. 443 pp.

ICES. 2013. Multispecies considerations for the North Sea stocks. ICES Advice 2013, Book 6, Copenhagen.

ICES. 2014a. ICES Working Group on Mixed Fisheries Advice for the North Sea. ICES, Copenhagen.

ICES. 2014b. Mixed-fisheries advice for Subarea IV (North Sea) and Divisions IIIa North (Skagerrak) and VIId (Eastern Channel). ICES Advice 2014, Book 6, Copenhagen.

Link, J. S., Ihde, T. F., Harvey, C. J., Gaichas, S. K., Field, J. C., Brodziak, J. K. T., Townsend, H. M., *et al.* 2012. Dealing with uncertainty in ecosystem models: the paradox of use for living marine resource management. Progress in Oceanography, 102: 102–114.

Lynam, C. P., and Mackinson, S. 2015. How will fisheries management measures contribute towards the attainment of good environmental status for the North Sea ecosystem? Global Ecology and Conservation, 4: 160–175.

Murawski, S. A. 1991. Can we manage our multispecies fisheries? Fisheries, 16: 5–13.

Pope, J. G. 1979. A modified cohort analysis in which constant mortality is replaced by estimates of predation levels. ICES Document CM 1979/H: 16.

Pope, J. G. 1991. The ICES Multispecies Assessment Working Group: evolution, insights, and future problems. ICES Marine Science Symposia, 193: 22–33.

Pope, J. G., MacDonald, D. S., Daan, N., Reynolds, J. D., and Jennings, S. 2000. Gauging the impact of fishing mortality on non-target species. ICES Journal of Marine Science, 57: 689–696.

Pope, J. G., Rice, J. C., Daan, N., Jennings, S., and Gislason, H. 2006. Modelling an exploited marine fish community with 15 parameters—results from a simple size-based model. ICES Journal of Marine Science, 63: 1029–1044.

Reeves, S. A., Marchal, P., Mardle, S., Pascoe, S., Prellezo, R., Thébaud, O., and Travers, M. 2009. From fish to fisheries: the changing focus of management advice. *In* Advances in Fisheries Science 50 Years on from Beverton and Holt, pp. 135–154. Ed. by A. Payne, J. Cotter, and T. Potter. Blackwell Publishing, Oxford. ISBN 978-1-4051-7083-3. xxi, 546 pp.

Rice, J. C. 2011. Managing fisheries well: delivering the promises of an ecosystem approach. Fish and Fisheries, 12: 209–231.

Rindorf, A., Schmidt, J., Bogstad, B., Reeves, S., and Walther, Y. 2013. A framework for multispecies assessment and management. ICES/Nordic Council of Ministers. ISBN 978-92-893-2584-4.

Rochet, M. J., Collie, J. S., Jennings, S., and Hall, S. J. 2011. Does selective fishing conserve community biodiversity? Predictions from a length-based multispecies model. Canadian Journal of Fisheries and Aquatic Sciences, 68: 469–486.

Shin, Y-J., Rochet, M-J., Jennings, S., Field, J., and Gislason, H. 2005. Using size-based indicators to evaluate the ecosystem effects of fishing. ICES Journal of Marine Science, 62: 384–396.

Sparre, P. 1991. Introduction to multispecies virtual population analysis. ICES Marine Science Symposia, 193: 12–21.

Spiers, D. C., Greenstreet, S. P. R., and Heath, M. R. 2016. Modelling the effects of fishing on the North Sea fish community size composition. Ecological Modelling, 321: 35–45.

STECF. 2014. Evaluation of Fishing Effort Regimes in European Waters – Part 2 (STECF-14-20). STECF, Copenhagen. ISBN 978-92-79-44712-9.

STECF. 2015. Evaluation of the multi-annual plan for the North Sea demersal stocks (STECF 15-04). STECF, Ispra. ISBN 978-92-79-48165-9.

Thøgersen, T., Hoff, A., and Frost, H. 2012. Linking effort and fishing mortality in a mixed fisheries model: comparing linear versus non-linear assumptions. Fisheries Research, 127–128: 9–17.

Thorpe, R. B., Le Quesne, W. J. F., Luxford, F., Collie, J. S., and Jennings, S. 2015. Evaluation and management implications of uncertainty in a multispecies size-structured model of population and community responses to fishing. Journal of Methods in Ecology and Evolution, 6: 49–58.

Ulrich, C., Reeves, S. A., Vermard, Y., Holmes, S. J., and Vanhee, W. 2011. Reconciling single-species TACs in the North Sea demersal fisheries using the Fcube mixed-fisheries advice framework. ICES Journal of Marine Science, 68: 1535–1547.

Vinther, M., Reeves, S. A., and Patterson, K. R. 2004. From single-species advice to mixed species management: taking the next step. ICES Journal of Marine Science, 61: 1398–1409.

Walker, P. A., and Hislop, J. R. G. 1998. Sensitive skates or resilient rays? Spatial and temporal shifts in ray species composition in the central and north-western North Sea between 1930 and the present day. ICES Journal of Marine Science, 55: 392–402.

Handling editor: Jörn Schmidt