Modelling the direct impact of bottom trawling on the North Sea fish community to derive estimates of fishing mortality for non-target fish species

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This study introduces a spatially explicit model that combines abundance data for all the main fish species in the demersal North Sea fish community with international effort data and estimates of gear-, species-, and size-dependent catch efficiency to determine the mortality of non-target fish species caused by bottom trawl fisheries and its spatial variation. Where necessary information was lacking, assumptions were made, and a sensitivity analysis performed to examine the impact of these issues on model results. Model outcomes were validated using international landings and discard data for five target species: cod, haddock, whiting, sole, and plaice. This showed that depending on its configuration, the model could reproduce recorded landings and discards of these species reasonably well. This suggests that the model could be used to simulate rates of fishing mortality for non-target fish species, for which few data are currently available. Sensitivity analyses revealed that model outcomes were most strongly influenced by the estimates of gear catch efficiency and the extent to which the distributions of fishing effort and each species overlapped. Better data for these processes would enhance the contribution that this type of model could make in supporting an ecosystem approach to fisheries management.

Keywords: catch efficiency, elasmobranchs, fishing effort, fishing impact, non-target species, spatial distribution.

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Introduction

Political commitments to ecosystem-based fisheries management (EBFM) are increasing (FAO, 2003). EBFM is intended to ensure that the planning, development, and management of fisheries will meet social and economic needs, without jeopardizing the options for future generations to benefit from the full range of goods and services provided by marine ecosystems (FAO, 2003). EBFM requires managers to take account of how fisheries impact a wide range of marine ecosystem components when setting ecosystem their objectives (Heslenfeld and Enserink, 2008). To achieve such objectives, the mechanistic relationships between the state of these components or attributes and one or more manageable anthropogenic activities need to be understood (Jennings, 2005). For scientists charged with the provision of advice in support of EBFM, determining the theoretical, mechanistic link between state and so-called pressure indicators often poses the greatest challenge (Greenstreet, 2008). In the past, changes in indices characterizing various aspects of the state of North Sea demersal fish or benthic invertebrate communities have generally been related to changes in the levels of fishing activity (Greenstreet and Hall, 1996; Greenstreet *et al.*, 1999; Jennings and Cotter, 1999; Piet and Jennings, 2005; Greenstreet and Rogers, 2006; Hiddink *et al.*, 2006; Tillin *et al.*, 2006). However, the underlying relationship between fishing activity and these reported changes in the state of fish communities is far from clear. Most theoretical models assume that population and community dynamics are governed primarily by four simple processes: birth, death, immigration, and emigration (MacArthur and Wilson, 1967; May, 1976). Anthropogenic activities affect populations or communities through their modulation of one or more of these processes. Consequently, in seeking to model how anthropogenic activities determine the state of biological systems, the activities are best quantified as the extent to which one or other of these processes, most often death, is altered by the activity in question.

Recent EU initiatives to encourage the systematic collection of discard data have further enhanced our overall understanding of the mortality caused by fishing among notably the commercial species. However, commercially exploited species make up a relatively small fraction of the 200 or more fish species inhabiting the North Sea (Yang, 1982), and information on the mortality suffered by non-target species is sparse. Considering the political importance placed on conserving marine biodiversity (Greenstreet, 2008), and the consequent need to protect vulnerable non-target species such as elasmobranchs (Walker and Heessen, 1996; Walker and Hislop, 1998; Greenstreet and Rogers, 2000), such shortcomings need to be addressed urgently, particularly if the time-scales imposed by the recent EU Marine Strategy Framework Directive are to be met. However, it is unlikely that the resources that would allow accurate direct estimation of fishing mortality rates in non-target species populations will ever be available; indeed, it is questionable whether such estimation is even possible. Under these circumstances, an alternative approach must be sought.

Two separate studies (Piet *et al.*, 2000; Pope *et al.*, 2000), both using variants of a "swept-area" approach, have suggested that rates of fishing mortality in different components of the marine ecosystem might be modelled from data that appropriately quantify spatial and temporal variation in the levels of fishing activity (Jennings and Cotter, 1999), along with abundance of the biota in question obtained from surveys and stock assessments (Künitzer *et al.*, 1992; Knijn *et al.*, 1993). The role of such modelling approaches in deriving the most effective indicators of pressure for use within the pressure–state–response frameworks as part of an EBFM has been defined by Piet *et al.* (2007). Four levels of pressure indicator were described. The first three incorporate more information on fishing vessels, time spent fishing, and the dimensions of the fishing gears being used to derive increasingly complex and detailed quantitative descriptions of the fishing activity taking place. At the fourth level, an estimate of the population or community level rate of fishing mortality is derived by incorporating estimates of the mortality suffered per fishing event (Collie *et al.*, 2000; Kaiser *et al.*, 2006). Piet *et al.* (2000, 2007) demonstrate their approach in developing estimates of fishing mortality for the benthic invertebrate community, and Pope *et al.* (2000) modelled rates of fishing mortality for some non-target fish species in the North Sea.

Here, we adopted and slightly modified the above approaches to determine the direct mortality caused by fishing to members of the North Sea demersal fish assemblage. Our modifications involve, first, the use of "true" estimates of spatial variation in fish abundance that take account of catchability in the gears used in the groundfish surveys (Fraser *et al.*, 2007). Second, we assumed gear-, species-, and size-dependent variable catch efficiencies in the two major fisheries, otter trawl and beam trawl, operating in the area. We then performed a sensitivity analysis to examine the extent to which our mortality estimates were affected by the various assumptions made. Finally, we validated the model by comparing model output with estimates of landings and discards of the main commercial demersal species derived from sampling programmes. Those analyses not only provided best estimates of fishing-induced mortality for the main fish species in the North Sea, but also insight as to which factors influenced those estimates most and which should therefore be considered for further research.

Material and methods

This study combines (i) abundance estimates of all the main fish species in the demersal North Sea fish community, (ii) international effort data, along with (iii) the estimates of species- and size-dependent catch efficiency in different fishing gears to construct a spatially explicit framework that provides estimates of fishing mortality for all demersal fish species sampled adequately in the international bottom trawl survey (IBTS) over the period 1998–2004.

Fish abundance

Estimates of spatial variance in North Sea demersal fish abundance were derived from two research vessel groundfish survey time-series: the third quarter IBTS and the Dutch beam trawl survey (DBTS). These surveys were considered to be complementary in that they primarily target different components of the demersal fish community. The IBTS uses a Grande Ouverture Verticale (GOV) otter trawl, which is more efficient at catching roundfish, whereas the DBTS uses a beam trawl, which is more efficient at catching flatfish. We analysed data for the third quarter collected since 1998, because from that year on, the gear used in the IBTS and the trawl duration were standardized to the GOV and 30 min, respectively. Observed survey-density data were raised to estimates of absolute abundance that took account of catchability in the beam trawl and the GOV, following methods developed by Sparholt (1990) and Fraser *et al.* (2007). These methods determine raising factors for the two survey gears for a number of commercial species per size class, by dividing the overall survey-based abundance estimates for the whole North Sea by the estimate of the total abundance in that area based on the stock assessments. The survey-based abundance estimates were then derived by multiplying the mean catch per area trawled in each ICES rectangle by the total area of that rectangle. For this study, we applied a method of arithmetic mean densities over all hauls in one ICES rectangle rather than geometric means to determine the raising factors for three assessed roundfish species, cod (*Gadus morhua*), haddock (*Melanogrammus aeglefinus*), and whiting (*Merlangius merlangus*), and two assessed flatfish species, plaice (*Pleuronectes platessa*) and sole (*Solea solea*). For the remaining non-assessed species, we adopted the catch ratios per 5-cm length class as in Fraser *et al.* (2007). As flatfish are not sampled adequately by the IBTS, we chose to base our estimates of absolute abundance on the DBTS data. We then followed Sparholt's (1990) procedure, assigning each flatfish species into one of two categories depending on whether they were considered sole-like or plaice-like. Finally, because the DBTS covers a smaller area than the IBTS, we based the absolute abundance estimates for flatfish in those ICES rectangles not covered by the DBTS on the IBTS, applying the appropriate raising factor for each gear.

Fish numbers-at-age for the five commercial species used to calculate the raising factors were available from VPA (ICES, 2007). As the numbers at age 0 were not available for cod, plaice, sole, or whiting, the age-1 raising factor was used for that age class. The model is based on weights, so the numbers were converted to biomass according to the formula

$$B_{s,l} = 10\,N_{s,l}L^3, \tag{1}$$

where $B_{s,\,l}$ is the biomass (kg) per species per length class, $N_{s,\,l}$ the numbers per species per length class, and L the length (m).

Fishing effort

An international fishing effort database was recently compiled by Greenstreet *et al.* (2007), including data for all nations with significant fishing interests in the North Sea, for the period 1997–2004. The database holds information on the numbers of hours fishing in each ICES rectangle in each year by four main gear categories: otter trawl directed at fish, otter trawl directed at Norway lobster (*Nephrops norvegicus*), beam trawl, and seine-net. For our purpose, we distinguished two major fisheries targeting fish: an otter trawl fishery targeting roundfish mainly operating in the northwestern North Sea and a beam trawl fishery targeting flatfish operating mainly in the southeastern North Sea. For these two fisheries, international effort data were used to determine trawl frequency in each ICES rectangle ($F_{g,i}$) in each year between

Table 1. Characteristics of the fisheries that determine their catch efficiency.

Gear	Engine power (kW)	Speed (km h^{-1})	Effective width (m)	Effective height (m)	Mesh size (mm)
Beam trawl	<221	7.8	8 (2× 4)	1	80
Beam trawl	>221	12.4	24 (2× 12)	1	80/100
Otter trawl	All	5.0	87	5–6	100

For beam trawls, the effective width was measured between the shoes, and for otter trawls between the boards.

1998 and 2004 from

$$F_{g,i} = \frac{W_g E_{g,i} S_g}{A_i}, \tag{2}$$

where W_g is the effective width of fishing gear g (m), $E_{g,i}$ the fishing effort in hours (h) in each ICES rectangle i, S_g the speed (m h^{-1}) that the gear is normally towed, and A_i the surface area of each ICES rectangle i (m^2). Parameter values used for otter trawls were based on Kynoch (1997), Kynoch and Penny (2006), and unpublished information recorded by observers placed on fishing vessels (unpublished data, IMARES, IJmuiden, the Netherlands, and Marine Laboratory, Aberdeen, UK), whereas those for beam trawls were obtained from Piet *et al.* (2007; Table 1). As the international effort does not distinguish between large vessels (>221 kW) and small "eurocutters" (<221 kW), the effort (in h) for the beam trawl was multiplied by the ratio between the hours fished by those two categories based on Dutch national logbook data.

Catch efficiency

We define catch efficiency as the proportion of fish present in the area swept by the gear that is retained in the codend at the end of any particular fishing operation. For beam trawls, the effective width of the gear is the actual width of the trawl (Table 1), but for otter trawls, the effective width is assumed to be the distance between the otter boards because those, along with the sweeps, herd fish into the path of the net. The interaction between fish and bottom trawls is a complex issue determined by fish behaviour in relation to gear characteristics, making the catch efficiency of a gear hard to quantify (Wardle, 1988; Dickson, 1993a, b). Based on the available literature (Engås and Godø, 1989a, b; Weinberg *et al.*, 2002), we developed a conceptual framework in which catch efficiency was determined by four factors:

(i) the positioning of fish in the water column;
(ii) the herding of fish by the gear;
(iii) the escapement of fish below the footrope of the net;
(iv) the retention of fish in the net.

Many other factors may affect catch efficiency, e.g. vessel noise (Dickson, 1993a, b), visibility, fishing speed, density-dependent catchability, diel variation, and mesh shape (Robertson and Stewart, 1988; Wardle, 1988; Godø *et al.*, 1999; Weinberg *et al.*, 2002; Benoit and Swain, 2003). However, the absence of quantitative data prevented us from incorporating these factors.

Fish position in the water column relative to the gear determines the likelihood that they will enter the mouth of the net. We assumed that 80% of the roundfish failed to escape over the headline of the otter trawl, but because of the much lower vertical opening of a beam trawl (Table 1), we assumed that just 20% of roundfish passed below the beam. Flatfish were assumed to be unable to escape over the top of either gear. We assumed that hardly any fish escaped sideways (95% retention), resulting in positioning values of, respectively, 76 and 19% (Table 2). Not all fish species in the area swept between the otter boards are herded towards the mouth of the net (Wardle, 1986; Engås and Godø, 1989a, b; Dickson, 1993a, b; Ramm and Xiao, 1995). Comparisons of catch rates between gears with different sweep lengths have demonstrated this herding effect. Catch rates of cod and haddock, particularly of larger fish, increase significantly as the distance between the otter boards increases (Engås and Godø, 1989a, b). Although herding may be assumed to be independent of fish length (Ramm and Xiao, 1995), we assumed that 30% of small roundfish ($\leq$29.5 cm) and 75% of large roundfish ($>$29.5 cm) in the path between the otter boards are herded into the net. There is little information available on herding in flatfish. According to Winger *et al.* (1999), larger flatfish should be capable of reaching the net opening, but they assumed a much slower towing speed (~2 knots) than is customary for the North Sea fishing fleet. Because flatfish endurance decreases rapidly with increasing swimming speed (Wardle, 1988), we assumed that just 40% of flatfish are herded into the net. For beam trawls, we assumed that herding is not an issue, because the gear is not equipped with otter boards and sweeps to initiate it. As we only considered the width of the beam trawl itself and not the distance between the otter boards as we did with the otter trawl, the herding factor for beam trawls was set to 1.0 for both fish types. The proportion of fish passing below the footrope of an otter trawl depends on species, size, fishing speed, and gear construction (Engås and Godø, 1989a, b; Dahm, 2000; Weinberg *et al.*, 2002). For an otter trawl, we therefore assumed that 5% of roundfish, 50% of smaller ($<$25 cm) flatfish, and 15% of larger ($\geq$25 cm) flatfish pass under the footrope, whereas for the beam trawl, we assumed escape rates below the beam trawl chain matrix and foot chain to be zero for both roundfish and flatfish. The values for water column positioning, herding, and footrope escape (small/large fish) were then multiplied to give initial catch efficiencies for these three processes (Table 2). As anticipated, this confirmed that the beam trawl was more selective for flatfish than the otter trawl (0.95 vs. 0.19 for small flatfish, and 0.32 for large flatfish), but less selective for large roundfish (0.19 vs. 0.54 for roundfish).

Table 2. Factors used to calculate the catch efficiency for beam trawl (BT) and otter trawl (OT) and different fish types, demersal roundfish (R) and demersal flatfish (F).

			Factor			
Gear	Fish type	Fish length (cm)	Positioning	Herding	Footrope	Initial overall
BT	R	All	0.19	1	1	0.19
BT	F	All	0.95	1	1	0.95
OT	R	<29.5	0.76	0.3	0.95	0.22
OT	R	>29.5	0.76	0.75	0.95	0.54
OT	F	<25	0.95	0.4	0.5	0.19
OT	F	>25	0.95	0.4	0.85	0.32

The initial overall factor was calculated by multiplying the positioning, herding, and footrope factors.

Fish are generally thought to escape mainly from the codend of the gear (Millar and Fryer, 1999), so most studies on gear selectivity have tended to concentrate on codend selection (Wileman *et al.*, 1996). Gear characteristics such as mesh size, codend extension length, codend diameter, and mesh shape have a significant influence on the selectivity of fishing gears (van Beek *et al.*, 1981, 1983; Robertson and Stewart, 1988; Reeves *et al.*, 1992; Zuur *et al.*, 2001). For our purposes, the proportion of fish of a given length retained in the net is calculated as a function of mesh size, using codend selectivity data summarized from several gear selectivity studies carried out over a period of more than 30 years (Wileman, 1992; Wileman *et al.*, 1996). A logistic curve is used to describe the relationship between the length of a fish and the proportion of fish entering the net that is retained in the codend (Casey, 1996):

$$S_L = [3^{(L_{50}-(L+\Delta L/2))/(L_{50}-L_{25})} + 1]^{-1}, \tag{3}$$

where S_L is the proportion of fish of length L and class width ΔL (equal to 1 for centimetre classes) that is retained, and L_{50} and L_{25} are the lengths (cm) at which 50 and 25%, respectively, of the fish entering the net are retained. L_{50} and L_{25} are calculated from the selection factor (S_F) and selection range (S_R), together with the mesh size M (cm; Wileman, 1992; Wileman *et al.*, 1996; see Table 3):

$$\begin{aligned} L_{50} &= S_F M \\ L_{25} &= L_{50} - \left(\frac{S_R}{2}\right). \end{aligned} \tag{4}$$

Values for M are those used by the commercial fleets, i.e. 10 cm for otter trawls and 8 cm for beam trawls in the southern North Sea, and 10 cm for beam trawls in the northern North Sea north of 55°N, or north of 56°N if east of 5°E. The initial catch efficiencies derived for the first three escape processes (Table 2) were then multiplied by the codend selectivity to produce the final overall catch efficiency ($P_{r,g,s,l}$) expressed as the proportion of fish at each 1 cm length class (l), of each species (s) in the path of the effective part of the gear (g) that is retained (r) in the codend. From the overall catch efficiency per species, the overall catch efficiencies for roundfish and flatfish were calculated by, respectively, taking the average of all three roundfish species and assuming that the overall catch efficiency of plaice applies for the flatfish species. As sufficient quantitative information to determine codend selectivity was only available for the five commercial species (MacLennan, 1992), the overall catch efficiencies determined for roundfish and flatfish were applied to non-target species after each non-target species had been assigned to the appropriate fish category (roundfish or flatfish; see Table 4).

Mortality rate

The proportion of individuals in any specified ICES rectangle i that escape being caught ($E_{g,i}$) by a specific gear g (otter, o, or beam, b) is calculated from the proportion retained ($R_{g,i}$) and the annual trawling frequency ($F_{g,i}$) of that gear in that specific ICES rectangle i, according to

$$E_{g,i} = (1 - R_{g,i})^{F_{g,i}}. \tag{5}$$

The total catch (C_i) from the two fisheries in specific ICES rectangle i is determined by the proportion ($E_{g,i}$) escaping both gears g (b, beam trawl; o, otter trawl):

$$C_i = B_i(1 - E_{b,i}E_{o,i}), \tag{6}$$

where B_i is the biomass present in the path of the gear, which is determined by the mean biomass per surface area in that rectangle. The fish caught can be divided into landed or discarded fish based on their characterization as commercial or non-target species and, for the former, the minimum landing size. The percentage discarded is calculated as the biomass discarded divided by the biomass caught. The mortality rate is defined as the biomass caught divided by the biomass present.

Sensitivity analysis

For several factors with the potential to affect critically our predicted estimates of fishery-induced mortality, information was sparse. Notable among these were actual data on catch efficiency, understanding of fish dispersion processes, and information regarding the extent to which the spatial distributions of the fishing effort and fish abundance overlapped and varied through the seasons. The impact of these on model outcomes was examined using a sensitivity analysis. To examine the influence of our estimates of catch efficiency, we varied the overall factor, rather than examining each of the processes contributing to catch efficiency separately. The effects of dispersion were mimicked by allowing the remaining fish to redistribute to their initial spatial distribution (i.e. based on the surveys) after a certain interval, yearly, quarterly, monthly, or weekly. With respect to the overlap in distribution between fishing effort and fish abundance, we assumed that throughout the year, fishers follow the seasonally changing spatial distribution of their resource such that they optimize their revenue. However, in our model, we combined fishing effort with a spatial distribution based on an annual average with

Table 3. Gear selectivity parameters for otter trawls and beam trawls, i.e. the lengths (cm) at which 50% (L_{50}) and 25% (L_{25}) are caught, for roundfish (R) and flatfish (F) species.

			Otter trawl mesh 100 mm		Beam trawl mesh 80 mm		Beam trawl mesh 100 mm	
Species	Species code	Fish type	L_{25}	L_{50}	L_{25}	L_{50}	L_{25}	L_{50}
Cod	COD	R	25.95	30	20.3	24	26.3	30
Haddock	HAD	R	27.7	31	21.5	24.8	27.7	31
Whiting	WHI	R	28.35	32	24.5	28	31.5	35
Plaice	PLA	F	24.2	25	15.65	17.6	20.05	22
Sole	SOL	F	29.95	32	23.75	25.6	30.15	32

Table 4. List of the 47 most important (>0.01% of total demersal fish biomass) demersal fish species in the North Sea that together constitute 99.9% of the biomass.

				Mortality (%)		
Latin name	**English name**	**Category**	**Biomass (%)**	**Beam trawl**	**Otter trawl**	**Total**
Agonus cataphractus	Pogge (armed bullhead)	R/NT	0.3	0	0	0
Amblyraja radiata	Starry ray	F/E	3.8	20	17	38
Anarhichas lupus	Wolffish	R/NT	0.0	1	52	53
Anguilla anguilla	European eel	R/NT	0.1	1	58	59
Arnoglossus laterna	Scaldfish	F/NT	0.5	8	0	8
Buglossidium luteum	Solenette	F/NT	0.2	5	0	5
Callionymus lyra	Common dragonet	R/NT	1.0	1	0	2
Callionymus maculatus	Spotted dragonet	R/NT	0.0	0	5	5
Chelidonichthys lucerna	Tub gurnard	R/NT	0.1	13	8	21
Echiichthys vipera	Lesser weever	R/NT	0.2	0	0	0
Enchelyopus cimbrius	Four-bearded rockling	R/NT	0.1	3	1	5
Entelurus aecquoreus	Snake pipefish	R/NT	0.4	0	44	44
Eutrigla gurnardus	Grey gurnard	R/NT	3.3	4	6	10
Gadus morhua	Cod	R/C	4.8	3	42	45
Galeorhinus galeus	Tope	R/E	0.1	9	7	16
Galeus melastomus	Black-mouthed dogfish	R/E	0.0	0	57	57
Glyptocephalus cynoglossus	Witch	F/NT	0.5	7	22	30
Hippoglossoides platessoides	Long-rough dab	F/NT	1.8	10	1	11
Hippoglossus hippoglossus	Halibut	F/NT	0.0	3	33	37
Lepidorhombus whiffiagonis	Megrim	F/NT	0.5	0	40	40
Leucoraja circularis	Sandy ray	F/E	0.0	0	36	36
Leucoraja naevus	Cuckoo ray	F/E	0.8	2	27	29
Limanda limanda	Dab	F/C	14.9	29	0	29
Lophius piscatorius	Anglerfish	R/NT	0.1	1	47	48
Lycodes vahlii	Vahl's eelpout	R/NT	0.0	0	12	12
Melanogrammus aeglefinus	Haddock	R/C	25.6	0	13	13
Merlangius merlangus	Whiting	R/C	7.4	2	9	11
Merluccius merluccius	European hake	R/NT	0.7	2	48	50
Microstomus kitt	Lemon sole	F/NT	3.0	14	14	28
Molva molva	Common ling	R/NT	0.2	0	69	69
Mullus surmuletus	Striped red mullet	R/NT	0.0	5	0	5
Mustelus mustelus	Smooth hound	R/E	0.3	10	23	33
Myoxocephalus scorpius	Bullrout	R/NT	0.1	1	1	2
Platichthys flesus	Flounder	F/NT	0.1	79	5	84
Pleuronectes platessa	Plaice	F/C	10.9	70	5	75
Pollachius pollachius	Pollack	R/NT	0.0	1	58	59
Pollachius virens	Saithe	R/C	13.5	0	42	42
Psetta maxima	Turbot	F/NT	0.2	76	5	81
Raja brachyura	Blond ray	F/E	0.0	97	1	98
Raja clavata	Thornback ray	F/E	0.2	60	11	71
Raja montagui	Spotted ray	F/E	0.3	48	10	58
Scophthalmus rhombus	Brill	F/NT	0.1	88	6	95
Scyliorhinus canicula	Lesser-spotted dogfish	R/E	1.7	4	38	42
Solea solea	Sole	F/C	1.5	50	1	51
Squalus acanthias	Spurdog	R/E	0.2	4	34	39
Trisopterus luscus	Bib	R/NT	0.0	1	1	2
Trisopterus minutus	Poor cod	R/NT	0.1	0	1	1

Each species is categorized based on morphology (F, flatfish; R, roundfish) and on being targeted by a fishery (C, commercial; NT, non-target; E, elasmobranch, which is a special category of non-target fish). The proportion of total North Sea biomass contributed by each species is provided together with predicted estimates of the percentage of each species' standing-stock biomass removed annually by beam and/or otter trawling.

the spatial distribution of the fish in the third quarter. To determine how this mismatch in overlap in spatial distribution between fishing effort and fish abundance may have affected the model output, we tested an alternative scenario where the spatial distribution of each fleet was such that they maximized their revenue. For otter trawlers, this was determined by their target species cod, haddock, and whiting, and for beam trawlers by their target species plaice and sole. Each of these species was given a value based on average Dutch market prices per kilogramme (cod €2.50, haddock €1.42, whiting €1.07, plaice €1.93, and sole €8.89), and their biomass in each rectangle was multiplied by this value, so giving an overall revenue value for each rectangle according to which the effort was distributed.

Validation

Fishing gears catch a proportion of all the fish in the path of the gear, including fish that may or may not have commercial value (Heessen and Daan, 1996). The part of the catch consisting of non-target species of no commercial value, and damaged or undersized target species, is considered as bycatch and is generally discarded (Hall and Mainprize, 2005). Onboard selection determines the parts of the catch that are landed or discarded. Casey (1996) suggested a logistic curve to approximate the selection process, but as we had no information to quantify this curve, we assumed L_{50} to be the minimum landing size (cod 35 cm; megrim, *Lepidorhombus whiffiagonis*, 20 cm; haddock 30 cm; whiting 27 cm; hake, *Merluccius merluccius*, 27 cm; plaice 27 cm; saithe, *Pollachius virens*, 35 cm; sole 24 cm), and L_{25} to be the minimum landing size minus 1 cm. We assumed that species with no specified minimum legal landing size had no commercial value, so all individuals of those species caught would be discarded.

For validation of the model outcomes, we had access to two sources of data: (i) total landings data of the mean international otter and beam trawl fleet for the period 1998–2004 (Greenstreet *et al.*, 2007), and (ii) North Sea discard data as estimated by CEC (2006), in which the information on the discards is presented as percentages of the total catch. For our validation, we compared the landings and discards from these sources with those predicted by the model, then calculated the ratio between predicted and observed estimates.

Results

Sensitivity analysis

To study the influence of catch efficiency of the gears on our predicted estimates of mortality, we varied the overall gear efficiencies of each species–size–gear group from 0 to 1, then determined the effect of this on total mortality. For both flatfish species, plaice and sole, mortality was mostly determined by the catch efficiency of the beam trawl and, because the standard catch efficiency was already as high as 0.95, varying the catch efficiency resulted in mostly lower estimates of mortality for plaice ranging from 13 to 113%, compared with the standard, and from 4 to 105% for sole (Figure 1). For the roundfish cod, haddock, and whiting, the mortality estimates were mostly affected by the catch efficiency of the otter trawl on the fish >29.5 cm. This may result in mortality estimates 89% lower to 67% higher for cod, 82% lower to 70% higher for haddock, and 67% lower to 52% higher for whiting (Figure 1). Whiting was the only roundfish species markedly affected by the beam trawl catch-efficiency parameter assumptions.

To assess how fish dispersion might affect predicted estimates of fishing-induced mortality, we changed the redistribution interval from weekly to annual, resulting in a marked decrease in mortality of all species (Figure 2). The standard redistribution interval was monthly. The strongest effects were observed for those species targeted by beam trawlers: plaice, sole, and to a lesser extent whiting. For those species, the lowest dispersion rate (annual interval) resulted in up to a 20% reduction in predicted mortality

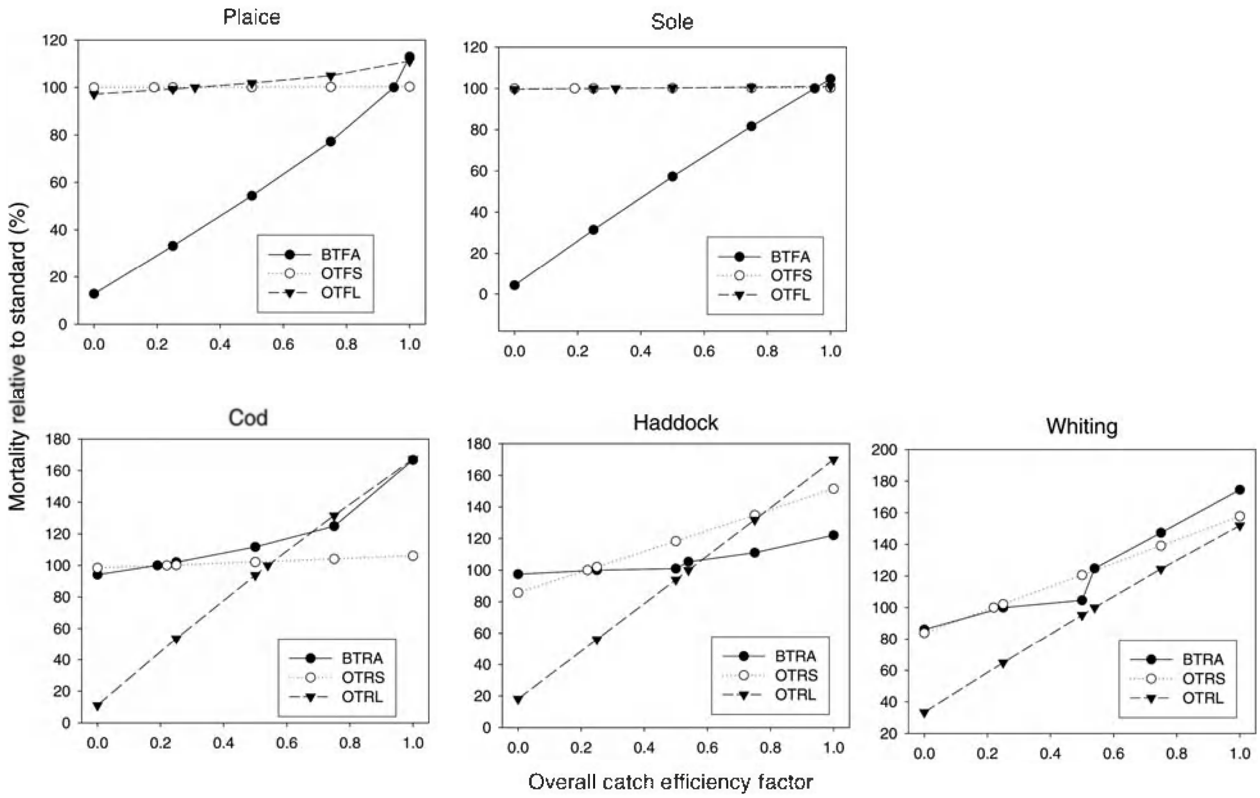

Figure 1. Sensitivity analysis of the effects of catch efficiency on the model-based mortality estimate per species. All mortalities are relative to the mortality estimated using standard settings for each gear–fish type–length category, as in Table 2 (BTFA, beam trawl, flatfish, all lengths; BTRA, beam trawl, roundfish, all lengths; OTFS, otter trawl, flatfish, <29.5 cm; OTFL, otter trawl, flatfish, >29.5 cm; OTRS, otter trawl, roundfish, <29.5 cm; OTRL, otter trawl, roundfish, >29.5 cm).

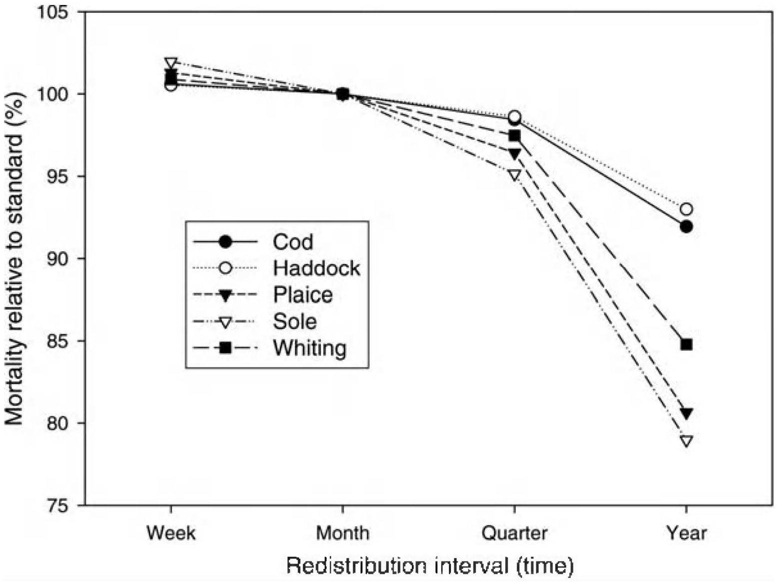

Figure 2. Sensitivity analysis of the effects of fish dispersion on the model-based mortality estimate per species. The dispersion process is simulated by redistributing the remaining fish according to their initial spatial distribution on an annual, quarterly, monthly, or weekly basis.

compared with the standard interval. Shortening the redistribution interval to weekly had a negligible impact on mortality estimates.

The possible effect of a reduced overlap between fishing effort and fish distribution was studied by determining the mortality when fishing effort was distributed to maximize revenue, assuming a perfect knowledge of the distribution of the fish. For cod, this resulted in an almost doubling of the mortality (192%; see Figure 3), whereas for most of the other species, it was 40–60% higher. Only plaice showed a slight decrease in mortality because to maximize their revenue, the beam trawl fleet was assumed to follow the distribution of the much more valuable sole.

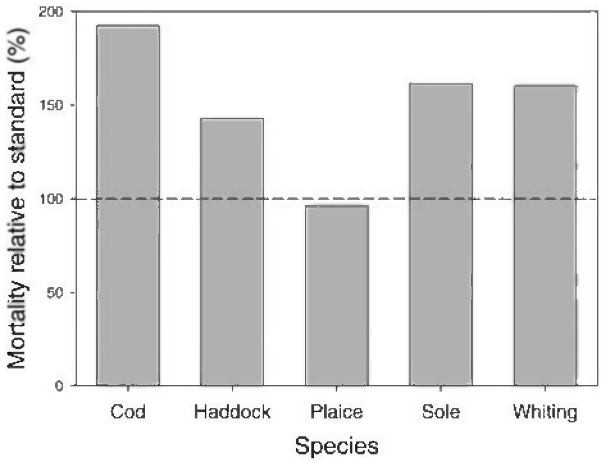

Figure 3. Sensitivity analysis of the effects of overlap in spatial distribution between fishing effort and fish abundance on the predicted mortality estimate of each species. The bars indicate the predicted mortalities when the fleet distribution is based on a maximization of their revenue, relative to the standard, which is based on the spatial distributions reflected in the data sources.

Validation

The model generally performed well in predicting the quantities of each species landed by the beam trawl and by the otter trawl (Figure 4). There was perhaps a tendency for roundfish landings in the otter trawl to be slightly overestimated and flatfish landings in the beam trawl to be slightly underestimated by the model. Generally, though, landings of species caught in gears where they were not the principal targets of the fishery concerned, e.g. flatfish in otter trawls and roundfish in beam trawls, tended to be underestimated by the model. Although these differences might have been quite high in terms of relative proportion, i.e. predicted sole landings from otter trawls were only 40% of observed landings, in absolute terms (the difference in tonnes), the discrepancy was small. However, more serious problems emerged regarding some of the discard predictions provided by the model, particularly in respect of the otter trawl data, where the model suggested levels of cod, whiting, and plaice discards that were considerably smaller than the actual levels of discarding suggested by sample data. For all other species and gears, the model predicted discard levels reasonably accurately.

Species-specific annual fishing mortality

Table 4 lists 47 species which, from analysis of research-vessel survey data, are considered to contribute at least 0.1% of the total North Sea demersal fish biomass, together with predictions derived from our model of the percentage of each species standing-stock biomass removed each year through fishing activity distinguishing beam and otter trawling. This suggests that many of the non-target species are experiencing levels of fishing mortality that are at least as high, if not higher, than those inflicted on species that are the principal targets of fisheries.

Size-specific mortality

Size-dependent variation in fishing mortality was examined among four distinct groups of fish: non-target roundfish,

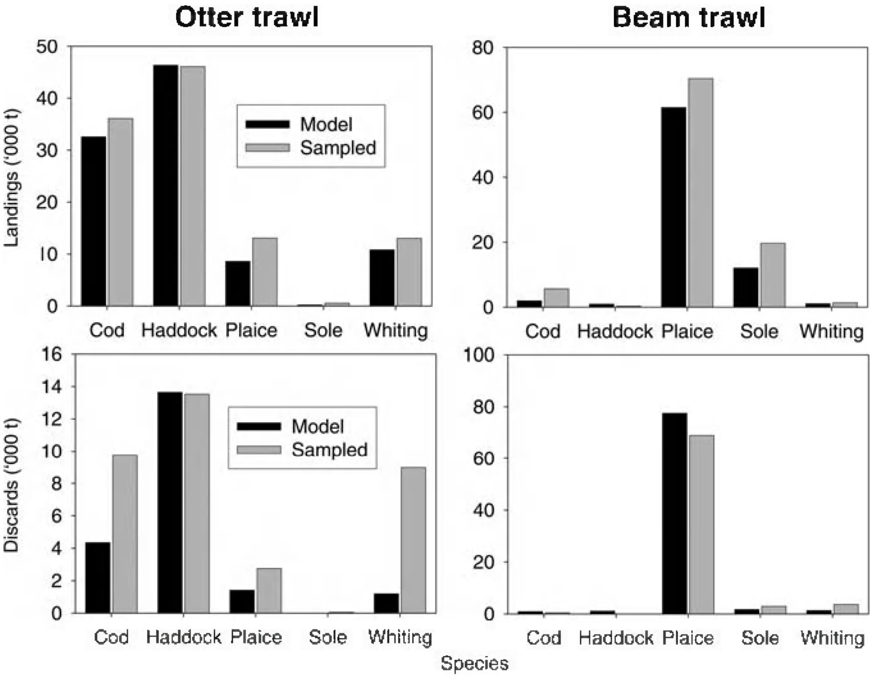

Figure 4. Comparison of predicted estimates of landings and discards with estimates obtained from sampling programmes for the main commercial species.

non-target flatfish, elasmobranchs, and commercial species. Patterns were similar for all four groups, both in absolute (Figure 5) and relative terms (Figure 6). In absolute terms, removals by fishing peaked in the small to medium-size categories of each species group (Figure 5), but the patterns reflected both variation in the catching power of the fisheries and variation in abundance at length within each of the groups of fish. The more telling patterns were observed in the relative mortality data, where fishing removals were expressed as a percentage of the biomass present. The overwhelming impact of fishing on the larger fish within each group was clear (Figure 6). Interestingly, the model suggested that fishing mortality in larger non-target flatfish was lower than among non-target roundfish of similar size. At body lengths >50 cm, there appeared to be little difference in the fishing mortality experienced by non-target roundfish and commercial species; indeed between 35 and 50 cm, non-target roundfish mortality may even exceed the mortality caused by fishing among commercial species (Figure 6).

Spatial distribution of fishing mortality

Spatial distributions of fishing mortality were determined for the same four groups of fish. Differences mainly reflected the distributions of the fisheries modelled: an otter trawl fleet in the northern North Sea primarily targeting roundfish and a beam trawl fleet in the southeast targeting principally flatfish (Figure 7). Mortality among fish belonging to commercial species was highest in the southeastern North Sea, suggesting that beam trawlers may have the greater impact on the demersal fish community in the North Sea. Interestingly, the model results suggested the presence of "sink" regions; ICES rectangles where the estimates of annual fisheries biomass removals actually exceeded the standing-stock biomass estimated to be present. Such a situation was maintained by the fish redistribution process operated within the model, implying that high levels of fishing activity in such areas could only be sustained through the continuous immigration of fish into these rectangles from adjacent areas.

Discussion

The progressive implementation of an EAFM imposes an increasing need for fisheries scientists to be able to assess the impact of fisheries on ecosystem components other than just commercial fish species. Here, we demonstrate how a simple intuitive model, requiring only basic fish abundance and fishing effort data, together with information on gear catching efficiency, can deliver plausible estimates of the impact of fishing on the entire fish community, including non-target species for which such information previously did not exist. Moreover, the development of our model provides a clear indication of precisely the type of data that are required, and where current data quality and availability need to be improved, to increase confidence in such estimates.

Abundance data

Our estimates of fish abundance data were based on two regular demersal trawl surveys: the IBTS and the DBTS, each using a different gear, otter and beam trawls, respectively, specifically designed to catch different types of fish. Knowing the catchability rates of fish of different size belonging to different species in the various survey gears is critical to converting survey catch-sample density estimates into estimates of true density at each sample location. Recent advances made in this respect (Fraser *et al.*, 2007) allowed us to estimate the levels of fish abundance in each ICES rectangle that were required by our model. However, we modified their approach slightly. Fraser *et al.* (2007) showed that the catchability of most flatfish species in the IBTS GOV trawl was low. Consequently, the estimates of local abundance that

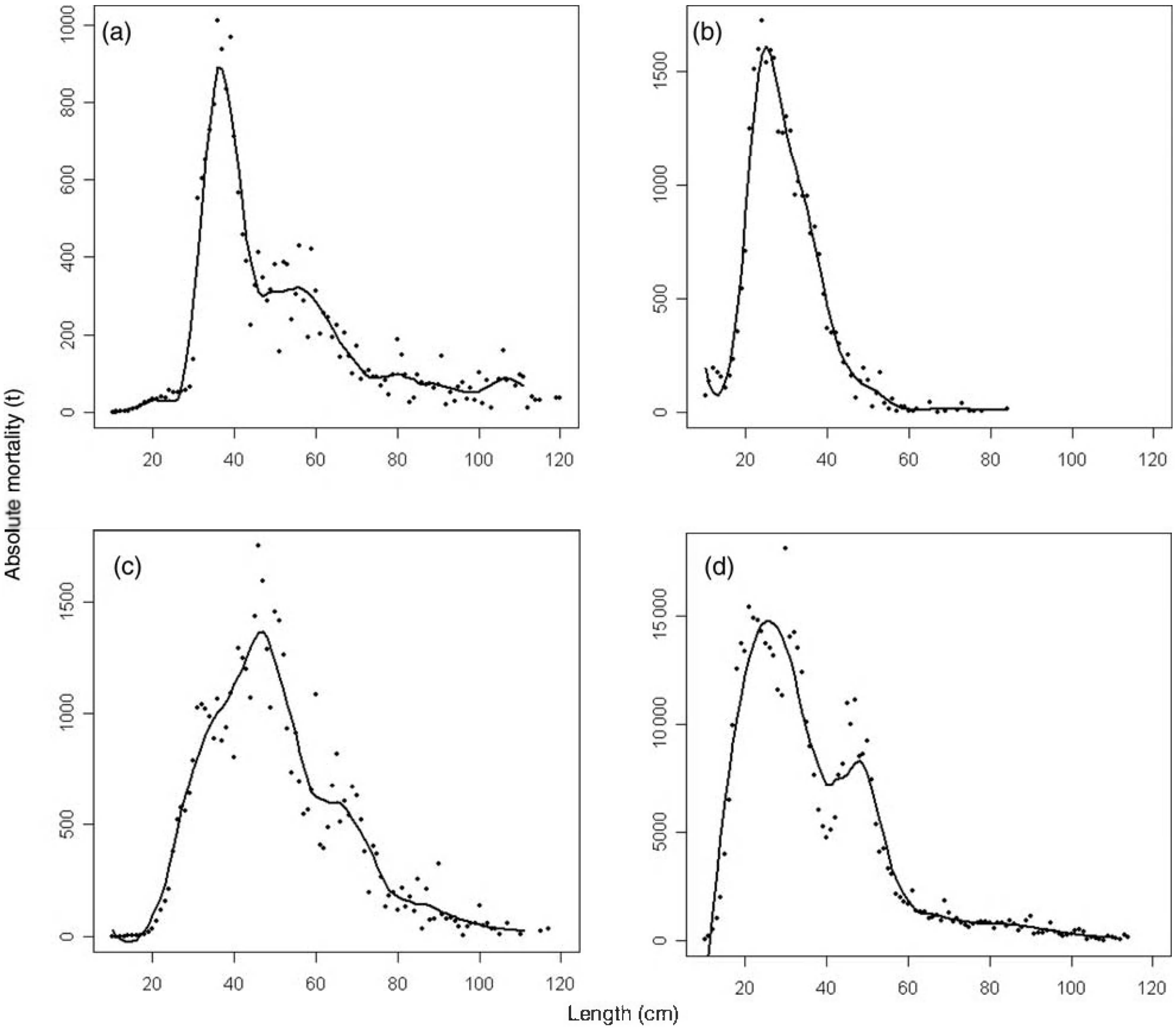

Figure 5. Absolute size-specific removals by fishing (tonnes) of fish assigned to one of the four major groups of fish: (a) non-target roundfish, (b) non-target flatfish, (c) elasmobranchs, and (d) commercial species. Dots are model estimates per 1-cm class, and lines are derived from a loess smoother.

took account of catchability in the GOV trawl relied heavily on the application of high raising factors. In contrast, the beam trawl used in the DBTS catches flatfish more effectively, such that the raising factors required to convert sample density estimates to estimates of actual density were considerably lower. Using the DBTS to provide estimates of flatfish abundance in the area covered by this survey meant, therefore, that as a rule, the raising factors necessary to derive estimates of local abundance tended to be lower across the demersal community as a whole, and this seemed to us to be a favourable situation. Reliable estimation of local fish abundance is a key requirement of our model. Better understanding of the processes that affect the catchability of different species in RV survey trawls would promote improved quantification of catchability rates and increase confidence in the estimates of fishing impact derived from the model.

We used Q3 survey data only to estimate local abundance. This is because the catchability raising factors that we have relied on were only derived for the Q3 surveys, and there is no evidence as yet that these can be applied reliably to survey data collected at different times of the year. With regard to our model, this is clearly not ideal. Incorporation of seasonal variation in local fish abundance (e.g. Zheng *et al.*, 2001; Hunter *et al.*, 2006), and better understanding of fish dispersion processes, would considerably enhance our model and improve the estimates of fisheries impacts on the demersal fish community.

Effort data

Access to fishing effort data that are reliable, updated regularly, collected at an appropriate spatial resolution, and distinguish between the necessary different fishing métiers is an essential requirement of our model.

Most of the commercial species are known to move over distances as large, if not larger than, an ICES statistical rectangle (Bolle *et al.*, 2005), so this spatial scale was probably appropriate for this first trial of our model. However, some non-target species may not move so extensively, and it may therefore be

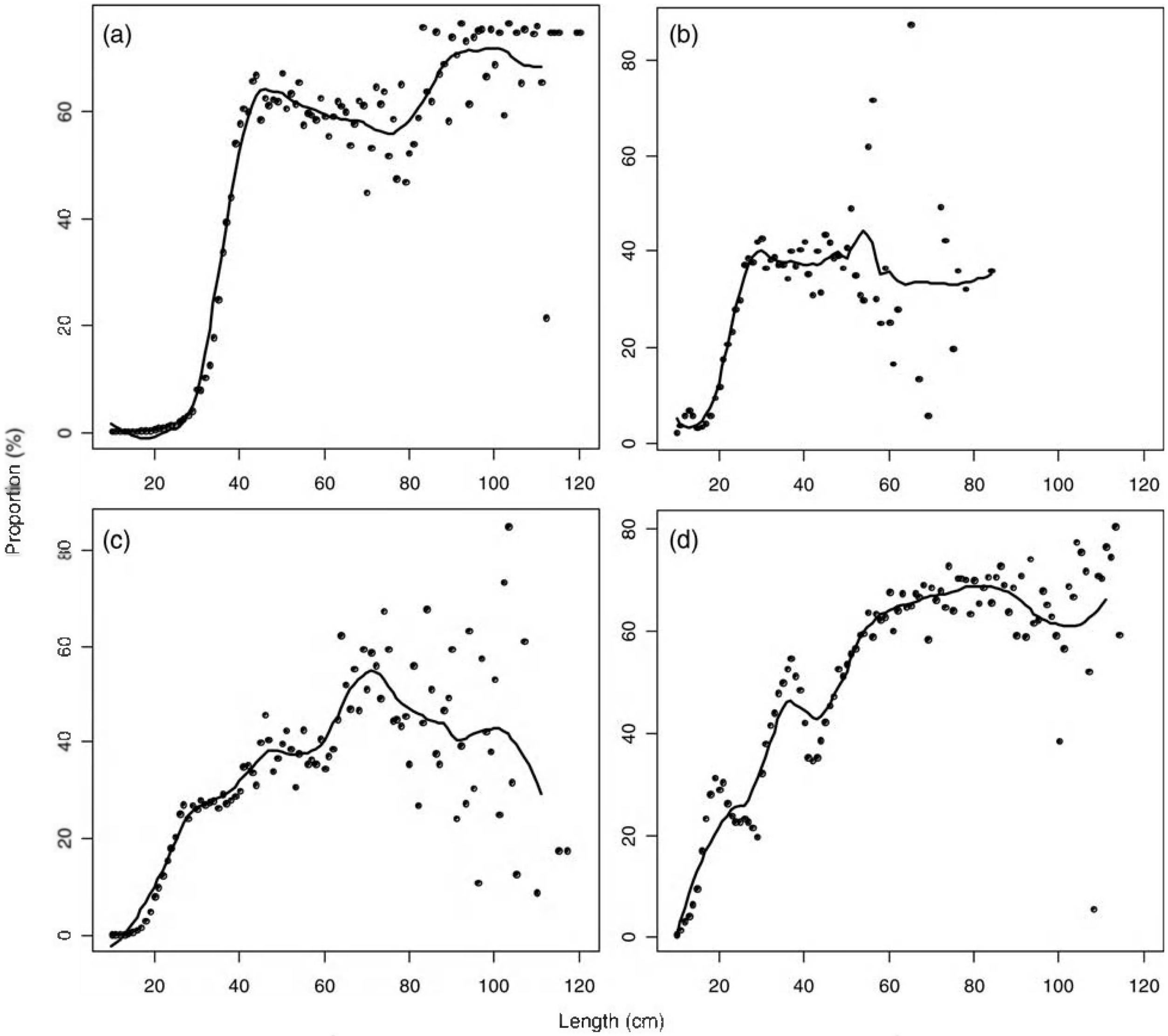

Figure 6. Relative size-specific fishing mortality (absolute removals expressed as a percentage of standing-stock biomass) experienced by fish assigned to one of the four major groups of fish: (a) non-target roundfish, (b) non-target flatfish, (c) elasmobranchs, and (d) commercial species. Dots are model estimates per 1-cm class, and lines are derived from a loess smoother.

necessary to consider smaller spatial scales in future (Piet and Quirijns, 2009). Moreover, fishers exhibit fine-scale patterns within ICES rectangles (Rijnsdorp *et al.*, 1998) to exploit hotspots with higher than average abundance of fish. However, application of an increased spatial resolution in this model would require information on the distribution of both the fishers and the fish to be available at such resolution. This is still not the case, but current moves to extend the availability of vessel monitoring by satellite (VMS) data would support such developments (Piet *et al.*, 2007; Fock, 2008).

Similarly, we consider the use of annual estimates of fishing effort to be appropriate for our current purposes, given that the information necessary to incorporate seasonal change in the distributions of different fish species into our model was not available. However, when seasonality in fish distribution can be incorporated into the model, it would then be more appropriate to use effort data summarized over shorter periods, e.g. quarterly or even monthly.

No fishing vessel operating in the North Sea will be identical to another and be able to fish identical gears in exactly the same way. In constructing our model, we needed to assign the vessels into a small number of discrete categories. Ultimately, we considered only four categories: an otter trawl fishery operating mainly in the northwestern North Sea, a beam trawl fishery with a mesh size of 10 cm in the central and the northern North Sea, a beam trawl fishery with a mesh size of 8 cm in the southern North Sea, and a beam trawl fishery operated by "eurocutters" with a lower speed and smaller gear operating mainly within the plaice box and 12-mile zone. Our model would no doubt benefit from the inclusion of a greater number of more tightly defined fishing métiers. However, the lack of similarly detailed fishing effort data and the scarcity of information necessary to parameterize

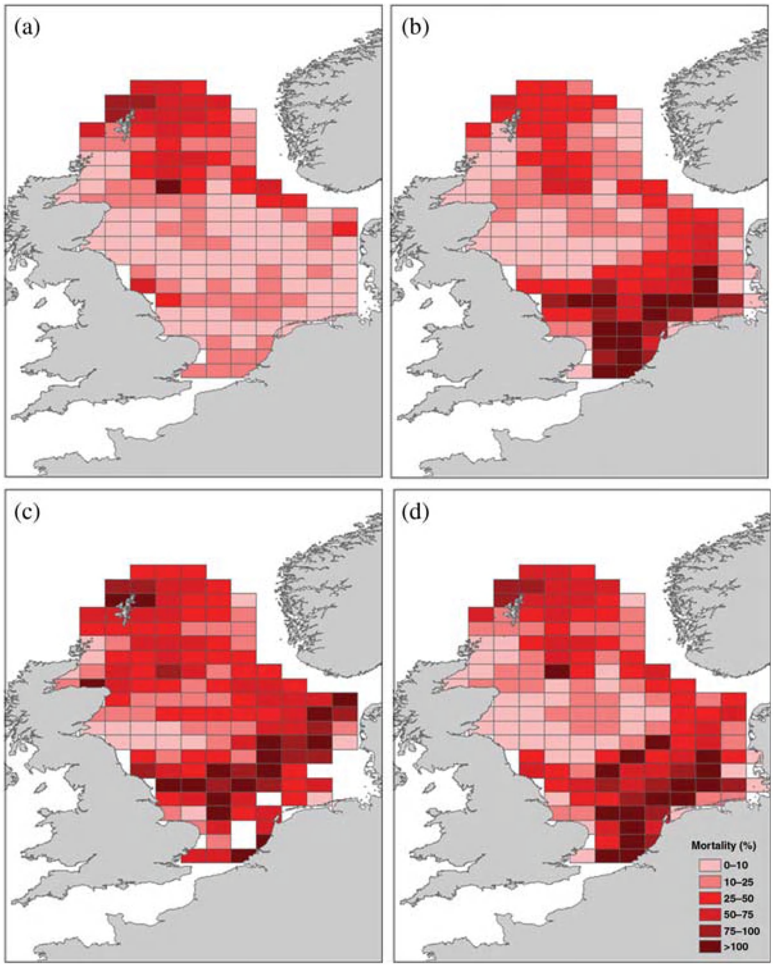

Figure 7. Spatial variation in modelled estimates of relative annual mortality (absolute biomass removed from the North Sea, expressed as a percentage of standing stock biomass) for (a) non-target roundfish, (b) non-target flatfish, (c) elasmobranchs, and (d) commercial species.

catchability equations for a larger number of métiers currently preclude this. Such restrictions may decline in future, providing the scope to develop our model further.

Gear efficiency

We only had limited information on gear efficiency for each of the fisheries we modelled. For some commercial roundfish and flatfish species, information on mesh-size-dependent selectivity in different gears was available, whereas information on herding effects was available for the otter trawl only. This information was assumed to apply to all species of the same type (i.e. roundfish or flatfish). For other factors affecting gear efficiency, such as the position of the fish in the water column relative to the gear, and escape rates under the footrope, almost no information was available and "heroic" assumptions were necessary. Our sensitivity analysis established that the gear efficiencies input to the model had a strong influence on the resulting estimates of fishing impact output by the model. Improving confidence in the values of our gear efficiencies is therefore a priority, and research is currently underway to address the issue (Reid *et al.*, 2007).

Validation

Validation of the model suggested that, in broad terms, it performed well and provided reasonably reliable estimates of mortality for the commercial species for which reliability could be assessed. Given the issues already discussed revolving around the current structure of the model, the choice of spatio-temporal scale, and the setting of various input parameter values, discrepancies between the predicted estimates of impact and estimates derived from sample data were generally small and could easily be resolved by relatively minor changes to parameters. The biggest problems emerged with regard to the model's inability to predict discard levels of roundfish accurately in the otter trawl fishery. One possible explanation is that our model does not take discarding through exhausted quota or highgrading into account. The latter is the practice whereby fishers discard fish of legal size so that they can instead fill their quota with larger fish of generally greater value per kilogramme. The net effect of this is to increase the size at which fish are landed, rather than discarded. Increasing the landing size of cod from 35 to 40 cm, and of whiting from 27 to 31 cm, was all that was required for predicted discard levels to match the levels implied in the sample data. Such small differences in landing size are well within the range that might be anticipated to arise as a result of highgrading. However, making such changes in landing size to make the model better fit the observed discard data weakened the fit of the model to the landings data; causing the model to underestimate landings. This, in turn, could be compensated for by the fishers' ability to locate patches within the ICES rectangles with greater-than-average abundance of commercially valuable fish, so increasing their landings.

For the non-target species, our model suggested that on average about half the standing-stock biomass of larger-bodied elasmobranchs was removed annually by fisheries (Figure 6). Although this constitutes species-specific mortalities varying between 29 and 98% (Table 4), reflecting the local circumstances of rare species whose distributions are probably not adequately reflected by the surveys, it coincides closely with instantaneous mortality rate estimates of 0.59 (=45%) for thornback ray (*Raja clavata*) and 0.70 (=50%) for starry ray (*Amblyraja radiata*) derived from catch-curve analyses (Walker and Hislop, 1998).

Mortality rate

The estimates of mortality rate for non-target species presented here are based on the assumption that the catchability of non-target roundfish or flatfish species is equal to that of commercial species of similar size and type. Catchability depends on both gear efficiency, which is defined as the fraction of the fish present in the path of a trawl that is retained by the gear, and the distribution of fish in relation to the distribution of the fleet (Rijnsdorp *et al.*, 2006). Although there is no evidence that the assumption does not hold for gear efficiency, it is likely that within an ICES rectangle, skippers are able to locate patches with higher-than-average abundance of commercial fish, so lowering the relative impact on non-target fish. The current estimates are therefore probably overestimated to an extent that depends on the skill of the combined skippers to find such patches.

Previous studies have demonstrated long-term changes within the non-target species component of the North Sea groundfish community. For example, declines in groundfish species diversity remain evident even when commercially targeted fish are excluded from the analysis (Greenstreet and Hall, 1996; Greenstreet *et al.*, 1999). Similarly, long-term changes in life-history trait composition favouring the typically fast-growing, early maturing species, which were demonstrated for the whole groundfish assemblage in an area where fishing effort increased markedly through the latter part of the 20th century (Jennings *et al.*, 1999), were also apparent when only non-target species abundance data were analysed (Greenstreet and Rogers, 2000). Previously, fishing has been assumed to be the cause of these trends in non-target fish. Now, our model for the first time provides direct evidence that many of the non-target species in the demersal fish community of the North Sea have been, and are being, impacted by fisheries to an extent that is at least as high, if not in some cases higher, than the species specifically targeted by fisheries. Although not specifically targeted by fisheries, long-term declines in the abundance of most elasmobranch species in the North Sea have been attributed to excessive fishing mortality (Brander, 1981; Walker and Heessen, 1996; Walker and Hislop, 1998), because their particular life-history traits render them especially vulnerable to raised levels of mortality (Hoenig and Gruber, 1990). Our predictions suggest that those elasmobranch species that remain sufficiently abundant to be included in our analyses are still subjected to levels of fishing mortality as high as many targeted species, a situation that may not bode well for their future. Interestingly, Greenstreet and Hall (1996) noted a long-term increase in the proportion of flatfish within the groundfish community in the northwestern North Sea. Our model suggests that the impact of fishing on non-target flatfish species is considerably lower than on non-target roundfish species, and this difference may explain the apparent long-term trend.

The spatial distribution of mortality shows marked differences between species groups. Roundfish species targeted by otter trawlers show the highest levels of mortality in the northern North Sea, whereas the spatial distribution of mortality of all other groups is determined by the distribution of the beam trawl fishery and shows the greatest mortality in the southern part of the North Sea. Predicted mortality rates exceeded 100% in some of the most heavily fished rectangles. To some extent, this may be explained by processes of growth and recruitment, but it may also suggest that high levels of fishing activity in some parts of the North Sea are only sustained by the immigration of fish into these areas from surrounding regions. Assuming that fishers concentrate their activities where catch rates are highest, that this is where fish densities are relatively high, and that fish concentrate in their preferred habitat, then the operation of some form of ideal free distribution mechanism might bring this about. The local depletion of fish, through catching them, in the most optimal habitat would provide the opportunity for fish in less good locations to move into the vacated better habitat to gain access to higher grade resources (Fretwell and Lucas, 1970; Partridge, 1978).

Recently, the ecological quality objective for the North Sea demersal fish community has focused on fish size composition (Heslenfeld and Enserink, 2008). Fishing is a size-orientated activity, driven by both economic pressures and regulatory restrictions. The effects of these on size-related mortality among the commercial species subject to stock assessments has long been documented (ICES, 2007). It has been assumed that mortality among non-target species taken as bycatch is similarly size-dependent, and this assumption has largely underpinned the focus on size composition when setting community level management objectives (Greenstreet, 2008). Our model provides firm evidence that this is actually the case, suggesting a step-like function between body length and mortality for both non-target roundfish and flatfish. For both groups of fish, mortality increased rapidly from negligible levels at body lengths up to 30 cm among roundfish and up to 20 cm among flatfish, to maximum rates of 60 and 40% of biomass removed annually at body lengths of 40 and 27 cm, respectively. Our model therefore confirms the basic effect that fishing is likely to have on the size composition of fish of all species that make up the North Sea demersal fish community.

Application

To implement an EAFM, indicators are required that describe the pressures affecting the ecosystem, the state of the ecosystem, and the response of managers (Jennings, 2005). Piet *et al.* (2007) evaluated potential pressure indicators, concluding that measures of annual fishing mortality were best because they could be linked directly, through specific theoretical processes, to variation in the chosen indicators of state (Piet and Jennings, 2005). Our model provides actual estimates of annual fishing mortality for many demersal fish species in the North Sea. When combined with information on life-history traits (e.g. age- and length-at-maturity, growth rate, maximum length), data that are becoming increasingly available for many non-target species (Jennings *et al.*, 1999; Greenstreet *et al.*, 2007), this opens up the possibilities of modelling the population dynamics of many more species in the North Sea than just the commercially important assessed stocks (Hall *et al.*, 2006). Such models could service the needs of managers addressing biodiversity issues associated with the impacts of fishing on threatened and declining species.

Acknowledgements

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