

Development of the EcoQO for the North Sea fish community

Simon P. R. Greenstreet^{1*}, Stuart I. Rogers², Jake C. Rice³, Gerjan J. Piet⁴, Emma J. Guirey¹, Helen M. Fraser¹, and Rob J. Fryer¹

¹Marine Scotland–Science, Marine Laboratory, Aberdeen AB11 9DB, UK

²Centre for Environment, Fisheries and Aquaculture Science, Pakefield Road, Lowestoft, Suffolk NR33 0HT, UK

³Department of Fisheries and Oceans, 200 Kent Street, Ottawa, ON, Canada K1A 0E6

⁴Wageningen IMARES, Haringkade 1, NL-1976 CP IJmuiden, The Netherlands

*Corresponding Author: tel: +44 1224 295417; fax: +44 1224 295511; e-mail: greenstreet@marlab.ac.uk.

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Development of the Ecological Quality Objective (EcoQO) for the North Sea demersal fish community is described. Size-based metrics were identified as the most effective indicators of the state of the community, but such metrics are also sensitive to environmental influence. Redefining the large fish indicator (LFI) produced a metric more sensitive to fishing-induced change and therefore more useful to managers. Fish stocks were thought to be exploited at a sustainable rate in the early 1980s, so in a process echoing the precautionary approach to fish stock management, this was considered the reference period for the LFI, suggesting a value of 0.3 as the appropriate EcoQO. The LFI declined from around 0.3 in 1983 to 0.05 in 2001, followed by a recovery to 0.22 in 2008. However, analyses of the longer-term groundfish survey data suggest that, even were fishing pressure to be reduced to early 20th century levels, the LFI would be unlikely to rise much above a value of 0.3. The response of the LFI to variation in fishing pressure suggested a more complex relationship than anticipated, underscoring the need for operational theoretical size-resolved multispecies fish community models to support management towards broader ecosystem objectives.

Keywords: demersal fish community, Ecological Quality Objectives, large fish indicator (LFI), North Sea, OSPAR, state indicators.

Introduction

The need to develop and implement an ecosystem approach to marine management (EAMM) was formally recognized at the Bergen North Sea Intermediate Ministerial Meeting in 1997 (Misund and Skjoldal, 2005). The organization with competence to undertake this task, the Oslo/Paris Commission (OSPAR; Johnson, 2008), subsequently recognized nine specific Ecological Quality Issues (EcoQIs) and set about establishing Ecological Quality Objectives (EcoQOs) for these as the basis for an EAMM in the North Sea (Heslenfeld and Enserink, 2008). Appropriate indicators of “state”, illustrating changes in the status of each EcoQI over time, were required to support each EcoQO (Johnson, 2008). One of the nine EcoQIs identified was fish communities. Many univariate community state metrics exist, including metrics of abundance and biomass, species richness and species evenness, and life history, trophic level, and body size composition (Piet and Jennings, 2005; Greenstreet and Rogers, 2006). In establishing an EcoQO for the North Sea fish community, two specific questions need to be addressed: which metric would be the best state indicator and how might the management target, the EcoQO, be set for this indicator?

Here, we review the background to the selection and subsequent development of the univariate indicator for the North Sea demersal fish community EcoQO and describe the process by which the EcoQO itself was established. Recent performance of the indicator is examined, and prospects for successfully

achieving the EcoQO are considered. We also review briefly the modelling needed to underpin formal scientific advice to management. The process described in this North Sea case study should be applicable to other ecosystem components and regions and hence promote a more unified approach to the implementation of an EAMM.

Development of the North Sea fish community state indicator

The International Council for Exploration of the Sea (ICES), asked to advise on indicators to support the North Sea fish community EcoQO, considered that the chosen metric would have to perform the state indicator role within an indicator-based management framework, e.g. the pressure–state–response (PSR) framework (Garcia and Staples, 2000; Figure 1). ICES established a set of criteria to select state indicators especially sensitive to changes in fishing pressure and insensitive to other drivers of change (Figure 1). These criteria also acknowledged the fact that to identify the appropriate management response (the reduction in pressure required to achieve the EcoQO for state), the mechanistic relationship between the state and fishing pressure indicators would need to be established clearly.

Because of political focus on conserving and restoring biodiversity, species diversity metrics had frequently been applied to groundfish survey data (Greenstreet *et al.*, 1999; Piet and Jennings, 2005; Greenstreet and Rogers, 2006). However,

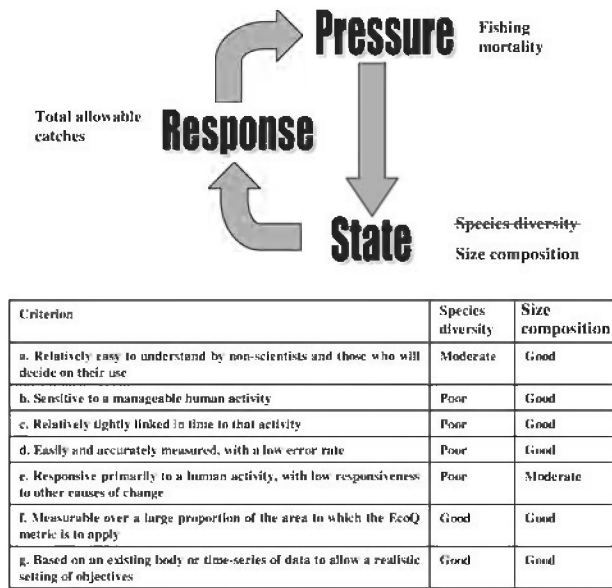


Figure 1. A schematic illustrating the PSR indicator-based management framework (after Garcia and Staples, 2000), with potential metrics suggested as examples in each role. Two possible types of state indicator are suggested; species diversity and size-composition metrics. The list of ICES criteria for identifying reliable state indicators (ICES, 2001a) and scores attributed to each type of metric (ICES, 2001b) are given for reference, illustrating the selection of size-composition metrics as the best performing indicators of state.

inconsistency in the methods employed prevented clear assessment of whether fishing actually influenced fish species diversity (Greenstreet and Piet, 2008), and the processes by which fishing might influence fish species diversity were also far from clear (Greenstreet, 2008). Hence, species diversity metrics scored poorly against the ICES criteria and were discarded as potential state indicators (ICES, 2001b; Figure 1). Life-history metrics, e.g. growth rate, ultimate body length, and age and length at maturity averaged across the community, were developed because of their theoretical relationship with changes in fishing mortality (Jennings *et al.*, 1999; Greenstreet and Rogers, 2000). These metrics, however, were deemed too complicated for use within a policy context (ICES, 2001b) and were likewise discarded (ICES, 2001a; Greenstreet, 2008).

The theoretical relationship between fishing mortality and fish population age (and hence size) composition was well-established (Beverton and Holt, 1957). Size-based trophodynamic theory also related reductions in large predator abundance, through reduced predation rates, to the increased abundance of small fish (Pope, 1991; Kerr and Dickie, 2001). The link between the observed shifts in the size composition of fish communities (Rice and Gislason, 1996; Daan *et al.*, 2005; Shin *et al.*, 2005) and increased fishing mortality was therefore strongly founded in theory. Consequently, size-composition metrics scored highly against the ICES criteria for good state indicators (ICES, 2001b; Figure 1), and as a result, “changes in the proportion of large fish and hence the average weight and average maximum length of the fish community” emerged as the Element of Ecological Quality for the fish community EcoQI at the Bergen 2002 North Sea Ministerial Conference (ICES, 2001a; Heslenfeld and Enserink, 2008).

Greenstreet and Rogers (2006) determined that just 5% (by number) of fish sampled over the entire 70 years of the Scottish August Groundfish Survey (SAGFS) time-series exceeded 30 cm in length. The proportion of fish >30 cm long was greatest in areas where fishing effort was low and stationary over time, whereas in areas of greatest fishing activity, because of significant long-term declines, the proportion of fish >30 cm was least. Concluding that a large fish indicator (LFI), defined as the “proportion by number of fish in the sample greater than 30 cm in length”, was sensitive to variation in fishing pressure, ICES (2006a) proposed short- and medium-term goals: to “halt the decline in the proportion of fish greater than 30 cm in length in survey catches immediately” and to “restore the proportion of fish greater than 30 cm in length to 1.4 times 1997 survey estimates by year X” (where year X is the date by which the target should be met). For the second indicator, ICES (2006a) expressed similar goals: to “halt the decline in the mean weight of fish in survey estimates immediately” and to “restore the mean weight of fish to 1.3 times 1997 survey estimates by year Y” (where again year Y is the date by which the target should be met). The medium-term goals were expressed in these terms because the SAGFS on which the two state indicators were based ceased in 1997. ICES (2006a) considered the early 1980s to be “the last period when ICES advice regarding the management of the exploited species was generally for the maintenance of *status quo* exploitation rates, suggesting that this was the last period when science experts considered fishing to be generally sustainable in the North Sea”. During the early 1980s, the values for the LFI and mean individual weight indicator (MIWI) were approximately 1.4× and 1.3× higher, respectively, than the values in 1997 (Greenstreet and Rogers, 2006). Even in the early 1980s, demersal stocks in the North Sea were heavily exploited (Worm *et al.*, 2009). The 1980s reference period therefore represented a pragmatic compromise between ecosystem and fisheries objectives.

Although the SAGFS could be used to define indicators and consider targets for them, because the survey was discontinued in 1997, it was no longer available for monitoring progress towards the targets. Such monitoring required the indicators to be based on alternative groundfish survey data. ICES currently coordinates two international bottom trawl surveys (IBTS) carried out annually in the first (Q1) and third (Q3) quarters of the year. These use standardized sampling gears and protocols, cover the entire North Sea (ICES Area IV), and aim to obtain two half-hour trawl samples from each ICES statistical rectangle (0.5° latitude by 1.0° longitude). Having (marginally) the longest time-series, from 1983 on (although Scotland continued to fish for 1 h rather than 30 min until 1999), the Q1 survey was considered the better dataset to fulfil this monitoring role (ICES, 2007a).

In keeping with previous studies applying univariate community metrics to North Sea groundfish survey data (e.g. Greenstreet *et al.*, 1999; Piet and Jennings, 2005; Greenstreet and Rogers, 2006), the North Sea fish community EcoQO only considers demersal fish (ICES, 2006a). In part, this is because the Grande Ouverture Verticale trawl used in the IBTS is more selective for bottom-dwelling than pelagic species.

Assessment of the performance of the two size-composition metrics (ICES, 2007a), repeated here using the updated Q1 IBTS database, revealed potentially serious flaws. Both metrics suggested recoveries in demersal fish size composition in the early 2000s that were short lived (Figure 2). This so-called recovery pulse was

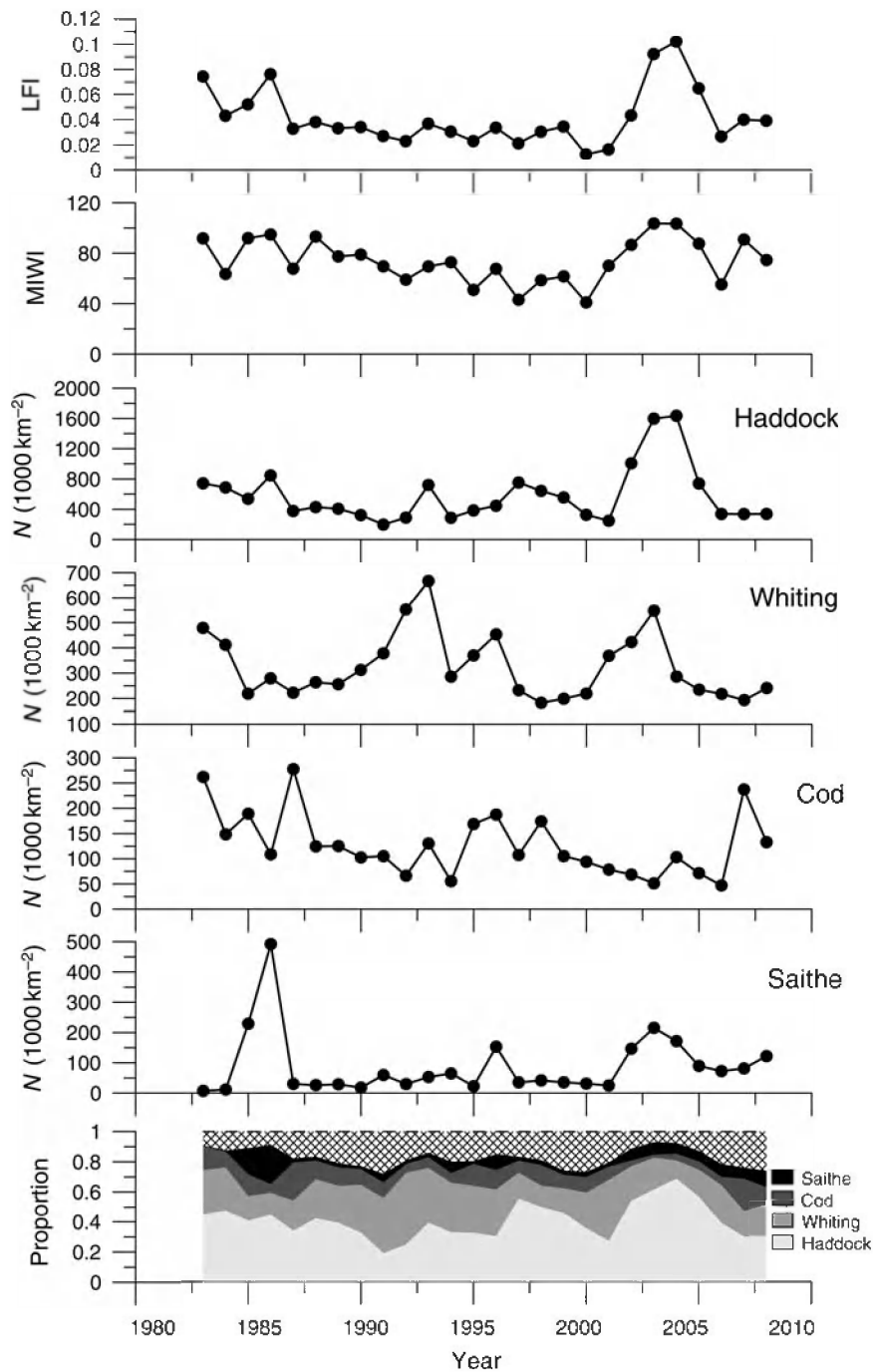


Figure 2. Variation over time in the proportion of LFI, defined as the proportion by number of fish >30 cm long, and the MIWI based on IBTS Q1 data. Temporal variation in the abundance (N) of haddock, whiting, cod, and saithe >30 cm long is also indicated, along with the proportion of all fish >30 cm long that these four species represent.

driven by exceptionally good recruitment of haddock (*Melanogrammus aeglefinus*). Four key gadoid species, including haddock, accounted for 72–93% of all fish >30 cm sampled over the full time-series. The 1999 haddock cohort was 2–10× more than any other cohort between 1983 and 2006 (ICES, 2007b). In 2002 and 2003, once these fish had grown to >30 cm, this cohort caused a marked increase in both size-composition metrics (Figure 2), but the recovery was not sustained beyond 2004 when the cohort became fully exploited. The two

metrics therefore performed poorly against criterion (e), “responsive primarily to a human activity, with low responsiveness to other causes of change” (ICES, 2001a); although clearly responsive to fishing, they were also strongly affected by environmentally driven recruitment variation.

The mean individual fish weight in the community was a fixed definition, but two aspects of the proportion of LFI could be altered to improve its compliance with criterion (e). First, the large-size threshold could be increased, increasing the time

required to exceed the threshold and giving fishing mortality more opportunity to reduce exceptionally large cohorts, and hence weakening the influence of environmentally driven recruitment events on the metric's performance. Second, the proportion could be calculated based on weight rather than numbers, i.e. downweighting the contribution of small fish to metric performance compared with that of large fish. The LFI was therefore redefined as the "proportion by weight of fish greater than 40 cm in length" (ICES, 2007c), i.e. $I_{LF} = B_{>40\text{cm}}/B_{\text{Total}}$, where $B_{>40\text{cm}}$ is the combined biomass in the sample of all fish larger than 40 cm, B_{Total} the total biomass of all fish in the sample, and I_{LF} the LFI. This redefined metric was much less affected by the 1999 haddock recruitment event (Figure 3). ICES (2007c) recommended that the MIWI was still required, but that it should only be used in a surveillance role, such that variation in the MIWI could aid interpretation of changes in the LFI. Contrary to expectation, correlation between the LFI and the MIWI was weak ($r^2 = 0.114$, $n = 26$, $p = 0.09$).

Setting an EcoQO for the new LFI

In 1983, the redefined LFI was ~ 0.3 (Figure 3), leading ICES (2007c) to propose this value as the EcoQO for the North Sea demersal fish community. To assess the validity of the early 1980s reference period, ICES (2007a) examined trends in spawning-stock biomass (B) and fishing mortality (F) for seven assessed North Sea demersal stocks, relative to their limit (lim) and precautionary (pa) reference points (Table 1). B_{lim} is the spawning-stock biomass at which recruitment potential is likely to be compromised, and F_{lim} is the level of fishing mortality likely to drive a stock down to B_{lim} . The precautionary reference points (B_{pa} and F_{pa}) take account of uncertainty in estimating actual levels of spawning-stock biomass and fishing mortality and provide a buffer to ensure that fishing mortality does not actually exceed F_{lim} and stock biomass is not actually below B_{lim} . ICES (2007a) derived the biomass-weighted values of average demersal fishing mortality and spawning-stock biomass for each year and constructed demersal community precautionary plots equivalent to those provided for individual species (ICES, 2006b). We here repeat this analysis using the most recent estimates of spawning-stock biomass and fishing mortality for each individual species

Table 1. Limit (lim) and precautionary (pa) reference points for spawning-stock biomass (B) and fishing mortality (F) for the seven main demersal fish stocks in the North Sea assessed using standard single-species stock assessments (ICES, 2009a).

Species	B_{lim} (t)	B_{pa} (t)	F_{lim}	F_{pa}
Cod	70 000	150 000	0.86	0.65
Haddock	100 000	140 000	1	0.7
Saithe	106 000	200 000	0.6	0.4
Whiting	225 000	315 000	0.9	0.65
Plaice	160 000	230 000	0.74	0.6
Sole	25 000	35 000	0.55	0.4
Norway pout	90 000	150 000	0.95	0.68

Emboldened values indicate reference points that are currently undefined, and italicized values are those assumed for the purposes of this analysis. Sole F_{lim} was estimated by multiplying F_{pa} by 1.374, the average difference for the five species with both values defined, giving a value of 0.55. Norway pout F varied in a similar manner to that of whiting and haddock over much of the single-species stock assessment time-series, so Norway pout was arbitrarily assigned F_{lim} and F_{pa} values of 0.95 and 0.68, respectively, the average reference point values for the other two species (ICES, 2007a).

(s ; ICES, 2009a), but instead standardize the data by expressing them relative to each species' precautionary reference points (Table 1), before calculating the community average biomass ($\bar{B}_{\text{com},y}$) and the mortality rate ($\bar{F}_{\text{com},y}$) values across all seven (S) commercial species in any 1 year (y), i.e.

$$\bar{B}_{\text{com},y} = \frac{\sum_{s=1}^S B_{s,y}/B_{s,\text{pa}}}{S} \quad \text{and} \quad \bar{F}_{\text{com},y} = \frac{\sum_{s=1}^S F_{s,y}/F_{s,\text{pa}}}{S}.$$

Full F and B time-series from 1963 to 2008 were provided for cod (*Gadus morhua*), haddock, plaice (*Pleuronectes platessa*) and sole (*Solea solea*). For whiting (*Merlangius merlangus*), saithe (*Pollachius virens*), and Norway pout (*Trisopterus esmarkii*), data were provided from 1990, 1967, and 1983, respectively, to 2008. However, the multispecies virtual population analysis (MSVPA) provided estimates for all seven species for the period 1963–2003 (ICES, 2006c), so for the three species where data for the earlier parts of the time-series were missing, linear regression of single-species stock assessment and MSVPA data over the periods for which both sets of data were available was used to fill in the gaps.

A community-averaged precautionary plot was then constructed by plotting $\bar{B}_{\text{com},y}$ against $\bar{F}_{\text{com},y}$ (Figure 4). As a consequence of the standardization, the community-averaged $\bar{B}_{\text{com},\text{pa}}$ and $\bar{F}_{\text{com},\text{pa}}$ reference points both equal one (Figure 4). To complete the plot, appropriate community-averaged B_{lim} and F_{lim} reference points needed to be determined. This was done by expressing each individual species $B_{s,\text{lim}}$ and $F_{s,\text{lim}}$ reference points relative to their $B_{s,\text{pa}}$ and $F_{s,\text{pa}}$ reference points, then calculating the average of these relative values:

$$\bar{B}_{\text{com},\text{lim}} = \frac{\sum_{s=1}^S B_{s,\text{lim}}/B_{s,\text{pa}}}{S} \quad \text{and} \quad \bar{F}_{\text{com},\text{lim}} = \frac{\sum_{s=1}^S F_{s,\text{lim}}/F_{s,\text{pa}}}{S}.$$

During the early 1980s, the demersal community average biomass exceeded the community-averaged B_{pa} reference point, but fishing pressure on the demersal fish community was too high, $\bar{F}_{\text{com},y} > \bar{F}_{\text{com},\text{pa}}$ (Figure 4), so $\bar{B}_{\text{com},y}$ was already on a

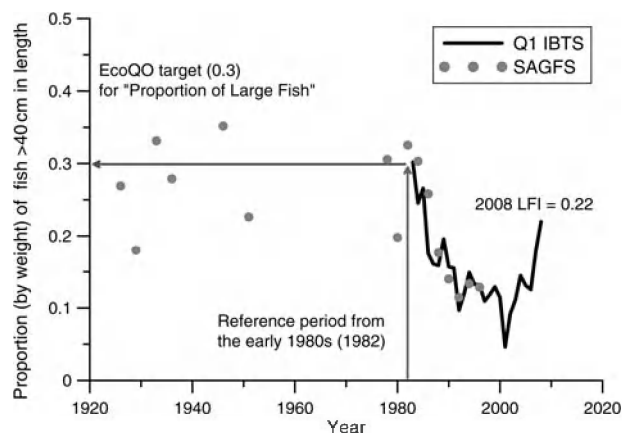


Figure 3. Variation in the redefined proportion of LFI calculated for both the Q1 IBTS and the SAGFS datasets. The current LFI value is indicated, as is the EcoQO level for the indicator of 0.3 for the North Sea demersal fish community.

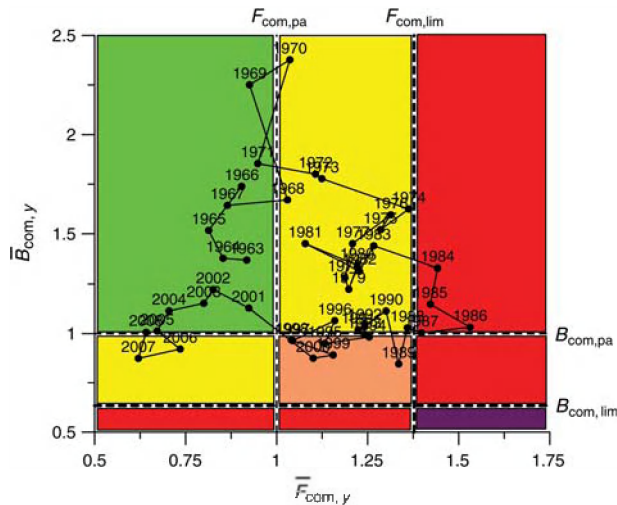


Figure 4. Demersal fish community “precautionary plot” based on community-averaged relative spawning-stock biomass ($\bar{B}_{com,y}$) and fishing mortality data ($\bar{F}_{com,y}$) for seven commercially important demersal species in the North Sea (cod, haddock, saithe, whiting, plaice, sole, and Norway pout). $B_{com,pa}$, $F_{com,pa}$, $B_{com,lim}$, and $F_{com,lim}$ are the community-averaged precautionary and limit spawning-stock biomass and fishing mortality reference points, respectively. All data shown are plotted relative to the community average precautionary (pa) reference points (see text for detail). Green zone, no precautionary points breached; yellow zone, one precautionary point breached; orange zone, both precautionary points breached; red zone, one limit point breached; purple zone, both limit points breached.

downward trajectory; explaining the rapid decline in the LFI from the 1983 start of the time-series (Figure 3).

In addition, ICES (2007c) compared the SAGFS and Q1 IBTS LFI trends over the period of survey overlap. The correlation was strong ($r^2 = 0.767$), suggesting that both LFIs were responding similarly to the changes in the demersal fish community (Figure 3). Between 1920 and 1982, there was no evidence of any trend in the SAGFS LFI, with values varying around 0.29 (Figure 3). Taken together, these two results suggest little reason to expect the Q1 IBTS LFI to increase much above 0.3, even if fishing pressure on the North Sea’s demersal fish community were to be reduced to levels prevalent near the start of the 20th century. Therefore, the EcoQO for the North Sea demersal fish assemblage is “the proportion (by weight) of fish greater than 40 cm in length should be greater than 0.3” (Heslenfeld and Enserink, 2008), with the LFI calculated using the Q1 IBTS dataset. Even under lower levels of exploitation, the LFI may on occasion fall below the EcoQO level of 0.3 owing to natural variability in the community, as seen in the SAGFS data.

Recovery of the LFI

Since 2000, the TACs of most North Sea stocks have become increasingly restrictive to facilitate stock recoveries (ICES, 2008b). Measures such as vessel decommissioning and restrictions on days at sea have also reduced fishing activity levels by up to 40% (Greenstreet *et al.*, 2009), which is likely to have reduced fishing mortality across the entire demersal fish community, including that on non-assessed species (Piet *et al.*, 2009). It is therefore

reassuring to see the considerable recovery in the LFI since 2001 (Figure 3).

This recovery was primarily initiated by a cohort of saithe growing above the threshold of 40 cm and subsequently maintained by a cod cohort also exceeding the threshold. Trends in the biomass of 11 of the most abundant species in the community suggest little change in the extent to which large fish were dominated by any particular species (Figure 5). Species richness of the large fish component of the community increased slightly over the whole period, whereas species evenness increased during the LFI decline phase and decreased during the subsequent recovery phase. In both cases, however, 2008 values still exceeded those at the start of the time-series (Figure 6). There was therefore no evidence of a recent reduction in diversity of the large fish component of the North Sea demersal fish community; a situation that might have prevailed if a single species was principally responsible for the recovery in the LFI, rather than a broad-scale recovery across the whole community.

Being a ratio metric, the LFI is influenced by changes in the biomass of fish both >40 and ≤ 40 cm long. We were interested to determine whether the indicator was more sensitive to variation in the biomass of one or other of these size categories. To do this, we used a Taylor series expansion to derive the approximate variance of I_{LF} :

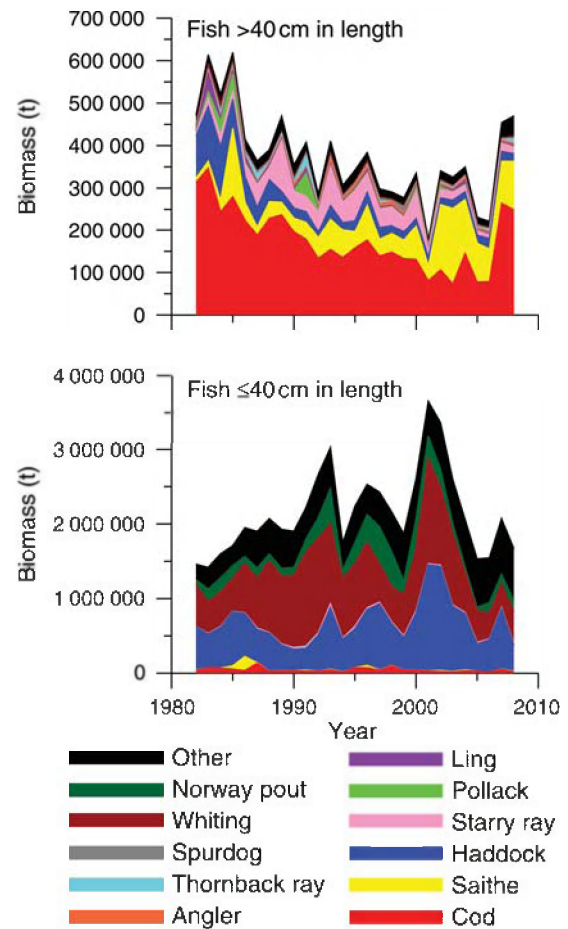


Figure 5. Trends in the biomass of key species making up the >40 and ≤ 40 cm length components of the North Sea fish community.

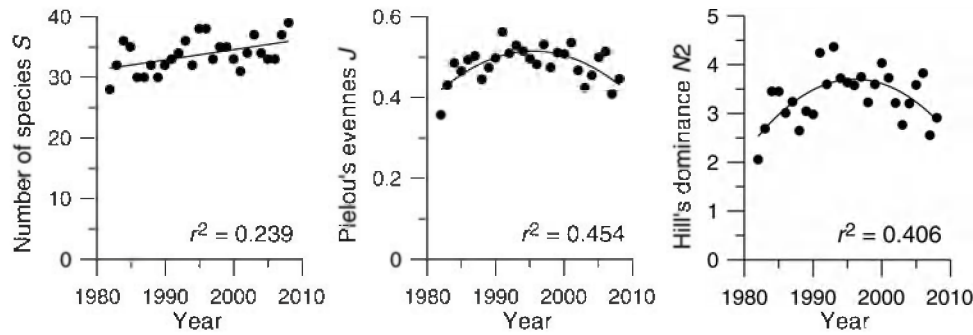


Figure 6. Trends in the number of species recorded (S), Pielou's species evenness (J), and Hill's dominance index (N_2) calculated on the biomass data for individual species in the >40 cm length component of the North Sea demersal fish community. Squared correlation coefficients (r^2) indicate the goodness of fit to linear (for S) and second degree polynomial (for J and N_2) models of the data.

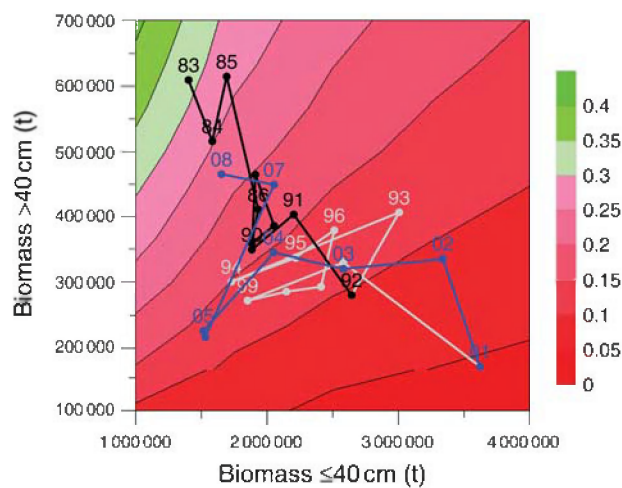


Figure 7. Contoured surface showing variation in the proportion of LFI for varying biomass of fish >40 and ≤ 40 cm long. The EcoQO value of 0.3 is indicated as the contour between green and red shades. The actual trajectory of the indicator is shown during three phases, the decline phase (black, 1983–1992), the unhealthy phase (grey, 1992–2001), and the recovery phase (blue, 2001–2008).

$$\text{var}(I_{LF}) \approx \bar{I}_{LF}^2 (1 - \bar{I}_{LF})^2 ((CV(B_{>40\text{cm}}))^2 + (CV(B_{\leq 40\text{cm}}))^2 - 2CV(B_{>40\text{cm}})CV(B_{\leq 40\text{cm}})\text{corr}(B_{>40\text{cm}}, B_{\leq 40\text{cm}}))$$

where \bar{I}_{LF} is the mean across years of the LFI, $CV(B_{>40\text{cm}})$ and $CV(B_{\leq 40\text{cm}})$ the interannual coefficients of variation in the biomass of fish >40 and ≤ 40 cm, respectively, and $\text{corr}(B_{>40\text{cm}}, B_{\leq 40\text{cm}})$ the correlation between $B_{>40\text{cm}}$ and $B_{\leq 40\text{cm}}$. As $CV(B_{>40\text{cm}})$ and $CV(B_{\leq 40\text{cm}})$ were estimated to be 29 and 26%, respectively, variation in the LFI has been driven by variation in the biomasses of fish >40 and ≤ 40 cm long in roughly equal measure. One management implication of this is that sudden fluctuations in the indicator value may be caused by environmentally driven influxes of small fish, and not necessarily by a fishing impact on the larger fish in the community. Such signals should, however, be picked up by the MIWI.

A contoured surface was generated showing variation in the LFI across all biomass combinations of fish >40 and ≤ 40 cm long. The LFI trajectory plotted on the top of this revealed three clear

phases (Figure 7). During the decline phase from 1983 to 1992, indicator values were driven down by both decreasing biomass of large fish and increasing biomass of small fish. Between 1992 and 2000, large fish biomass altered little, and variation in the indicator was influenced primarily by frequent changes in the biomass of small fish. A further decline in the biomass of large fish combined with a substantial increase in small fish biomass drove the indicator down to its lowest point in 2001. The recovery phase from 2001 to current was characterized by a declining biomass of small fish combined since 2006 with an increase in the abundance of large fish. Variation in the biomass of small fish was heavily influenced by changes in the biomass of some species, e.g. whiting, that rarely grow to a length where they influence the variation in the biomass of large fish (Figure 5).

The relationship between pressure and state

The current definition of the LFI was chosen to increase its sensitivity to fishing pressure, so it is pertinent to explore its relationship with $\bar{F}_{\text{com},y}$ (Figure 8a). Large piscivorous fish must be removed by fisheries before predation-pressure release causes the abundance of small fish to increase. Figure 7 suggests that the direct and indirect effects of fishing were indeed non-synchronous, i.e. that periods of declining biomass of large fish were followed by periods of increasing biomass of small fish, and vice versa. This implies a lagged relationship, $I_{LF,y} = f(\bar{F}_{\text{com},y-L})$, and the lags (L) involved may last a decade or longer (Daan *et al.*, 2005). A 1-year time-lag was considered the baseline because the LFI, derived from the Q1 IBTS carried out at the start of any year, would have been influenced by fishing pressure in earlier years. Significant negative correlations between the two time-series were found at lags of 12–20 years (Figure 8b).

To determine whether shorter-term changes in fishing pressure had any immediate effect on the LFI, both time-series were detrended by fitting loess smoothers (degree 2 with a 20-year neighbourhood; Figure 8a), and correlations between the residuals were explored. We were only interested in negative correlations with short time-lags, reflecting the effect of, for example, a rapid reduction in fishing mortality on the increased survival of large fish and increased “recruitment” of fish <40 cm long into the larger fish category. A 1-year lag was again the baseline. There was a significant, though weak, negative correlation between the residuals at a lag of 2 years. All other correlations were not significant (Figure 8c).

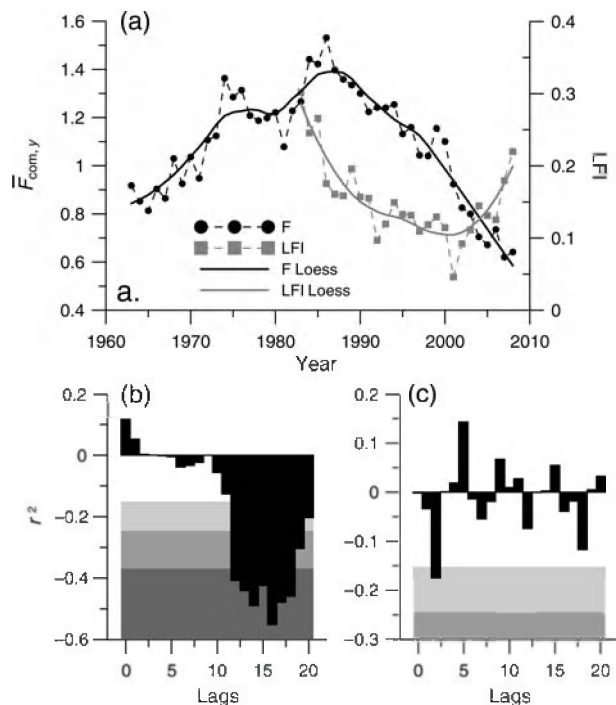


Figure 8. (a) Community-averaged fishing mortality ($\bar{F}_{com,y}$) and the LFI with fitted loess smoothers. (b) Cross-correlations, expressed as r^2 , between LFI and $\bar{F}_{com,y}$ at various time-lags (shading indicates significance levels: light grey, $0.05 > p > 0.01$; intermediate shade, $0.01 > p > 0.001$; dark grey, $p < 0.001$). (c) Cross-correlations, expressed as r^2 , between the detrended LFI and $\bar{F}_{com,y}$ time-series at various time-lags (shading indicates significance levels: light grey, $0.05 > p > 0.01$; intermediate shade, $p < 0.01$).

Looking into the future: advice to management

Providing sound scientific advice for management requires sound understanding of the relationship between fishing pressure and the LFI, but this relationship is clearly more complex than initially thought. Ideally, the processes involved should be represented in theoretical mechanistic models, which can be used to provide ecosystem advice in much the same way that stock assessment models currently underpin fisheries management advice. An important question for such models would be that if fisheries management were to achieve individual stock objectives (stocks exploited with $F < F_{pa}$ and spawning-stock biomass maintained at $B > B_{pa}$), would this also be sufficient to achieve the LFI EcoQO? If the answer is yes, then the current single-species fisheries management plans alone are adequate to meet current broader ecosystem goals for the whole North Sea demersal fish community. Modelling simulations could then address how long it might take for the EcoQO to be achieved. If the answer is no, then what further reductions in fishing mortality below F_{pa} would be required to attain the EcoQO for the LFI, and over what time-scale might different fishing scenarios achieve the target (ICES, 2009b)? To initiate this process, ICES (2008a, 2009b) posed a series of F -reduction scenarios that size-resolved, multispecies models needed to address. Theoretical models addressing these require several attributes. Fishing mortality must be represented realistically to model the consequences of different fishing-pressure scenarios. They must be dynamic to predict the time-scale of

change in the LFI, they need to represent the key ecosystem processes driving community size structure, and they must be ecosystem-specific.

Discussion

Analysis of the initial LFI and MIWI defined by ICES (2006a) suggested that they could be strongly influenced by recruitment events, which are primarily environmentally driven (Dippner, 1997; Kempf *et al.*, 2009). Environmental and density-dependent effects on growth and recruitment can influence metrics of fish size regardless of the level of fishing activity (Badalamenti *et al.*, 2002; Lekve *et al.*, 2002; Wilderbuer *et al.*, 2002), and a recent review by Shin *et al.* (2005) concludes that size-based metrics may indeed be more sensitive to other drivers of change in fish communities than was initially thought. Size-based metrics may also be poor indicators of overexploitation in fish communities dominated by species characterized by small ultimate body size, high growth rate, and early age and small size at maturation (Stobberup *et al.*, 2005). High fishing mortality has caused the North Sea demersal fish community to become increasingly dominated by species characterized by such r -type life-history traits (Jennings *et al.*, 1999; Greenstreet and Rogers, 2000), perhaps implying that the more the fish community is affected and altered by overexploitation, the less effective size-based metrics become at monitoring the impacts of fishing.

Marine ecosystems are directly and indirectly impacted by fishing (Jennings and Kaiser, 1998). Direct effects are those changes caused by fishing, i.e. the mortality of large piscivorous fish caught in the trawl. Indirect effects are the changes that follow as a consequence of these direct effects, e.g. the proliferation of small fish resulting from reduction in their natural rates of predation mortality (Pope, 1991; Daan *et al.*, 2005). Following exploitation, increases in the biomass of small fish generally exceed reductions in the biomass of large fish (Greenstreet *et al.*, 1999; Blanchard *et al.*, 2005; Daan *et al.*, 2005). If large fish have diets consisting of small fish (Greenstreet, 1996) and assuming trophic level transfer efficiency of 10% (Baumann, 1995; Kerr and Dickie, 2001; Jennings *et al.*, 2002) and that small fish specific growth productivity is double that of large fish (Banse and Mosher, 1980), then simple trophodynamic theory suggests that the increase in the biomass of small fish following release from predation may be $5\times$ the decrease in the biomass of large predatory fish. Figure 7 confirms that this was approximately the case for the demersal fish community in the North Sea; the biomass of large fish (>40 cm) varied by $\sim 440\,000$ t, but that of small fish (≤ 40 cm) by $2\,300\,000$ t.

The choice of size range over which size-based metrics are determined affects their capacity to detect changes in trend (Jennings and Dulvy, 2005). Redefining large fish as fish >40 cm and calculating the LFI by weight rather than number reduced the indicator's responsiveness to variation in the abundance of small fish, in effect increasing its sensitivity to the direct effects of fishing and reducing its sensitivity to indirect effects. Even so, analysis of the sensitivity of the LFI to changes in the abundance of both small and large fish suggests that the LFI remained about equally responsive to both direct and indirect effects of fishing on the demersal fish community. However, the LFI signal-to-noise ratio was markedly improved, increasing its power to detect trend direction changes over time-spans sufficiently short (e.g. 5–10 years) to satisfy management needs (Nicholson and Jennings, 2004; Maxwell and Jennings, 2005).

The values of LFI and community-averaged fishing mortality were significantly correlated only at time-lags >12 years. Considering such lags as implausible implies that the LFI is not responsive to fishing pressure on the fish community, but the evidence to the contrary is simply too overwhelming to be ignored. Theory linking the change in individual stock age (and hence size) structure to variation in mortality is well established (Beverton and Holt, 1957). Theoretical models of fish communities have universally demonstrated an inverse relationship between fishing mortality and average fish size (Hall et al., 2006; Pope et al., 2006; Blanchard et al., 2009), and many empirical studies have demonstrated fishing-induced shifts towards smaller size in fish communities (Daan et al., 2005; Shin et al., 2005; Greenstreet and Rogers, 2006). Increased mean size of fish inside marine reserves is one of the most frequently observed responses following the cessation of fishing (Molloy et al., 2009). Finally, a more immediate inverse response was observed when both time-series were detrended. That this significant relationship occurred with a lag of 2 years suggests perhaps that the growth of smaller fish to exceed 40 cm was responsible for the relationship, rather than increased survival among fish >40 cm long; the extra year (beyond the baseline lag of 1 year) being required to allow time for this growth.

Instead of a lagged linear relationship, a simple step function could link the LFI and $F_{com,y}$ trends. Fishing pressure exceeding $F_{com,lim}$ between 1983 and 1987 could have caused the LFI to decline sharply, and fishing pressure below $F_{com,lim}$ but above $F_{com,pa}$ (1988–2000) resulted in a gradual decline in LFI; only when fishing pressure on the fish community was reduced to below $F_{com,pa}$ from 2001 did the LFI start to increase.

Alternatively, the relationship between the pressure and state indicators is generally linear and the >12 year lag in the LFI response is real (Daan et al., 2005). Although the removal of large fish from the community is immediate, depending on the population dynamics of the species involved, 5 or more years may be required following an increase in the mortality rate before a new equilibrium population age structure is reached (Beverton and Holt, 1957). The increase in the abundance of small fish following a release from predation pressure may continue for several years after the abundance of large fish has declined (Daan et al., 2005). Increased fishing mortality also alters the life-history composition of the fish community, increasing representation of small species (Jennings et al., 1999; Greenstreet and Rogers, 2000), and this may allow the abundance of small fish to increase long after the initial increase associated with predation-pressure release.

Fishing pressure on the North Sea demersal fish community has declined steadily since 1986. The LFI time-series, which only started in 1983, therefore generally reflects changes in the demersal fish community associated with reducing fishing mortality. Although fishing might immediately remove large piscivorous fish, there is no equivalent immediate replacement of these fish when fishing mortality is reduced. Few species have ultimate body lengths >40 cm (Jennings et al., 1999). Most fish >40 cm will therefore already be close to their ultimate body length and hence grow relatively slowly. Reducing fishing mortality among such fish therefore serves more to maintain the current LFI values than to stimulate its recovery. It is the increased survival of small individuals of large-bodied species, and their growth to exceed the 40 cm threshold, that provides most impetus to an upward LFI trend. The time required for this depends on the

growth rates and population dynamics of the species involved. However, across the whole demersal fish community, once initiated by a reduction in fishing mortality, this process may well continue for 10 or more years before a new equilibrium is reached.

Moreover, when top-down control on a trophic level is reduced, bottom-up processes take over; competition within the trophic level increases as abundance rises (Carpenter et al., 1987; Shurin and Seabloom, 2005). Following release from predation, the increasing abundance of small fish may raise the level of competition among fish in smaller size classes, possibly impairing the growth rates of juveniles of larger-bodied species growing through these size classes (Lekve et al., 2002), and further delaying the recovery of the LFI following reduced fishing mortality. Such an explanation has been posited for the slow recovery of the demersal fish community on Georges Bank following remedial management to undo decades of overexploitation (Fogarty and Murawski, 1998). Time-lags in recovery situations may be longer than those associated with increasing disturbance. In a recent meta-analysis, the greatest increase in the abundance of large fish species was observed in marine reserves that had been in place for >15 years (Molloy et al., 2009).

Advising managers how best to achieve the North Sea demersal fish community EcoQO requires the relationship between fishing pressure and the LFI to be properly understood. To underscore this point, for illustrative purposes let us assume that the relationship is linear and time-lagged; the current state of the fish community is a consequence of the fishing mortality regime prevailing 12 or more years ago. The community-averaged fishing mortality time-series can therefore be used to look at least 12 years into the future. Seven linear regression models relating the LFI to $F_{com,y}$ lagged by 12–18 years, respectively (the lags giving significant correlations at $p < 0.01$), were used to predict the LFI forward from 1983 to 2020. The averaged prediction, well fitted to the observed data, suggested that the LFI should continue to increase through to 2020 (Figure 9). Rather than being a response to steps taken to reduce fishing mortality since 2000, improvement in the LFI since 2001 may be a consequence of the steady decline in fishing mortality since the mid-1980s. Further, steepening of the LFI trajectory from 2013 on may reflect the broader ecosystem benefits of increasingly stringent fisheries management policy since 2000 (Greenstreet et al., 2009) that have still to be reaped. From an advisory perspective, Figure 9 suggests that the management action already taken to conserve the principal commercial species may be sufficient to achieve the EcoQO for the North Sea's demersal fish community by 2020. However, without better understanding of the actual processes that link changes in the LFI to changes in fishing pressure, such advice would be risky.

We remain convinced that the LFI is sensitive to changes in fishing pressure and useful within a management context. However, we are forced to conclude that the relationship between these pressure and state indicators is more complicated, and our understanding of it less advanced, than was believed to be the case at the start of the process when application of the ICES criteria first selected size-based metrics to support the fish community EcoQO. We have posited two possible relationships to explain our results. At present both explanations are hypothetical; neither has been nor can be tested empirically. The solution to this dilemma lies in the development of appropriate

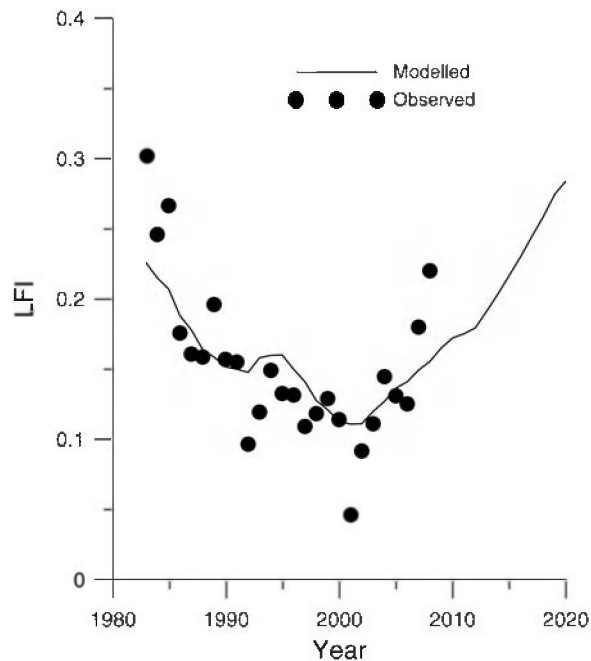


Figure 9. The average predicted LFI from seven linear models that regressed the observed LFI on the community-averaged fishing mortality lagged by 12–18 years, respectively. Observed Q1 IBTS LFI values are shown for comparison.

size-structured and species-resolved theoretical fish community models. The experiments needed to test our hypotheses can then be made as simulations.

Currently, candidate models fall into two general categories: size-resolved life-history models (e.g. Shin and Cury, 2004; Hall *et al.*, 2006; Pope *et al.*, 2006), and community size-spectra models (e.g. Benoît and Rochet, 2004; Maury *et al.*, 2007; Blanchard *et al.*, 2009). Neither fulfils all the requirements, but both have key attributes and indicate how suitable models might be developed. The former explicitly represents size-structured populations, parametrizes ecosystem processes using a combination of size-based functions and empirically tractable life-history traits, and includes length and species-based fishing mortality. However, such models are steady state, fish growth is prescribed, and stock–recruit relationships are constant, compromising their ability to predict the dynamic consequences of changes in fishing pressure on fish community structure. In the second group, species are not explicitly represented; instead fish in the community are represented simply as numbers of all individuals belonging to given size classes. All processes (food intake, growth, mortality) are parametrized based on size alone. Size-based predation and an underlying bioenergetics model governing the flow of energy through the system provide the major driving processes and are generally dynamic. These models can incorporate a size-based fishing mortality, but not fishing mortality affecting different size classes of different species. Current work focuses on combining the two modelling approaches but this presents both technical and conceptual difficulties. However, in a preliminary analysis, one such derivative model (Andersen and Beyer, 2006; Andersen *et al.*, 2009) was used to explore the response of the LFI to reduced fishing mortality. Following a 50% reduction in fishing mortality (the maximum tried in the

experiment), the LFI equilibrated at a new stable value after 10 years, and lag-time increased the more fishing mortality was reduced (ICES, 2008a). Since 1986, community-average fishing mortality has declined by 58%. These early results again suggest that recovery of the LFI may be a lengthy process, but if so, it is one that has been ongoing since 1987.

Other results have emerged from our analyses that are difficult to explain without insight from these models. For instance, although recovery in the LFI has been considerable since 2001 (Figure 3), when fishing mortality averaged across the seven assessed demersal species finally dropped below F_{pa} , no similar recovery in community-averaged spawning-stock biomass ($B_{com,y}$) was observed. This metric has simply fluctuated around the community-averaged precautionary reference point (Figure 4). Although the size composition of the demersal fish community is recovering, spawning-stock biomasses of the main commercial stocks in general are not. Perhaps reducing fishing mortality to more sustainable levels has produced a situation where the biomass removed by fisheries is compensated for by growth of the survivors, so the spawning component of each stock consists of an ever-dwindling number of increasingly large fish, with little or no change in spawning-stock biomass. In such circumstances, recovery of spawning-stock biomass might be reliant on the recruitment of strong juvenile cohorts, a situation that has been rare in recent years (ICES, 2009a). Resolving such issues is clearly an important aspect of a successful EAMM, in which improving the current situation for fisheries is a major concern.

The ICES criteria for a good state indicator identify metrics that best reflect the effects of human activities on ecosystem components, but other criteria also exist (Rice and Rochet, 2005; Piet *et al.*, 2008). Rice and Rochet (2005) stress the need to identify user requirements before developing a list of candidate indicators, then to apply criteria to select the best. Most treaties and conventions driving development of an EAMM, including the Marine Strategy Framework Directive, focus on ecosystem health and biodiversity. Restoring fish size composition should generally benefit the health of the North Sea demersal fish community, but it remains to be seen whether achieving the EcoQO will simultaneously reverse the long-term declines in species diversity attributed to fishing (Greenstreet *et al.*, 1999; Greenstreet and Rogers, 2006). Poor understanding of the relationship between fishing disturbance and species diversity, in part, led ICES (2001b) to disregard species diversity indicators (Greenstreet, 2008). Insight gained through the development of these theoretical size- and species-resolved models could address this shortcoming, perhaps allowing a process similar to the one described here to set realistic management goals for species diversity state indicators.

Rice (2009) describes a conceptual three-stage model underlying the development and implementation of a precautionary approach to North Sea fisheries management. Development of the North Sea demersal fish community EcoQO has followed a similar process, supporting Rice's (2009) contention that this three-stage model is broadly applicable and, in generalized form, suitable as a basis for establishing management frameworks that address wider marine ecosystem conservation issues. This is important because the major stakeholders, fishers and fisheries managers, have both accepted the precautionary approach to fisheries management and are more likely therefore to accept a process for implementing an EAMM that is conceptually similar. For example, the EcoQO states that the proportion (by weight) of

fish >40 cm long should be >0.3 (Heslenfeld and Enserink, 2008). When the LFI is below 0.3, management's short-term objective should be to restore it to >0.3. However, once past this threshold, it should not be management's intent to exploit demersal fish so that the LFI is held at or close to a value of 0.3. In this sense, the EcoQO has a similar role to B_{pa} , which is not a management target, but is rather considered the permitted lower bound to spawning-stock biomass. However, if $B < B_{pa}$, then *de facto* B_{pa} becomes a short-term target to stimulate recovery. However, if $B > B_{pa}$, it should never be management's goal to exploit the stock down to this precautionary limit (Piet and Rice, 2004). Once the size composition of the North Sea's demersal fish community has been restored, size-resolved multispecies theoretical models can be used to identify sustainable levels of exploitation that maintain the LFI above 0.3.

The three-stage precautionary approach to fisheries management differs from the fish community EcoQO in one important aspect. The latter has just the single reference value, whereas the former uses both a precautionary reference point (B_{pa}) and a limit reference point (B_{lim}). Depending on where B lies with respect to these two reference points dictates the management response. Rice (2009) posits a variety of different options for dealing with situations where biomass lies between B_{pa} and B_{lim} , all of which involve increasingly stringent control of the exploitation rate as biomass declines further below B_{pa} and approaches B_{lim} . At present, we have no equivalent management procedures for use when the LFI drops below 0.3. Perhaps in the future, theoretical size-resolved multispecies models might be used to determine an LFI equivalent to B_{lim} for the demersal fish community, enabling a similar set of management procedures to be implemented. Alternatively, there may be no need for a management procedure as complex as this. It may be sufficient to know that the LFI is below the acceptable limit, and the theoretical models then simply used to identify the remedial action necessary to resolve the issue within a specified time-scale.

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