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Population trends of bycatch species reflect improving status of target species

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Abstract

Synthesis studies of fish stocks worldwide suggest improving status of mainly target species that are fully assessed. Other analyses, primarily based on catch data alone, but which include a wider range of species as well as bycatch, present a different view. Catch-only analyses could be more robust if fishery-independent data were used and discards accounted for. We develop a model that uses only survey biomass at length and landings data to estimate fishing mortality, spawning stock biomass (SSB) and discards. An analysis of species from the North Sea shows the model results compare well with most fully assessed stocks. When applied to bycatch species with limited data, trends in fishing mortality and SSB typically reflect those of the target species. In the last decade, mean fishing mortality rates have tended to decline, while mean SSB has increased. Despite increasing SSB, recent mean recruitment appears to have been lower than previously which may limit future biomass recovery. Species usually associated with more northerly distributions appear to show the greatest effect of weaker recruitment, which may be linked to climate. Estimated discards have tended to decline in magnitude as a result of reduced fishing mortality and associated lower total catches. The model offers a simple way to use both landings and survey data to obtain more detailed population trends for data limited species.

KEYWORDS
bycatch, data limited methods, discards, North Sea, population trends, stock assessment

1 | INTRODUCTION

Some recent studies have documented improving status of many fish stocks worldwide (Cardinale et al., 2013; Fernandes & Cook, 2013; Worm et al., 2009). These analyses suggest that the decline in some of the world's major stocks has halted or reversed in the last decade and that overfishing in relation to MSY reference points is less prevalent. While such improvement is important in demonstrating that fishery management may be effective, it is mainly based on the analysis of the species or stocks for which adequate data for assessment exist. Meta-analyses that examine predominantly target species may give only a partial impression of the broader status of exploited fish stocks.

A number of authors have argued on the basis of analyses of catch trends that, globally, fish stocks are deteriorating and that the status of assessed stocks gives a biased impression of all stocks (Froese, Zeller, Kleisner, & Pauly, 2012). Costello et al. (2012), for example, suggest that the status of unassessed species may be worse than that of assessed species, while Piet, van Hal, and Greenstreet (2009) suggest that the fishing mortality on bycatch species may be higher than target species in the North Sea. However, the use of catch data alone to assess stock status is controversial (Branch, Jensen, Ricard, Ye, & Hilborn, 2011; Pauly, Hilborn, & Branch, 2015) principally because a trend in catch cannot be unambiguously explained by a trend in either biomass or fishing mortality unless conditioning assumptions are made (Martell

& Froese, 2013). One of the main concerns is that a decline in catches can be misinterpreted as a decline in biomass when it might, in reality, be the result of a reduction in fishing mortality. This difficulty can be exacerbated by the fact that the catch data used are often the landings rather than to total removals including discards. Where discarded quantities are variable over time, this may be a serious issue.

The main target species that are typically subject to detailed assessments represent only a fraction of all species caught. In fisheries with non-selective gears such as trawls, there is usually a mixture of many other species that contribute to the catch. Such species are often less abundant but may nevertheless comprise an important component of the catch value and for convenience are referred to somewhat loosely here as "bycatch." Assessments of bycatch species are often absent or limited, and as a result, less is known about their status. An important question is therefore whether the status of bycatch species in mixed fisheries reflects that of the target species with which they are associated and, in particular, whether the apparent improvement seen in the assessed target fish stocks is mirrored in the bycatch. To examine this issue, we analysed data for 24 species (or species groups) caught in mixed fisheries in the North Sea where complex assessments are limited or absent for most and compared them to the status of the few species where stock status is better known. As a survey index is available for these species, it is possible to attribute trends in catches explicitly to trends in fishing mortality and biomass.

In the North Sea, both otter trawl and beam trawl fisheries have a substantial bycatch with the former targeting mainly Atlantic cod (*Gadus morhua*, Gadidae), haddock (*Melanogrammus aeglefinus*, Gadidae) and whiting (*Merlangius merlangus*, Gadidae) and the latter targeting plaice (*Pleuronectes platessa*, Pleurnectidae) and sole (*Solea solea*, Soleidea). These five target species are routinely assessed by the International Council for the Exploration of the Sea (ICES). Their assessments are comprehensive making use of catch-at-age data that include both landings and discards, and multiple research vessel surveys designed to sample these species. For cod, haddock, plaice and sole, state-space statistical assessment methods are used (Aarts & Poos, 2009; Gudmundsson, 1994; Nielsen & Berg, 2014), while for whiting, a VPA approach is applied (Shepherd, 1999). Overall, the quality of the assessments is considered suitable for evaluating stock status and the provision of management advice. All target species show improvement with lower fishing mortality rates (*F*) and increasing or stable spawning stock biomass (SSB) in recent years (ICES, 2015a). However, the whiting assessment is subject to greater uncertainty, at least in part due to concerns about the catch data.

Of the many bycatch species from these fisheries, only turbot (*Scophthalmus maximus*, Pleurnectidae) is subject to a full age-based assessment, while megrims (*Lepidorhombus* spp., Pleurnectidae) are assessed with a Schaefer production model (Schaefer, 1954) using Bayesian methods. A small number of stocks, such as seabass (*Dicentrarchus labrax*, Moronidae), are assessed but over a much larger area than the North Sea proper. Most other species are either assessed by examining abundance trends from research vessel surveys or not assessed at all. While a yield/biomass ratio can be calculated as an index of exploitation rate, unless discards are accounted for, interpreting the trend in exploitation is difficult. Hence, for many species, stock status is highly uncertain or unknown.

As research vessel surveys for many of the main target species use gears based on otter trawls, they retain bycatch species in a similar fashion to the commercial the otter trawl fleet. It should therefore be possible to use survey indices for many bycatch species as a basis for assessment. Furthermore, landings records for many of these species are available which, subject to assumptions about natural mortality, could be used to scale an assessment to absolute biomass. The principal limitation is the veracity of these data as a record of catch and, in particular, whether discards comprise a substantial part of the removals. An analytical method that is able to make use of the survey and landings data so that estimates of *F*, SSB and recruitment can be made while accounting for all fishery removals is necessary.

Even where age data are absent, trawl survey data generally provide observations not only on number per tow but also on an associated size distribution, which enables a variety of possible approaches to assessment. While a fully length-based assessment method such as catch-at-size analysis (Sullivan, Lai, & Gallucci, 1990) or a method with an age-structured model that uses length observations such as Stock Synthesis (Methot & Wetzel, 2013) might be applicable, these generally work best with well-sampled length frequency distributions to estimate model parameters. These are not always available for species of low abundance. To mitigate these problems, we develop a method of assessment based on the Collie–Sissenwine approach (Collie & Sissenwine, 1983) that is only weakly dependent on the survey length data to determine discards given the landings (Heath & Cook, 2015). This approach uses a population dynamics model cast purely in terms of numbers, avoiding the need to estimate age or growth rates. We apply the model to the five target species with comprehensive assessments to show that it adequately characterizes important trends and then use it to estimate *F*, SSB and recruitment for a range of bycatch species taken in the mixed fisheries.

We use the International Bottom Trawl Survey (IBTS) that has been widely used to document relative biomass trends in the North Sea fish stocks (Daan, Richardson, & Pope, 1996). Heessen and Daan (1996) showed that up to the mid-1990s, most bycatch species were increasing in abundance based on the IBTS data. More recent ICES assessments (ICES, 2015b) have used the survey to update these trends for some species but they do not estimate absolute biomass or exploitation rate. Sparholt (1990) used estimates of survey catchability to scale survey indices and obtain a point estimate of total biomass, while more recently, Piet et al. (2009) adopted a similar approach using catchability, gear selectivity and fishing effort to estimate yield:biomass ratios as a proxy for fishing mortality averaged over a number of years. The studies provide a snapshot of biomass and exploitation rate but as adequate survey data exist at least since the early 1980s, it is possible to reconstruct trends over three decades using the new model and compare these with those obtained by complex assessment models.

2 | METHODS

2.1 | Data

2.1.1 | Survey data

The longest running and most comprehensive survey of demersal fish in the North Sea is the IBTS. The survey covers all of ICES

Subarea 4 (North Sea) and Division 3a (Skagerrak) (Figure 1) using a standardized high headline otter trawl, the "GOV." Multiple research vessels sample each 30 × 30 km statistical rectangle at least twice in the period January–March each year with a 30 or 60 min tow. Sampling protocols are generally considered to have been standardized since 1983 (ICES, 2015b) and this forms the base year for the analysis presented here. The data were downloaded from the ICES DATRAS data centre (http://www.ices.dk/marine-data/data-portals/Pages/DATRAS.aspx, accessed 31/03/2016). We extracted data on number at length to calculate an overall mean number per hour and an associated length distribution for each species each year for the period 1983–2015. This allowed the calculation of the associated annual mean weight of individual fish by species (i.e. the mean weight averaged over all size classes). Length-to-weight conversions were carried out using standard relationships for the North Sea given in Coull, Jermyn, Newton, Henderson, and Hall (1989). In addition, the IBTS data contain estimates of maturity, which we used to obtain the length at which 75% of fish were mature in order to be able to calculate SSB. The maturity data are sporadic but provide the best available information on maturity for the area, so for each species, we aggregated the data across all years to obtain a mean value for the whole period.

In the case of anglerfish and megrim, the ICES assessment includes ICES Division 6a (West of Scotland, Figure 1). To obtain a survey index consistent with this assessment unit, we included the Scottish first quarter survey data with the IBTS data to calculate an abundance index for the combined area. The Scottish survey uses the same sampling protocol as the IBTS and takes place at the same time of year and was therefore treated simply as an extension of the North Sea survey.

2.1.2 | Landings and discard data

Official statistics on the total annual landed weights of all species, by all nations engaged in fisheries in North Sea, between 1983 and 2015 were accessed from the FAO/ICES FishSTAT data set (accessed 31/03/2016). In some cases, species were aggregated into broader taxonomic groups to accommodate national and temporal differences in the way species were recorded. These were "mullets" (Mugilidae), "skates and rays" (Rajidae) and "tub and red gurnards" (*Chelidonichthys cuculus* and *Chelidonichtys lucernus*, Triglidae). Where ICES assessment working group estimates of landings differed from the official landings, the working group values were used and taken from ICES (2015b) or ICES (2015c) in the case of anglerfish and megrim. Discard data, where available, were taken from the reports of ICES assessment working group (ICES, 2015b, 2015c). Data on the effective minimum landing sizes (EMLS) of fish were taken from Heath and Cook (2015). The EMLS is either the legal minimum landing size, or where no such restriction occurs, it is the typical minimum size landed as estimated from ICES assessments.

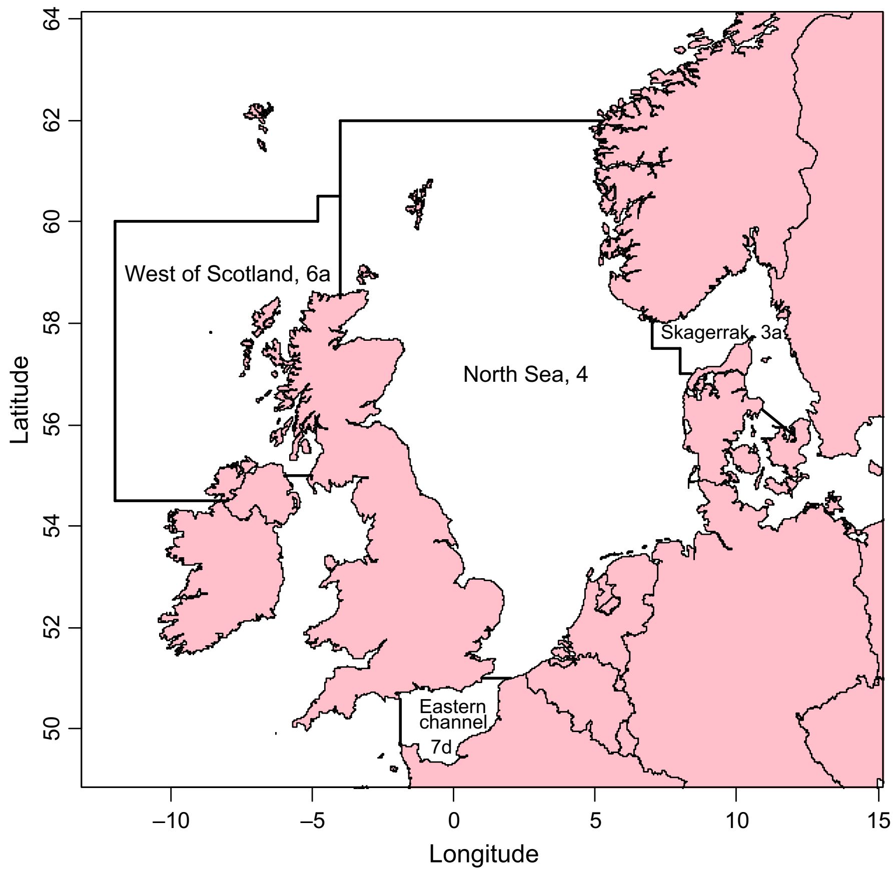

FIGURE 1 Map showing the location of International Council for the Exploration of the Sea stock areas used in the analysis [Colour figure can be viewed at wileyonlinelibrary.com]

2.1.3 | Species chosen

We selected all species, or groups of species, that had a coherent time series of landings data and that were routinely sampled in the IBTS survey. The principal limitation was the landings data where official records frequently do not accurately identify species or geographical area. A full list of the 24 species considered is given in Table 1, which includes the five target species.

2.2 | Assessment model

As few of the bycatch species have age-based data available, the model developed here is framed in terms of numbers of fish where the population is split between two stages, recruits and post-recruited fish as proposed by Collie and Sissenwine (1983). Such Catch-Survey Analysis models have been shown to be capable of providing reliable information about general stock trends (Mesnil, 2003). We extended the model to account for discards and refer to it as the Landings-Survey-Discard (LSD) model.

The equations defining the model are given in Table 2. A simple projection Equation 2.1 describes the number of fish, N, in year $t + 1$ as a function of the population in the previous year, t, and recruitment, R, in year $t + 1$. While it may be possible to describe R as a function of the spawning stock using one of the conventional stock–recruitment relationships, we simply assume that recruitment is a log-normally distributed random variable (Equation 2.2). In common with many current assessment models, we assume that the fishing mortality rate, F_t, follows a random walk through time (Equation 2.3).

The observation Equations 2.4 and 2.5 show how landings, L, and the survey index, u, are related to the population in the sea. The observed landings and survey indices are assumed to have lognormal observation errors (Equations 2.6 and 2.7).

The survey index is taken to be proportional to the number of fish in the population through a constant catchability, q. This assumption is perhaps questionable given that smaller fish are usually retained less efficiently in trawls. Where there are large annual changes in recruitment, it might be expected that q averaged over all size classes will fluctuate as a result. For simplicity, we retain the constancy assumption as later analyses suggest that departures from it are less important in determining stock trends.

The mean weight of individual fish, $\bar{w}_t$, and the proportion of the total catch biomass discarded by size π_t are necessary input values for the model and were calculated directly from the observed survey biomass at length. They are defined by Equations 2.8–2.10. The mean weight can be calculated by summing over the biomass at length in the survey and dividing by the total number of fish.

The calculation of the proportion of the catch discarded, p_t, requires some assumptions about the process of discarding, and here, we follow the model described in Heath and Cook (2015) where it is expressed as a combination of size-related discarding and non-selective "bulk" discarding. In the former process, fish below a certain size are discarded due to legal constraints or commercial value, while in the latter process, fish are simply discarded regardless of size. This may occur when quota limits encourage discarding or species of low value are not retained, regardless of size.

Heath and Cook (2015) show that the length distribution in the survey catch can be used to estimate the proportion of fish discarded by size, π_t, given an estimate of the EMLS. We assumed $\pi_l = 1$ for fish below the EMLS and zero for larger lengths in Equation 2.10. The EMLS values are given in Table 1.

2.3 | Parameter estimation

The values $\bar{w}_t$ and π_t are treated as known and were calculated directly from the survey length frequency as described above. Natural mortality, M, was assumed to be dependent on mean weight and given by the equation $M_t = 3.69\bar{w}_t^{-0.305}$ in Lorenzen (1996). However, to give some comparability to assessments that assume either a constant or fixed values of M, the annual values from the Lorenzen equation were rescaled to give the same or similar mean as the M value used by ICES. This allows M to change with the annual values of mean weight but with the long-term mean the same as the conventional value. ICES uses a constant of $M = 0.1$ for plaice and sole but for cod, haddock and whiting M values are age- and year-specific and calculated externally to the assessment from multispecies models (ICES, 2015b). For these species, the scale of M was arbitrarily set at 0.2, 0.3 and 0.3, respectively. While arbitrary, these values will serve to show that model results are insensitive to the choice of M when compared to full ICES assessments (see Results and Discussion). For those species for which there is no ICES assessment, annual values calculated from the Lorenzen equation were scaled to a conventional mean value of $M = 0.2$. Table 1 shows the value used for each species.

We fitted the model within a Bayesian framework using Stan (Carpenter et al., 2016) with the interface rstan (Stan Development Team, 2016). This requires priors to be specified for the model parameters. For all the error distributions, σ, and the log of survey catchability, q, uniform priors were used. For the initial fishing mortality, the F from ICES assessments of the target species (cod, haddock, whiting, plaice and sole) in 1983 was examined which gave a range of 0.5–1.0. A weakly informative lognormal prior was then chosen with a mean of log(0.7) and a standard deviation of 0.5. The choice of prior to the initial value of F is important because there is likely to be some confounding of F and q. An informative prior is necessary under these circumstances, or q must be specified for identifiability. This problem is elaborated further in the Discussion.

Priors were set for bulk discarding proportion, ρ, related to the market value of the species. It was assumed that for high-value species bulk discarding is very low and these were given a beta (1.2, 24) prior that has a mean of 0.04 and is highly informative. For the low-value species, a prior of beta (2, 3) was used which gives a mean of 0.4 and is only weakly informative. Table 1 shows the q prior used for each species.

In the case of mean log recruitment (Equation 2.3), we expressed this quantity as a proportion, p_r, of the mean survey index, $\bar{u}$, over the time series:

TABLE 1 Species included in the analysis with corresponding values used to configure the model. The EMLS is taken from Heath and Cook (2015). For skates and rays, there is no relevant length at maturity for the group so the SSB values in Figure 3 represent total biomass. The last two columns show the stock–recruitment curve and its associated R^2 when fitted to the model output. ICES stock areas are 3a = Skagerrak, 4 = North Sea, 6 = West of Scotland, 7d = Eastern channel. The greater North Sea is the combined area 4 + 3a

Stock	Acronym	Species	ICES stock area	M	Length at 75% maturity	EMLS (cm)	ρ prior	MCMC samples	Stock–recruitment curve	R^2 for stock–recruitment model
Cod	cod	*Gadus morhua*, Gadidae	3a + 4 + 7d	0.2	68.5	35	Beta (1.2, 24)	200,000	Ricker	0.354
Haddock	had	*Melanogrammus aeglefinus*, Gadidae	3a + 4	0.3	27.5	30	Beta (1.2, 24)	200,000	Ricker	0.074
Whiting	whi	*Merlangius merlangus*, Gadidae	3a + 7d	0.3	21.5	27	Beta (1.2, 24)	200,000	Beverton–Holt	0.058
Plaice	ple	*Pleuronectes platessa*, Pleuronectidae	4	0.1	25.5	27	Beta (1.2, 24)	200,000	Beverton–Holt	0.016
Sole	sol	*Solea solea*, Soleidea	4	0.1	22.0	24	Beta (1.2, 24)	100,000	Beverton–Holt	0.152
Turbot	tur	*Scophthalmus maximus*, Scophthalmidae	3a + 4	0.2	25.0	30	Beta (1.2, 24)	50,000	Beverton–Holt	0.000
Megrims	meg	*Lepidorhombus whiffiagonis*, *Lepidorhombus boscii*, Scophthalmidae	3a + 4 + 6	0.2	25.5	20	Beta (1.2, 24)	100,000	Ricker	0.031
Witch	wit	*Glyptocephalus cynoglossus*, Pleuronectidae	3a + 4	0.2	42.5	28	Beta (1.2, 24)	200,000	Ricker	0.103
Lemon sole	lso	*Microstomus kitt*, Pleuronectidae	3a + 4	0.2	18.5	25	Beta (1.2, 24)	200,000	Ricker	0.005
Common dab	cda	*Limanda limanda*, Pleuronectidae	3a + 4	0.2	11.5	25	Beta (2, 3)	200,000	Ricker	0.002
Flounder	flo	*Platichthys flesus*, Pleuronectidae	3a + 4	0.2	18.0	27	Beta (2, 3)	200,000	Ricker	0.056
Brill	bri	*Scophthalmus rhombus*, Scophthalmidae	3a + 4	0.2	25.5	30	Beta (1.2, 24)	50,000	Ricker	0.054
Anglerfish	ang	*Lophius piscatorius*, *Lophuis budegassa*, Lophiidea	3a + 4 + 6	0.2	29.0	32	Beta (1.2, 24)	100,000	Ricker	0.029
Ling	lin	*Molva molva*, Lotidae	3a + 4	0.2	48.5	63	Beta (1.2, 24)	50,000	Beverton–Holt	0.002
Spurdog	spu	*Squalus acanthius*, Squalidae	3a + 4	0.2	70.0	50	Beta (1.2, 24)	100,000	Beverton–Holt	0.003
Pollack	pol	*Pollachius pollachius*, Gadidae	3a + 4	0.2	40.0	30	Beta (1.2, 24)	100,000	Beverton–Holt	0.202
Halibut	hal	*Hippoglossus hippoglossus*, Pleuronectidae	3a + 4	0.2	70.0	45	Beta (1.2, 24)	30,000	Beverton–Holt	0.161
Mullet	mul	Mugilidae	3a + 4	0.2	15.0	16	Beta (1.2, 24)	30,000	Beverton–Holt	0.014
Tusk	tus	*Brosme brosme*, Lotidae	3a + 4	0.2	42.0	40	Beta (1.2, 24)	10,000	Beverton–Holt	0.009
Seabass	bas	*Dicentrarchus labrax*, Moronidae	3a + 4	0.2	31.5	36	Beta (1.2, 24)	30,000	Beverton–Holt	0.205
Gurnards	gur	*Chelidonichthys cuculus*, *Chelidonichtys lucernus*, Triglidae	3a + 4	0.2	22.5	30	Beta (1.2, 24)	30,000	Ricker	0.253
Grey gurnard	ggu	*Eutrigla gurnardus*, Triglidae	3a + 4	0.2	22.0	30	Beta (2, 3)	30,000	Beverton–Holt	0.150
Skates + rays	ray	Rajidae	3a + 4	0.2	NA	40	Beta (1.2, 24)	100,000	Beverton–Holt	0.000
Wolf-fish	wol	*Anarhichas lupus*, Anarhichadidae	3a + 4	0.2	35.0	30	Beta (1.2, 24)	20,000	Beverton–Holt	0.050

EMLS, effective minimum landing sizes; ICES, International Council for the Exploration of the Sea; SSB, spawning stock biomass; MCMC, Markov Chain Monte Carlo.

TABLE 2 Summary and definition of model equations

Population model		
2.1	$N_{t+1} = N_t \exp(-F_t - M_t) + R_{t+1}$	Projection equation for population, N, where F and M are fishing and natural mortality rates, R is recruitment and t is a subscript for year
2.2	$F_t \sim \text{Lognormal}\left(\log\left(F_{t-1}\right), \sigma_f\right)$	Fishing mortality follows a random walk, with process error standard deviation, σ_f
2.3	$R_t \sim \text{Lognormal}(\bar{r}, \sigma_r)$	Recruitment is log-normally distributed with mean $\bar{r}$ and process error standard deviation, σ_r
Observation equations		
2.4	$L_t = (1 - p_t) F_t N_t \overline{w_t}(1 - \exp(-Z_t))/Z_t$	Landed biomass, L, is given by the Baranov equation where $\bar{w}$ is the mean weight of an individual fish, $Z = F + M$ and p is the proportion of fish discarded
2.5	$u_t = qN_t$	The survey index, u, is proportional to the population, N, with a constant catchability, q
Observation errors		
2.6	$\hat{u}_t \sim \text{Lognormal}\left(\log\left(u_t\right), \sigma_u\right)$	The observed survey index, $\hat{u}_t$, has a lognormal error distribution with standard deviation σ_u
2.7	$\hat{L}_t \sim \text{Lognormal}\left(\log\left(L_t\right), \sigma_L\right)$	The observed landings, $\hat{L}_t$, have a lognormal error distribution with standard deviation σ_L
Constants used in the model calculated directly from the survey data		
2.8	$\overline{w}_t = \frac{\sum_l u_{l,t} w_l}{u_t}$	The mean weight, $\overline{w}_t$, is calculated by summing over the biomass at length, l, in the survey and dividing by the total number of fish
2.9	$p_t = \pi_t + \rho_t - \rho_t \pi_t$	Proportion of catch discarded is a function of the proportion discarded by size, π_t, and the proportion discarded by bulk, ρ_t
2.10	$\pi_t = \frac{\sum_l u_{l,t} w_l \pi_l}{u_{l,t} w_l}$	Proportion of catch biomass discarded by size, π_t, in year t is derived from the proportion of fish discarded at length, π_l

TABLE 3 Equations used to calculate the derived quantities, spawning stock biomass (SSB) and discards after fitting the model

3.1	$SSB_t = m_t u_t \overline{w}_t / q$	SSB, is a function of proportion mature biomass, m_t, using fitted estimates of survey abundance and catchability
3.2	$m_t = \frac{\sum_l u_{l,t} w_l m_l}{u_{l,t} w_l}$	Proportion of mature biomass, m_t, in year t is derived from the proportion of fish mature at length, m_l, calculated from the survey data
3.3	$D_t = p_t L_t / (1 - p_t)$	Discards, D_t, are calculated from the fitted landings and proportion discarded

$$\bar{r} = \log(p_r \bar{u}) - \log(q)$$

We then set a uniform prior on p_r.

For each species data set, four MCMC chains were run with the burn-in period determined by increasing the number of iterations, n, until the Rhat statistic was equal to one (Gelman & Shirley, 2011) for all parameters when using the last $n/2$ samples. This gave burn-in periods ranging from 5,000 to 100,000 iterations, which are shown for each species in Table 1.

After fitting the model, we calculated SSB and the discards using equations in Table 3. For simplicity, because some of the maturity samples were sparse making estimation of a conventional length-dependent model difficult, we assumed $m_l = 0$ for lengths below which maturity was less than 75% and $m_l = 1$ for those above. The 75% maturity lengths are given in Table 1.

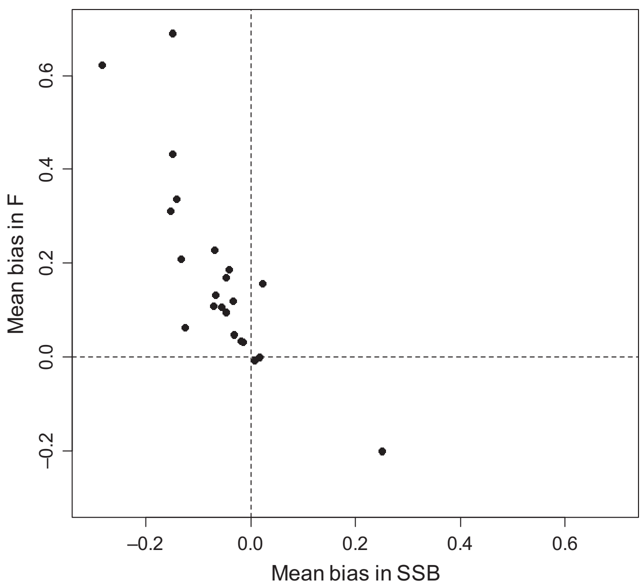

FIGURE 2 Results of the retrospective analysis for bias in F and spawning stock biomass (SSB) using Mohn's rho. Points show the result for each species plotted in phase space. The dashed lines correspond to zero bias in each quantity. Unbiased assessments should lie close the intersection of the lines

2.4 | Model validation

To show that the model adequately estimates stock trends, we considered three aspects of model performance. These were (i) retrospective analysis (Mohn, 1999) to assess model consistency (ii) that the trends in recruitment, SSB and *F* were consistent with those stocks assessed by ICES with a full age-based analysis and (iii) that the estimated discards were consistent in scale and trend with those species for which real observations are available.

2.4.1 | Retrospective analysis

Stock assessment models frequently show retrospective bias where the addition of one more years' data results in a systematic upward or downward revision of SSB and *F*. Retrospective analysis is a widely applied test to evaluate this problem where the assessment is repeated

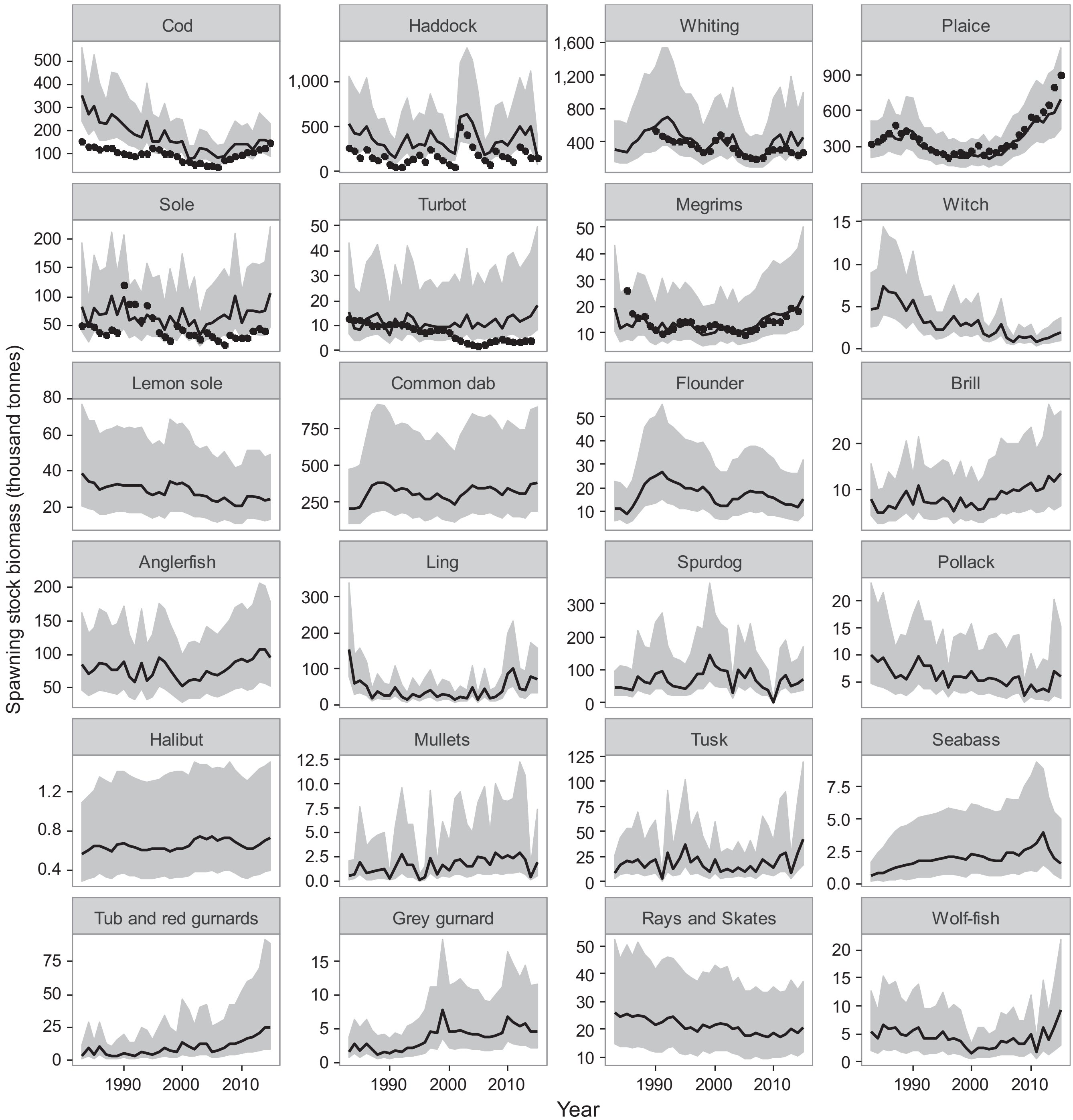

FIGURE 3 Spawning stock biomass. Lines and shaded area show the median and 95% CI estimated from the model. Dots show the values from the International Council for the Exploration of the Sea (ICES) assessments, where available. In the case of megrim, the ICES values, which are reported only on a relative scale, have been rescaled to the mean of the model used in this paper

successively dropping off the end-year data point. The end-year estimates of F and SSB are then compared to the values obtained from the full data set. We ran a retrospective analysis over 10 years for all species and calculated Mohn's rho (Mohn, 1999) which measures the mean bias relative to the full data assessment.

2.4.2 | Comparison with fully assessed species

ICES performs full assessments on cod, haddock, whiting, plaice, sole and turbot which provide estimates of recruitment, SSB and mean F. In addition, a surplus production model is used to assess megrim and provides an index biomass and fishing mortality. We compared trends in these quantities using the assessments reported in ICES (2015b, 2015c). We also compared the LSD model estimates to those from Piet et al. (2009) by converting their percentage yield-biomass ratios (YBR) to F by solving the Baranov equation:

$$\mathrm{YBR} - F(1 - \exp(-F - M)/(F + M) = 0$$

Although in principle the model described in this study gives estimates that are comparable to those of ICES, there are likely to be differences in scale. This is partly because of differences in assumptions about natural mortality and also because in the case of F, the measure used by ICES is a simple mean calculated over the ages of full selection, whereas the F calculated in the LSD model is effectively an abundance-weighted average over all ages. It is likely to be lower than the ICES F if younger fish have lower selectivity in the fishery. As an additional metric, we therefore calculated the correlation coefficient between the ICES values and the model estimates as a measure of similarity of trends.

2.4.3 | Discards

For cod, haddock, whiting and plaice comprehensive estimates of discards are provided by ICES (2015b) covering the whole period of this analysis. For sole, a few estimates exist but most estimates are derived from the assessment model (Aarts & Poos, 2009). In addition, there are estimates for 1–3 recent years for turbot, megrim, witch, lemon sole, common dab, flounder, brill and anglerfish. We compared these values with those estimated from the LSD model.

2.5 | Analysis of model output

For each species assessed, we examined changes in landings, discards, recruitment, SSB and F over the three decades 1986–1995, 1996–2005 and 2006–2015. We took the mean value for each quantity over consecutive decades and calculated the relative change. Thus, if x_t and x_{t+1} are the means for the first and second decades, we calculated the ratio $(x_{t+1} - x_t)/x_t$ and so on for all decades and relevant quantities.

As formulated, the model assumes random recruitment to avoid forcing a structural dependence on SSB. It is also important to bear in mind that recruitment in Equation 2.1 in Table 2 does not necessarily correspond to a true year class spawned from a specific SSB; it simply expresses the number of fish entering the population as seen by the survey and may encompass more than 1 year class. Nevertheless, it is of interest to see whether there is any apparent relationship between estimated recruitment and estimated SSB. We therefore fitted a standard Ricker (Ricker, 1954) or Beverton–Holt curve (Beverton & Holt, 1957) to the posterior stock–recruit estimates and present results for the function that explained the most variance in recruitment.

3 | RESULTS

Generally, the model fitted both the survey index and landings data well (Figure S1, model fit to the survey index and Figure S2, model fit to the landings data). Typically landings showed the superior fit but with turbot, mullets and grey gurnard as exceptions. The retrospective analysis shows the model has a tendency to overestimate F and underestimate SSB (Figure 2). The bias in SSB is generally lower than that for F and less than 0.1 in 14 of the 24 species. Where large bias exists, this is mainly due to a change in scale in the estimates each time a data year is dropped (Figure S3, retrospective analysis of F and SSB) rather than revisions of the time series trend. This indicates that the estimated trends are more robust but that the scaling is sensitive to the range of data used in the model.

The trends in SSB of the species assessed by ICES are similar to the model estimates and are highly correlated with the exception of sole and

TABLE 4 Correlation coefficients between model estimates of F, SSB and recruitment and the corresponding ICES assessment values

Species	Log recruits	SSB	F	Discards
Cod	0.462 (.0068)	0.720 (.0000)	0.761 (.0000)	0.6786 (.0000)
Haddock	0.822 (.0000)	0.825 (.0000)	0.894 (.0000)	0.7986 (.0000)
Whiting	0.375 (.0590)	0.753 (.0000)	0.585 (.0021)	0.6271 (.0001)
Plaice	0.582 (.0004)	0.955 (.0000)	0.885 (.0000)	0.7351 (.0000)
Sole	0.279 (.1226)	0.242 (.1829)	0.615 (.0002)	0.3327 (.0628)
Turbot	0.137 (.4534)	0.042 (.8176)	0.221 (.2239)	NA
Megrim	NA	0.541 (.0020)	0.785 (.0000)	NA

ICES, International Council for the Exploration of the Sea; SSB, spawning stock biomass.
p values are shown in parentheses.

turbot (Figure 3, Table 4). For turbot, the model shows a similar trend for the early period but diverges markedly in recent years. Significant correlations are also evident in log recruitment in the case of cod, haddock, plaice and possibly whiting (Table 4) indicating that the model can identify some year class signal from the survey and landings data.

With the exception of turbot, fishing mortality for the assessed species also shows a high correlation between the ICES and model values (Figure 4, Table 4), although as expected, the model estimates are often lower than those of ICES. Values for *F* in 1983 are shown in Figure 5 compared to the assumed prior. Most estimates lie within the 95% CI of the prior but generally have a lower median value. The posterior 95% CI for *F* from the LSD model includes 62% of the values obtained by Piet et al. (2009). In some cases, the Piet et al. values are extremely large and appear to be unrealistic. The LSD estimates are overall less variable reflecting the influence of the prior.

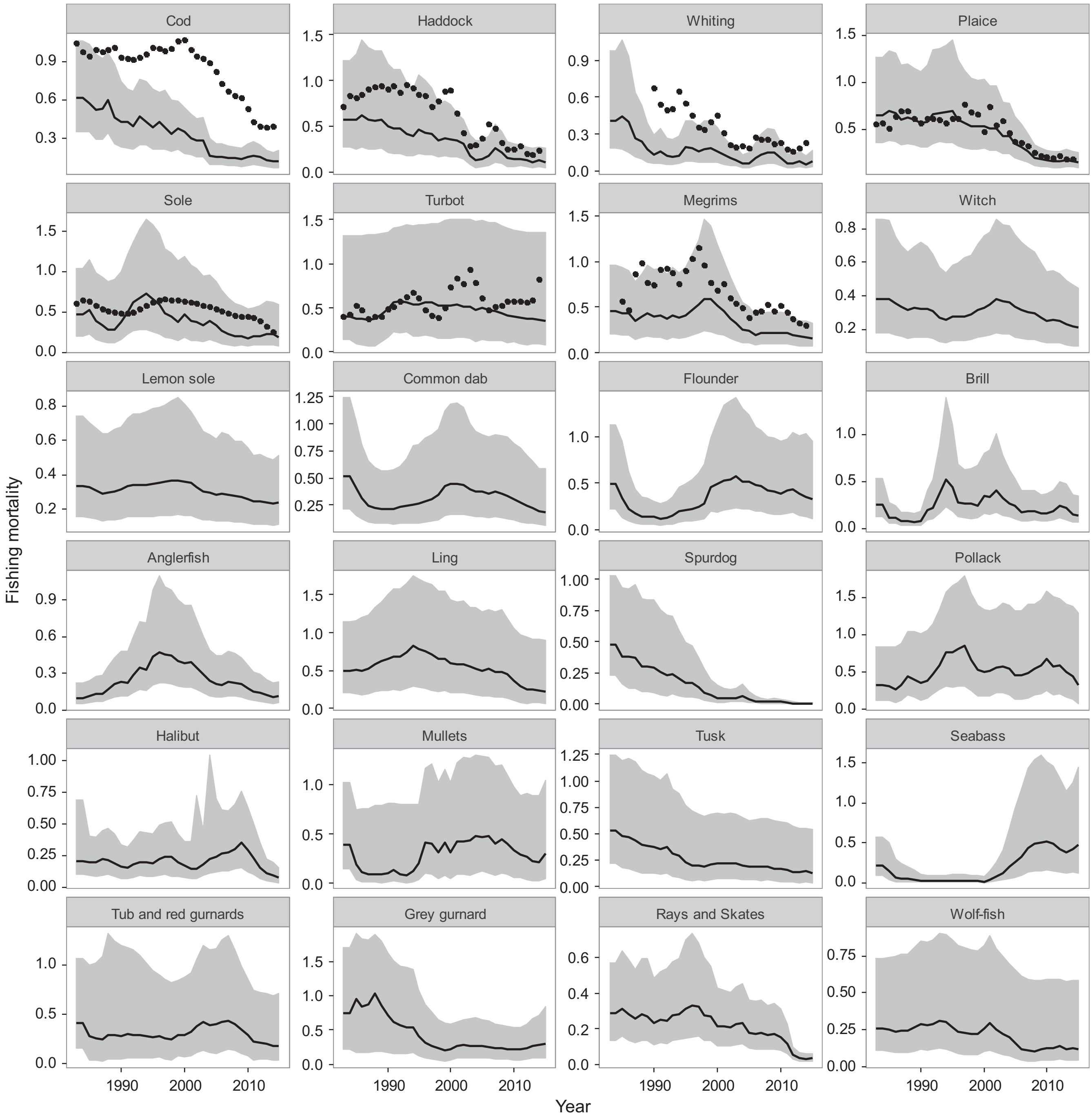

FIGURE 4 Fishing mortality. Lines and shaded area show the median and 95% CI estimated from the model. Dots show the values from the International Council for the Exploration of the Sea (ICES) assessments, where available. The ICES values represent the mean *F* over a conventional age range and do not necessarily correspond to the same scale as the current model

Discard estimates derived from the model are compared to the available observations in Figure 6. As these observations were not used in fitting the model, the agreement between the estimates is a measure of the adequacy of assumption of size and bulk-related discarding (Table 2, Equations 2.9 and 2.10). In most cases, the model estimates the correct scale of discarding, and where a full-time series of data are available, there is a high correlation with the observed values (Table 4). There is a tendency to underestimate plaice and megrim discards and overestimate quantities for sole.

Between the first and second decades, both the landings and discards tended to increase but this has reversed in that last decade (Figure 7). Similarly, the fishing mortality was increasing for most species in the early period but now is more typically declining, with the exception of seabass. The converse is true for the change in SSB where the majority of species now show an increase following the declines in earlier years. However, in the case of recruitment, there is an overall deterioration with the more recent decade showing a lower mean for most species.

There is an indication of a stock–recruitment relationship for some of the species (Figure 8). The Beverton–Holt curve has the largest R^2 for over half the species (Table 1), although Ricker is preferred for a number of cases including witch and gurnards. For both haddock and anglerfish, recruitment estimates seem to follow the descending limb of a Ricker curve. As with most stock–recruit estimates, the R^2 values are generally small, although in seven cases, they are 15% or more.

4 | DISCUSSION

An important aspect of the LSD model is the potential confounding effect of survey catchability q and F. This can be seen from Equations 2.4 and 2.5 in Table 2 where F and q appear in the equations for landings. We used a weakly informative prior on F to assist the model in estimating both F and q. The choice of mean for the prior on the initial F value is influential on the level of estimated F, although it simply scales the trend. Where initial values can be accurately drawn by analogy with fully assessed species, this level should be satisfactory but it does mean that the analysis should be interpreted in relation to trends rather than absolute values. The retrospective analysis re-enforces this conclusion as truncating the data range changes the scale of the estimates. The independent method of estimating F used by Piet et al. (2009) generally gives values comparable to the level in the LSD model, offering some external support for the scale of the estimates. Their method in essence takes the alternative approach of estimating q and then scaling this by fishing effort. Consequently, their estimates are conditioned on an assumption on the level of q rather than a prior on F.

The confounding effect of F and q also has implications for the assumption of constant survey catchability. Clearly, if q varies, or worse of it exhibits a trend, these departures from the assumption will directly affect the estimates of F. It is quite likely that survey q does change since, although survey protocols are reasonably standard, there have been changes over the years to participating vessels and

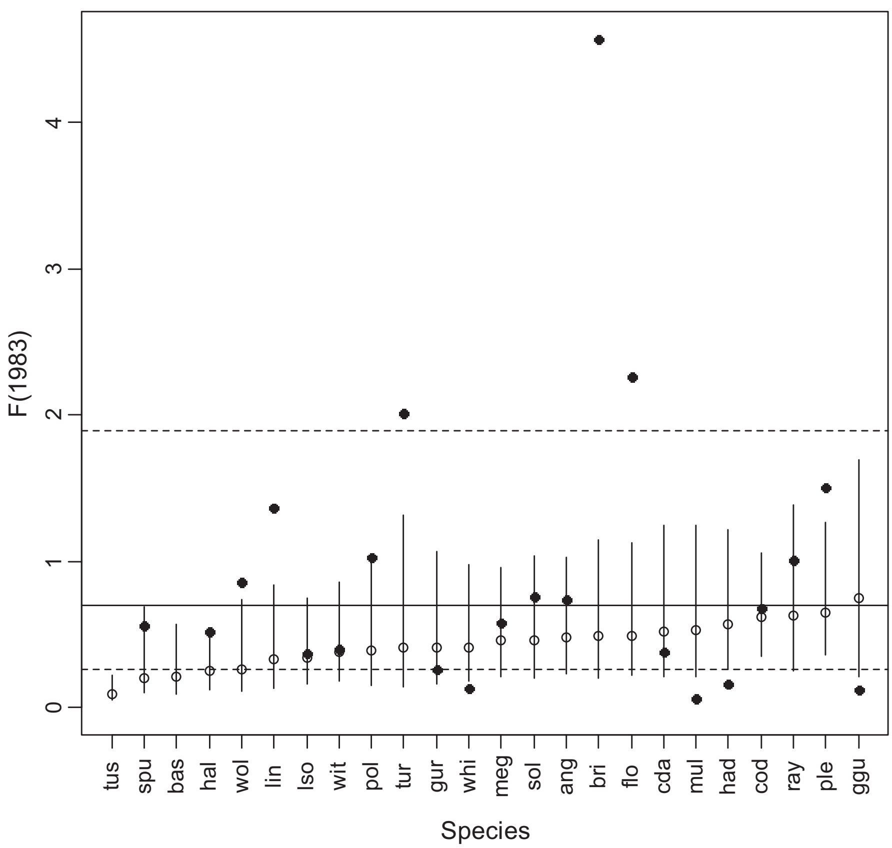

FIGURE 5 Fishing mortality estimates from the Landings-Survey-Discard model in 1983 (open circles) compared to estimates from Piet et al. (2009) (filled circles). Horizontal lines show the median (solid line) and 95% CI (dashed line) of the prior used on F. Vertical lines on the open circles show the 95% CI on the posterior estimates of F in 1983

length of tow. Many assessments assume constant *q* for the IBTS but it remains a source of uncertainty. Nevertheless, for ICES-assessed species, trends in *F* follow the LSD estimates suggesting that variability in *q* may not be a serious problem.

Comparing our results with ICES stock assessments shows that the LSD model is able to capture many of the main elements of fully age-structured analyses that make use of more comprehensive data including catch-at-age and discard data. Typically, the trends in SSB and *F* are similar, and for some species, this is also true of recruitment. While the trends are similar, there are differences in scale. Fishing mortality in the LSD model is abundance-weighted, while in the conventional age-structured models used in ICES assessments, it is not and this will result in scale differences. In addition, there are differing assumptions about natural mortality. *M* acts primarily as a scaling constant with little effect on the annual changes. The choice of *M* for cod, haddock and whiting illustrates this point. Even when making an arbitrary (and

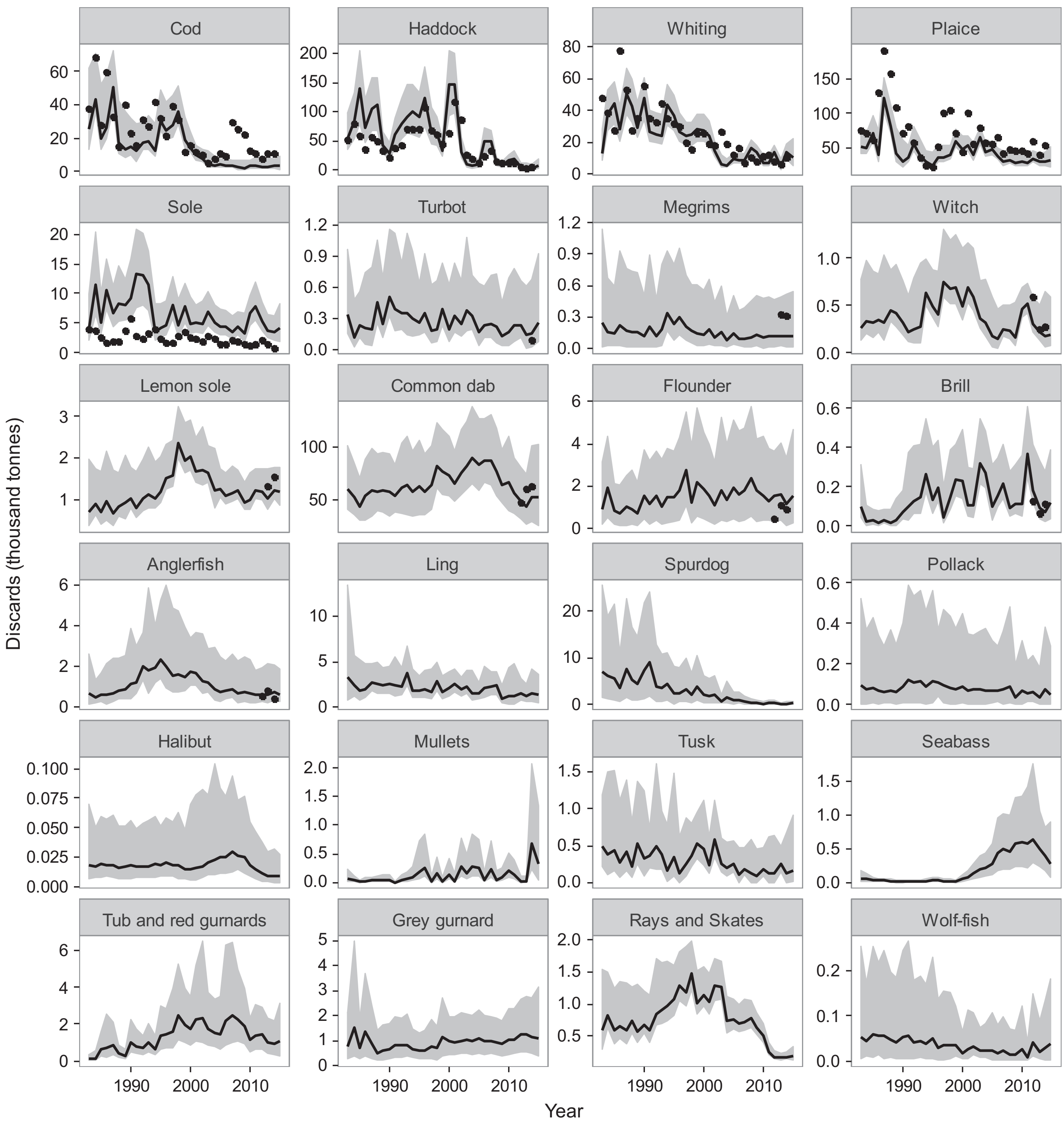

FIGURE 6 Discards. Lines and shaded area show the median and 95% CI estimated from the model. Dots show the observed values reported by International Council for the Exploration of the Sea (ICES), where available. ICES discard data were not included in the model fit

arguably incorrect) assumption about M, these stocks show close agreement between trends estimated by the LSD model and ICES assessments. Although we are interested in absolute values for discards, these estimates are not sensitive to the choice of M as they are, in effect, estimated as a proportion of the landings which are fixed (see Figure S4, two model runs for whiting with M = 0.3 and M = 0.6).

There is a notable difference between the LSD estimated stock trends compared to the ICES assessment for turbot and to some degree sole. For turbot, the survey index shows little long-term trend, while the landings show a decline (Figures S1 and S2). The model interprets this as a decline in F and is at variance with the ICES assessment. The IBTS survey indices show a different trend compared to beam trawl surveys used in the ICES assessment (ICES, 2017a) and are the likely cause of the discrepancy. A similar problem may affect sole as additional surveys are used in the ICES assessment.

As well as reflecting stock trends, the LSD model also estimates discards that bear a close resemblance to actual observations. For haddock, whiting and plaice, the model estimates both the trend and the scale of the discards well, and as these values are generated largely from the assumption of discards related to the EMLS, it indicates that size-based discarding is likely to be the main process responsible. Recent trends indicate that the total quantity of discards has declined reflecting the reduction in fishing mortality and lower catches.

While cod discards are estimated well for some of the time period, there are differences in the more recent years where observed discards are much larger than those estimated from the model. It is known that during this period catch restrictions caused increased discarding by bulk and is the likely cause of the difference (Heath & Cook, 2015). In principle, the model should be able to capture this process but the strongly informative prior on bulk discarding for this species (and which assumes it is small) forces the estimates to reflect only size-based discards. Relaxing this assumption for these years would improve model fit. In the case of sole, the LSD estimates are much larger than ICES values, probably unrealistically so, but the latter are assessment model generated values rather than true observations making comparisons difficult. A few data points from recent years exist for a number of other species, and the model is able to capture the scale correctly with

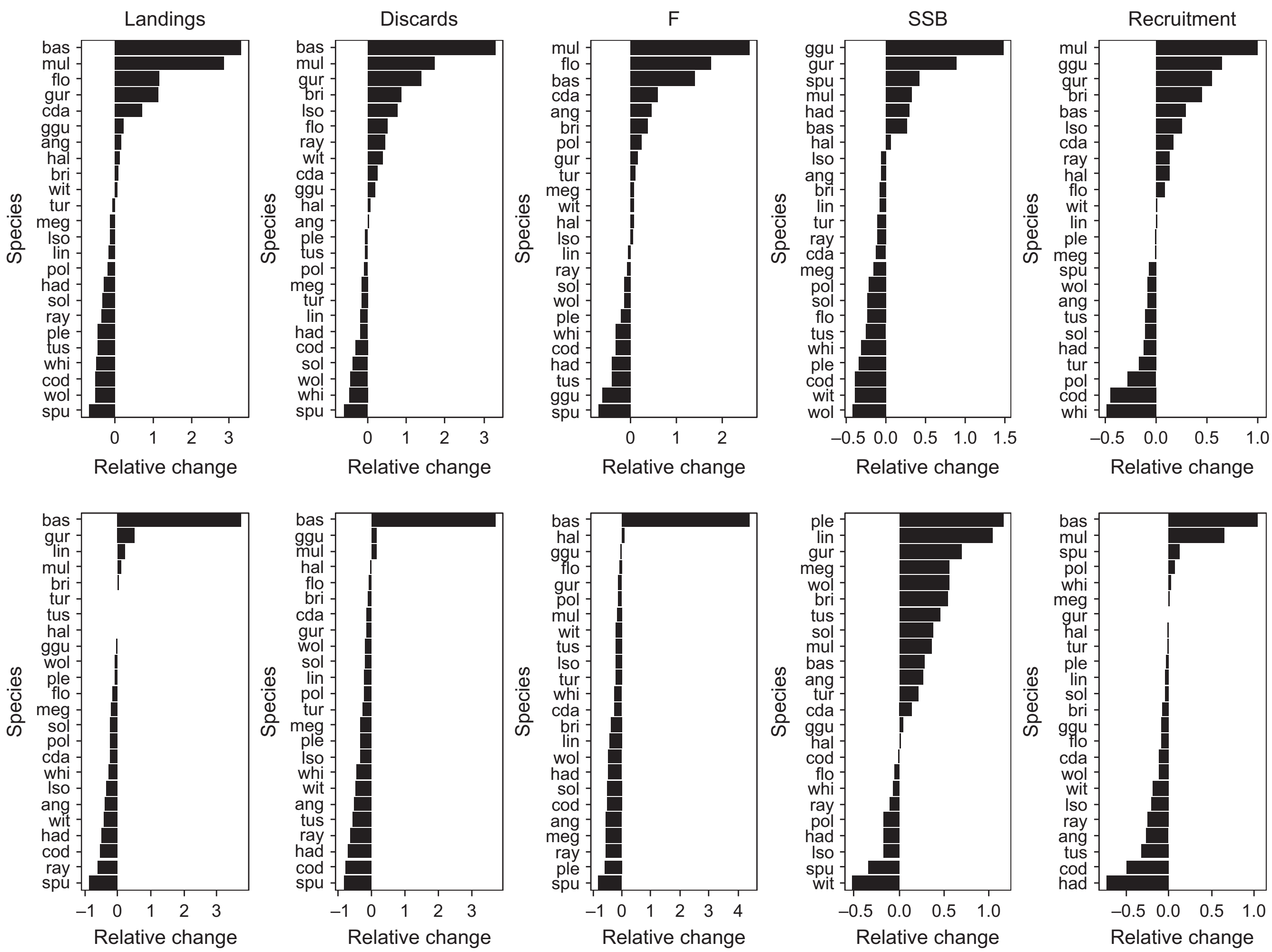

FIGURE 7 Relative change in decadal mean landings, discards, F, spawning stock biomass (SSB) and recruitment, across three decades. Upper row shows the change between the first and second decades, and the lower row shows the change between the second and third decades. Each panel is ordered by rank

all the observations lying within the 95% CI. Overall, the model is able to account for discards in a way that is consistent with the limited available data but if there is sporadic bulk discarding the discard estimates from the model may be unreliable. As the time series of discard data develops in future years, it will be possible to use the observations in the model and estimate the EMLS and

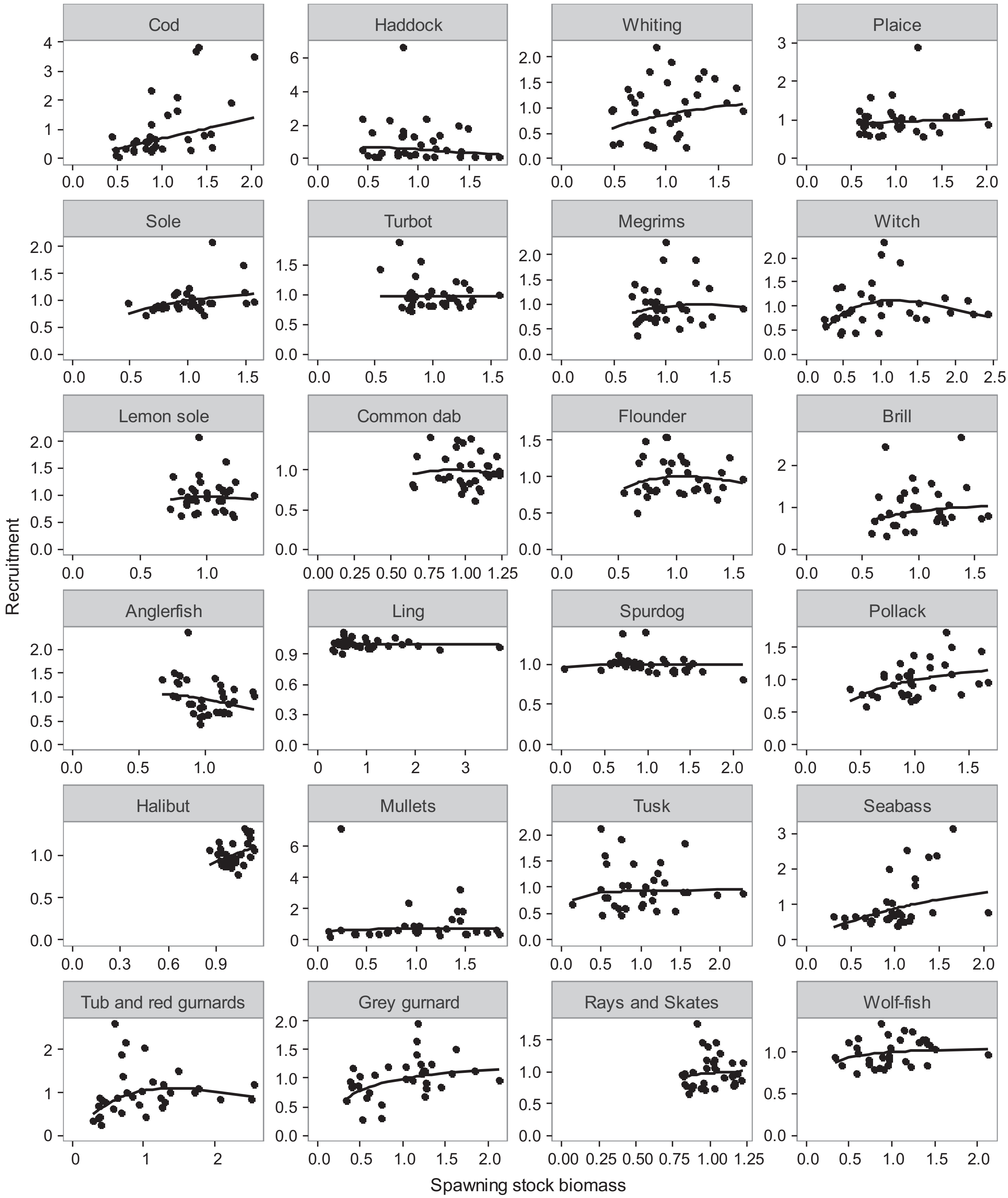

FIGURE 8 Stock–recruitment relationships fitted to the model output using either Beverton–Holt or Ricker relationships. Coefficients of variation are shown in Table 1. Data on each axis are scaled to the series mean

annual values of ρ internally rather than specifying them either as a constant or an informative prior.

Until 2016, discarding was necessary to comply with minimum size regulations and catch quota limits. Changes to European Union regulations have meant that from 2016 many fishing fleets are subject to a "Landing Obligation" which requires all fish caught to be landed and is in effect a ban on discards. The obligation does not apply to all species but it does affect most stocks subject to a Total Allowable Catch. While the regulation does not affect the analysis presented here as it deals with an earlier period, it means that in the future the size-based assumption used to derive much of the discards in the model may no longer be appropriate. The most recent ICES assessments that cover 2016 do not yet show any change in discarding behaviour (ICES, 2017a, 2017b) but it is likely to change in the future.

The LSD model provides estimates of time trends in fishing mortality and SSB as well as recruitment and offers a more comprehensive overview of stock dynamics than simple survey trends. The historical perspective shows that during the late 20th century, most bycatch species were in decline with rising fishing mortality rates. This has reversed in the last decade. However, recent recruitment appears, if anything, to have deteriorated despite increasing SSB (Figure 7, lower panel). If these trends are correctly estimated, it suggests that reduced fishing mortality rate is the principal cause of increasing biomass but that future increase may be limited by lower mean recruitment. The species with the greatest negative change in recruitment are those typically associated with more northerly distributions (e.g. cod, haddock, tusk and lemon sole), while those with the greatest positive change have distributions that extend further to the south (seabass, mullets, whiting and pollack) (Figure 7). Such changes are consistent with the effects of climate change, which in the North Sea favours more southern species (Beare et al., 2004; Blanchard et al., 2005; Drinkwater, 2005). Cook and Heath (2005) estimated negative effects of temperature on recruitment for cod, plaice and sole but a positive effect for whiting and this appears to be consistent with the current analysis.

Skates, rays and dogfish are of concern to conservationists as their size makes them vulnerable to capture and their reproductive rates tend to be lower than bony fish (Dulvy, Metcalfe, Glanville, Pawson, & Reynolds, 2000). At least four species occurring in the North Sea, common skate (*Dipturus batis*, Rajidae), cuckoo ray (*Leucoraja circularis*, Rajidae), shagreen ray (*Leucoraja fullonica*, Rajidae) and spurdog are listed in threatened categories by the IUCN (2014). As landings data for skates and rays do not adequately distinguish between species, the analysis presented here groups them all into a single category. As a group, the biomass shows a long-term decline until 2005 when there is some increase (Figure 3). This change in abundance is likely to reflect a change in the species composition with larger species such as common skate declining, while smaller species such as thornback ray (*Amblyraja radiata*, Ragidae) are increasing (Walker & Hislop, 1998). Overall, the fishing mortality on skates and rays appears to have reduced substantially from around 0.3 to less than 0.05 although the larger, more vulnerable species may still be at risk. For spurdog, there is no strong trend in SSB over time but the mean abundance in the most recent decade is about 40% lower than previously despite a large reduction in fishing mortality. These trends are similar to those estimated by De Oliveira, Ellis, and Dobby (2013) for the Northeast Atlantic in the years 1983–2005.

There is some, albeit weak, evidence of stock–recruitment relationships where lower recruitment is associated with lower SSB (witch, flounder, brill, pollack, seabass and gurnards). A stock–recruitment submodel could be included in the LSD model and the parameters estimated internally. This could provide a basis for calculating reference points and making forward projections. However, doing so would require modelling the effects of fishing mortality on mean weight as higher *F* would be expected to result in fewer older and larger fish leading to a lower mean weight. For projections under status quo *F*, this problem may be minor but where large departures from status quo are considered significant bias may occur.

Trends in the bycatch species show many similarities with those of the principal target species with a period of high exploitation and declining biomass in the late 20th century but an improvement in recent years. Typical values of *F* do not show major differences in magnitude from the target species, although this may be driven by the prior distribution used in the model and assumptions about *M*. The limited discard data available are consistent with predominantly size-based selection, at least for the species considered of high value. Thus, while non-selective discarding in bulk caused bycatch regulations may occur, it does not appear to be prevalent. It seems therefore that the assessments of target species give a broad indication of the likely exploitation and biomass trends in bycatch species, although clearly there will be individual differences depending on the species and fisheries concerned.

Our analysis stops short of classifying stock status according to MSY criteria. Methods such as those of Froese et al. (2012), Martell and Froese (2013) or Froese, Demirel, Coro, Kleisner, and Winker (2016) could be used, although these rely on estimates of resilience and may require conditioning assumptions about the historical development of the fishery. In our analysis, fishery-independent data in the form of a trawl survey are invaluable in providing more robust indicators of stock trends while accounting for fish discarded. Importantly, the analysis shows that the decline in landings is not due to declining stock biomass. This does not mean stocks are sustainably exploited, merely that their condition has improved. Nevertheless, it would be possible to extend the model to include a stock–recruitment relationship and attempt a full estimation of MSY.

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