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Misspent youth: does catching immature fish affect fisheries sustainability?

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The "spawn-at-least-once" principle suggests that sustainability is secured if fish become vulnerable to commercial gears only after they have spawned. However, some studies suggest that protecting immature fish is not essential to sustainability because extrinsic factors determine both recruitment and stock status. A meta-analysis was conducted to quantify the independent effects of exploitation pattern and exploitation rate on current stock status. The analysis used empirical data for 38 fish stocks of 13 species in the NE Atlantic. Two metrics of exploitation pattern were used and their sensitivity was compared. As expected, exploitation rate had a significant negative effect on current stock status. Exploitation patterns associated with high proportional fishing mortality of immature fish also had a significant negative effect on current stock status, providing empirical support for the "spawn-at-least-once" principle. When the fishing mortality of immature fish exceeds half that of mature fish, stock status falls below precautionary limits. Our results suggest that a sensitive metric of exploitation pattern could provide useful information about an aspect of exploitation that is currently overlooked by fisheries management regimes that focus primarily on exploitation rate.

Keywords: exploitation pattern, exploitation rate, fisheries management, ICES, meta-analysis, spawn-at-least-once, sustainability.

Introduction

Fisheries science distinguishes between two distinct aspects of exploitation of commercial species: exploitation rate (fishing intensity) and exploitation pattern (fishing selection). Exploitation rate is defined as the proportion of the population or biomass of a fish stock that is removed each year (FAO, 2010) and is usually quantified by fishing mortality (*F*; ICES, 2008). The precautionary approach to fisheries in Europe and elsewhere has led to the adoption of reference points for *F* and spawning-stock biomass (SSB; Wakeford *et al.*, 2007; ICES, 2008), and the main focus of fisheries management has been regulating *F* (Froese *et al.*, 2008; ICES, 2008). Exploitation pattern is defined as the distribution of fishing mortality over the age or length composition of the fish population (FAO, 2010), and it depends on the selectivity of the gears used in a fishery and on the extent to which particular age/size classes are targeted. Both factors are associated with design specifications of the fishing gear, fish behaviour, the temporal and spatial distribution of fish, and skipper expertise (Wileman *et al.*, 1996). Exploitation pattern also represents the proportion of immature fish caught, because size in fish reflects their maturity stage. This relationship between size and maturity stage can be impacted by directional shifts in the maturation rate, for instance if, over time, fish tend to mature smaller (de Roos *et al.*, 2006).

The need to protect immature fish to enhance fishery yields is an intuitive concept pre-dating fisheries science itself (Halliday and Pinhorn, 2002; Kennelly and Broadhurst, 2002). This traditional approach assumes that exploitation pattern plays an important role in fisheries management and that a key element of good management is the use of fishing gears that retain large fish in the catch while allowing juveniles to escape (Armstrong *et al.*, 1990). The "spawn-at-least-once" principle presumes a significant correlation between the size of the spawning stock and subsequent recruitment (Myers and Barrowman, 1996; Myers and Mertz, 1998), so allowing fish to survive long enough to spawn is assumed to prevent recruitment-overfishing (Myers and Mertz, 1998; Halliday and Pinhorn, 2002). There are many studies focusing on specific stocks of pelagic and demersal species throughout the world that suggest that implementation of the "spawn-at-least-once" principle through the adoption of suitable exploitation patterns promotes sustainability and profitability of fisheries (e.g. Mori *et al.*, 2001; Enberg, 2005; Klanjšček and Legović, 2007; Ishida *et al.*, 2009). Additionally, the protection of immature fish, which are typically small, could indirectly reduce the potential for growth-overfishing. The yield-per-recruit theory (Beverton and Holt, 1957) indicates that for a given stock there is an optimal size at first capture (L_{opt}) for a given level of *F* that can maximize fisheries yield, this optimal size being usually larger than the size at which 50% of the individuals have reached maturity (L_{50}; Froese *et al.*, 2008).

The validity of the "spawn-at-least-once" principle is, however, open to challenge. The very existence of a connection between SSB and subsequent recruitment has been disputed (Gilbert, 1997), and other factors such as environmental variability (Jarre-Teichmann *et al.*, 2000), physiological condition (Marshall *et al.*, 1999), and the age range of the spawning population (Marteinsdottir and Thorarinsson, 1998; Cardinale and Arrhenius, 2000) have been shown to play an important role in determining levels of recruitment. In addition, protecting older and more experienced females (megaspawners) that tend to produce more, larger, and qualitatively superior eggs (Marteinsdottir and Steinarsson, 1998; Cardinale and Arrhenius, 2000; Trippel and Neil, 2004) has been suggested as a more effective alternative management strategy for promoting sustainability and profitability (Caddy and Seijo, 2002; Palakovich Carr and Kaufman, 2009). Moreover, doubts have been raised about the likelihood of obtaining the predicted increased yields by increasing the size at first capture, because of uncertainty in the calculation of the fish population parameters implemented in yield-per-recruit analyses (Haddon, 2001; Halliday and Pinhorn, 2002). Finally, fishing populations more evenly instead of selectively catching large fish has also been proposed as a means of reducing the impact on stocks and ecosystems (Zhou *et al.*, 2010). All these perspectives support the perception that exploitation patterns leading to the capture of fewer immature fish have no effect on stock status and that it is exploitation rate and/or protection of other demographic components of the fish population that matter.

These two competing hypotheses, i.e. whether or not immature fish should be protected, essentially describe two different approaches to fisheries management. Froese (2004) synthesized elements of both approaches to suggest that fisheries sustainability can be achieved by fishing at L_{opt} and that the "spawn-at-least-once" principle should be implemented in conjunction with the protection of megaspawners. Therefore, he proposed that both the percentage of mature fish and the percentage of megaspawners in the catch should be used as simple indicators of sustainability. That approach requires fishing mortality to vary with age/size class and hence inherently implies that exploitation pattern does indeed play a role in sustainability.

To date, therefore, the debate about the potential benefits from protecting immature fish remains unresolved. To test empirical support for the "spawn-at-least-once" principle, a metric of exploitation pattern that characterizes the relative exploitation of immature to mature fish is needed. Only a few such metrics have been suggested in the literature. Froese (2004) proposed the percentage of mature fish in the catch as a straightforward and simple proxy for exploitation pattern that could be also used as an indicator of fisheries sustainability. Petrakis and Stergiou (1997) used the divergence between the length where 50% of the fish are retained (L_{r50}) and L_{50} as a proxy for exploitation pattern and to assess the stock sustainability associated with different fishing gears. This divergence can be calculated easily for a single type of gear applied in a specific fishery, but its calculation would be more complex in multinational, multigear fisheries, as for NE Atlantic stocks. In this study, exploitation pattern is quantified as the proportion of fishing mortality of immature fish to that of mature fish. This metric has the advantage of taking into account the whole fish population and being easy to calculate for many assessed stocks.

The effect of historical values of exploitation pattern and exploitation rate on current stock status through a meta-analysis of 38 fish stocks in the NE Atlantic is investigated. The data used are widely available empirical data from ICES stock assessment reports. The meta-analysis approach was taken because general trends are more easily observed when all data are combined (Myers, 2001). Meta-analyses have already demonstrated the overall negative effect of exploitation rate on stock status (Sparholt *et al.*, 2007; Worm *et al.*, 2009). However, the overall effect of historical values of exploitation pattern on current stock status has not yet been assessed using empirical data for a wide range of species and stocks. As outlined above, an *F*-based metric of exploitation pattern is used, but the adequacy of the simpler catch-based proxy proposed by Froese (2004) is also explored. The ability of these metrics to capture the effect of exploitation pattern on current stock status and their potential to act as indicators of fisheries sustainability are also discussed.

Material and methods

Data from 38 stock assessment reports (13 fish species) conducted by nine ICES Working Groups were used in the analysis (Table 1). For each stock, the information available was year-specific values of catch-at-age ($C_{a,y}$), SSB (B_y), fishing mortality-at-age ($F_{a,y}$), abundance-at-age ($N_{a,y}$), maturity-at-age ($M_{a,y}$), and the stock-specific precautionary spawning biomass limits (B_{pa}) set by ICES.

As the available time-series were not of equal length for all stocks and different stocks exhibited different maturity rates, the interval used for the analysis was standardized according to the approximate number of generations represented rather than using a fixed, arbitrary number of years. For each stock, a time interval $T = 4 \times A_{50}$ years rounded to the nearest integer was used to calculate the average levels of exploitation rates and patterns, where A_{50} is the age at which 50% of the fish have matured (Table 1). The value of A_{50} was calculated for each stock by fitting a logistic model to maturity-at-age data provided by ICES. For the stocks where different maturity-at-age data were given for each year, the average values of the most recent 3 years were used. The time interval *T* was regarded as long enough to capture the effect of recent trends in exploitation pattern and exploitation rate on current stock status. The effect of using a different time interval was also investigated by repeating the analysis using a fixed 20-year interval for all stocks (except for the Rockall haddock stock, because just 18 years were available).

Model variables

The variables used in the models included current stock status as a response variable, and exploitation rate, exploitation pattern, and assessment effect (*AE*) as explanatory variables. Each of these is described in detail below.

The status of each stock in year *y* is defined by ICES as the value of B_y in relation to the precautionary level, B_{pa}, set by the Working Group assessing the stock. In this analysis, the average value of B_y/B_{pa} over the final 3 years of each assessment was used to describe current stock status (*CSS*):

$$CSS = \frac{1}{3}\sum_{y=fy-2}^{fy}\frac{B_y}{B_{pa}}, \tag{1}$$

where *fy* is the final year of the assessment (2009 usually). In this context, $CSS > 1$ indicates a stock within its precautionary biomass limit, and $CSS < 1$ indicates a stock below that level.

In ICES assessment reports, the exploitation rate in year *y* is expressed by $\bar{F}_y$ (year^{-1}), i.e. the average fishing mortality of the age classes primarily targeted by the fisheries. The age classes used to estimate $\bar{F}$ vary across stocks (Table 1). In this analysis, mean exploitation rate (*ER*) was defined accordingly as the average value of $\bar{F}$ in the last T years of each stock assessment:

$$ER = \frac{1}{T}\sum_{y=fy-T+1}^{fy} \bar{F}_y. \quad (2)$$

To express exploitation pattern, two alternative metrics were used. The more complicated metric (EP_1) was an F-based quantity directly quantifying exploitation pattern as the proportion of fishing mortality of immature (F_{imm}) to that of mature fish (F_{mat}). In year y, $F_{imm,y}$ and $F_{mat,y}$ were calculated for each stock as

$$F_{imm,y} = \frac{\sum_{a=1}^{n} N_{a,y} I_{a,y} F_{a,y}}{\sum_{a=1}^{n} N_{a,y} I_{a,y}} \quad (3)$$

$$F_{mat,y} = \frac{\sum_{a=1}^{n} N_{a,y} M_{a,y} F_{a,y}}{\sum_{a=1}^{n} N_{a,y} M_{a,y}}. \quad (4)$$

where $a = 1, 2, 3, \ldots n$ are the age classes, and $I_{a,y} = 1 - M_{a,y}$ is the proportion of immature fish. EP_1 was then calculated for every stock using Equations (3) and (4):

$$EP_1 = \frac{1}{T}\sum_{y=fy-T+1}^{fy} \frac{F_{imm,y}}{F_{mat,y}}. \quad (5)$$

Fish belonging to age class 0 were excluded from the estimations of $F_{imm,y}$ and $F_{mat,y}$ because relevant data were not available for most stocks and fishing mortality in that age group was usually negligible (Table 1). If there were missing age classes in a stock's dataset (Table 1), they were reconstructed using the formula

$$N_{a,y} = N_{a+1,y+1} e^{M'_{a,y}} \quad (6)$$

where $M'_{a,y}$ is the natural mortality at age a in year y (Haddon, 2001). This is essentially a back-calculation, so missing abundance values ($N_{a,fy}$) in the last year of the time-series (terminal N values) were calculated as the average of the numbers-at-age of the three preceding years (Haddon, 2001).

The second metric used to approximate exploitation pattern (EP_2) was the simpler catch-based proxy proposed by Froese (2004), and it was expressed as the proportion of immature fish in the total catch:

$$EP_2 = \frac{1}{T}\sum_{y=fy-T+1}^{fy} \left(\frac{\sum_{a=1}^{n} C_{a,y} I_{a,y}}{\sum_{a=1}^{n} C_{a,y}}\right). \quad (7)$$

This metric is easy to calculate, but it takes into consideration only a subgroup of the total fish population, i.e. those caught. Its potential to act as an indicator of sustainability, as Froese (2004) proposed, has not been fully explored. When the analysis was conducted based on a 20-year time interval, T was replaced by 20 in Equations (2), (5), and (7) to obtain the mean values of the variables over a 20-year period.

The categorical variable AE was included to express the potential degree of underestimation of immature fish exploitation (F_{imm}) attributable to unaccounted discarding in every stock. Discard estimates were explicitly included in the respective ICES assessments for just 10 of the 38 stocks, whereas for 16 other stocks, discards were not included but reported to be negligible, so not affecting the estimates of the assessments (Table 1). Those 26 stock assessments were classified at level A (low bias owing to the effect of discarding on the assessment estimates). For the other 12 stocks, discards were not included in the assessments but were reported to be potentially high, causing underestimation of the levels of fishing mortality, especially at younger ages. Those 12 stock assessments were classified at level B (possible high bias owing to the unaccounted effect of discarding).

Statistical analysis

The relationship between EP_1 and EP_2 was investigated by plotting one against the other and examining their correlation and the outliers. The reasons that five stocks appeared to be outliers in this relationship were investigated in relation to the different characteristics of the two metrics.

The dependence of CSS on the explanatory variables ER, EP_1, EP_2, and AE was investigated using generalized linear models (GLMs). Data exploration before model fitting was carried out as described by Zuur *et al.* (2007); it involved checking for collinearities and investigating the properties of the data to set the type of GLMs to be used. All correlation coefficients and values of variance inflation factor (VIF) were low (VIF < 1.5) in terms of explanatory variables used within the same models, indicating the absence of collinearities. The response variable (CSS) comprised positive values with a positively skewed distribution, so a gamma distribution was chosen for the GLMs (Zuur *et al.*, 2009). CSS was also non-linearly related to the continuous explanatory variables, so a log-link function was implemented.

Candidate models (GLMs) were constructed including all possible combinations of explanatory variables (ER, EP_1 or EP_2, AE) plus the interaction term ER:EP. The interaction term was added as a result of some patterns being observed during the data-exploration process, as well as the fact that the possible existence of such an interaction has been suggested in the literature (Myers and Quinn, 2002).

Model selection and the estimation of model selection uncertainty were based on the information theoretic approach (Burnham and Anderson, 2002). In this context, the best models were selected based on Akaike's information criterion (AIC): AIC $= -2$(log-likelihood) $+ 2$(number of parameters) (Crawley, 2007). AIC is known as a "penalized log-likelihood", so when comparing multiple models, the smaller the AIC value, the better the model (Burnham and Anderson, 2002; Crawley, 2007). The differences in AIC between the model with the lowest AIC value (AIC_{min}) and the rest of the candidate models, $\Delta i = AIC_i - AIC_{min}$, were also computed (Burnham and Anderson, 2002; Katsanevakis, 2006). The model with the lowest AIC value and the model with the fewest parameters (i.e. the most parsimonious) within two units of the lowest AIC ($\Delta i < 2$) were identified and parameter estimates obtained (Burnham and Anderson, 2002; Katsanevakis, 2006). Model validation was carried out by checking for normality and patterns in the residuals and by plotting the residuals against the explanatory variables to check for patterns and homoscedasticity (Zuur *et al.*, 2007).

Finally, effect plots were constructed for the optimal model to visualize the separate effects of ER and EP_1 on CSS on a meta-scale when the other explanatory variables are fixed at typical (mean) values (Fox, 2003). This allows controlling for ER when exploring the impact of EP_1, and vice versa. All statistical analyses were carried out using R statistical software version 2.12.1 (R Development Core Team, 2010).

Results

A positive correlation ($r = 0.57$, $p < 0.01$) was found between EP_2 and EP_1, as expected. However, there was a distinct group of stocks with high values of EP_2 for low values of EP_1 (Figure 1). These stocks (NE Arctic cod, NE Arctic haddock, and three stocks of saithe) displayed large proportions of immature fish in the catch but also large numbers of immature fish that were not captured by the fisheries. Moreover, all five stocks were within the precautionary limits ($CSS > 1$) despite 45–73% of the fish caught being immature (Table 1).

EP_1 and EP_2 were alternatively used to quantify exploitation pattern in the GLMs. The model with the lowest AIC value (m_1) included ER, EP_1, and AE, whereas the most parsimonious within two units of the lowest AIC (m_2) included just EP_1 and ER (Table 2). This suggests that both exploitation rate and exploitation pattern as described by the historical values of proportional fishing mortality of immature fish affect current stock status. All other models were either less parsimonious or had $\Delta i > 2$, so they were not investigated further. In particular, all models including EP_2 had $\Delta i > 4.9$ (Table 2), showing that EP_1 is better at explaining the variation in CSS and that it is a more sensitive metric of exploitation pattern than EP_2. The high AIC values of models including EP_2 suggest that different proportions of immature fish in the catch have no effect on CSS because they do not correspond to the proportion of immature fish removed from the population.

The comparison of models m_1 and m_2 demonstrates that inclusion of the factor AE in the model increased the deviance explained by ~4% (Table 2). Moreover, the model including only ER (m_5) explained 18.4% of the deviance more than the model including only EP_1 (m_{12}), showing the higher explanatory power of ER than EP_1.

Both ER and EP_1 had a negative effect on CSS in both m_1 and m_2 (Tables 3 and 4). In other words, higher historical values of exploitation rate and of relative fishing mortality of immature fish decreased current stock status. Level B of factor AE also had a negative effect on CSS in m_1, indicating overestimation of CSS values when there is high bias in the estimation of F_{imm} because of the unaccounted effect of discarding, but at a non-significant level (Table 3). The residuals of both models (m_1, m_2) were normally distributed, and no patterns were observed when the residuals were plotted against the fitted values or any of the explanatory variables. The effect plots for m_1 showed that the CSS fell below the precautionary limit ($CSS < 1$) when ER exceeded 0.63 (for mean values of EP_1 and AE; Figure 2a) and when EP_1 exceeded 0.50 (for mean values of ER and AE), i.e. when F_{imm} exceeded half of F_{mat} (Figure 2b).

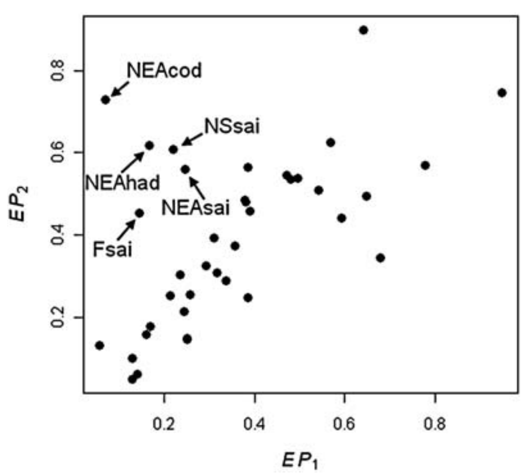

Figure 1. Plot of EP_2 against EP_1. The arrows indicate stocks with large proportions of immature fish in the catch (high values of EP_2) and small proportional fishing mortality of immature fish (low values of EP_1). NEAcod, NE Arctic cod stock; NEAhad, NE Arctic haddock stock; NEAsai, NE Arctic saithe stock; NSsai, North Sea saithe stock; Fsai, Faroe saithe stock.

Conducting the same analysis using a time interval of 20 rather than T years produced qualitatively similar results, suggesting that the effect of ER, EP_1, and EP_2 on CSS is not sensitive to the number of years over which the explanatory variables are averaged.

Discussion

The meta-analysis presented here of empirical dataseries covering 38 stocks and 13 species of the NE Atlantic shows that stock status is negatively affected by (i) a high fishing mortality of immature fish compared with the fishing mortality of mature fish, and (ii) a high overall exploitation rate.

The first conclusion demonstrates that historical exploitation patterns that result in large proportional fishing mortality of immature fish lead to lower current stock status. This supports the "spawn-at-least-once" principle which suggests that recruitment-overfishing is avoided when relatively more immature fish are given the chance to contribute to recruitment (Myers and Mertz, 1998). The result could be attributed to some extent too to the reduced growth-overfishing attributable to the capture of relatively fewer small fish (Froese *et al.*, 2008). Values of $EP_1 > 0.50$ (F_{imm} more than half F_{mat}) appear to be related to the stock status falling below precautionary limits ($CSS < 1$; Figure 2b), and it is also worth noting that the nine stocks having F_{imm} values <20% of F_{mat} ($EP_1 < 0.2$) were well within precautionary limits ($CSS > 1$; Table 1). These results suggest that fishing gears with relatively sharp selection ranges and with L_{r50} values close to L_{50} seem to be preferable to those with flat selection ranges and $L_{r50} < L_{50}$.

The second conclusion reached here illustrates the unsurprisingly negative effect of exploitation rate on stock status that has already been suggested by earlier meta-analytical studies (Sparholt *et al.*, 2007; Worm *et al.*, 2009). The effect of exploitation rate on current stock status appears to be stronger than that of exploitation pattern; the model containing ER alone (m_5) has a lower AIC value and explains a greater percentage of the deviance than the one containing just EP_1 (m_{12}; Table 2). High exploitation rates can result in both growth- and recruitment-overfishing (Myers and Quinn, 2002; Worm *et al.*, 2009), so pushing stock status below precautionary limits.

The assessment effect attributable to unaccounted discarding does not appear to be very strong. Model m_1 was only slightly better than m_2 (Table 2), and the negative effect from the non-inclusion of high discards in the assessment was not statistically significant (Table 3). Nevertheless, the outcome of stock assessments has been reported to be highly sensitive to the inclusion of discards and to the raising procedures (ICES, 2007). It is therefore expected that assessments producing less biased estimates of F as a result of the inclusion of discards would lead to more accurate results for meta-analyses based on ICES data.

The F-based metric of exploitation pattern (EP_1) constructed within this study appears to be more sensitive and efficient than the catch-based proxy (EP_2) proposed by Froese (2004).

Table 1. The dataset used in the analysis.

ICES Working Group	Species	Area	Report time-series	Time interval for ER, EP	Age classes	Age classes for ER	A_{50} (years)	ER (year^{-1})	EP_1	EP_2	CSS	AE	Comments on discarding	Other comments
NWWG	Cod	Faroe (Vb)	1961–2008	1997–2008	2–10	3–7	3.12	0.599	0.234	0.303	0.481	A	No discards mentioned	Age 1 reconstructed
WGBFAS	Cod	West Baltic (22–24)	1970–2008	1996–2008	1–7	3–6	3.14	1.069	0.495	0.538	1.152	A	Discards included in the assessment	–
WGNSSK	Cod	North Sea (IV, IIIaN, VIId)	1963–2008	1994–2008	1–7	2–4	3.73	0.866	0.642	0.898	0.298	A	Discards included in the assessment	–
AFWG	Cod	NE Arctic (I, II)	1946–2008	1983–2008	3–13	5–10	6.61	0.697	0.070	0.729	1.505	A	Low discard rates (less than 10%)	Ages 1, 2 reconstructed
WGCSE	Cod	West of Scotland (VIa)	1978–2008	2001–2008	1–7	2–5	2.11	0.962	0.778	0.570	0.243	A	Discards included in the assessment	$F = Z - 0.2$
WGCSE	Cod	Irish Sea (VIIa)	1968–2008	2001–2008	1–5	2–4	2.02	1.387	0.381	0.479	0.204	B	High. Underestimated F at age 1	–
WGSSDS	Cod	Celtic Sea (VIIe–k)	1971–2007	1999–2007	1–7	2–5	2.30	0.847	0.472	0.546	0.534	B	High. Up to 65% after 2003	No assessment conducted in 2009
WGNSSK	Haddock	North Sea (IV, IIIa)	1963–2008	1999–2008	0–8	2–4	2.62	0.411	0.542	0.508	1.706	A	Discards included in the assessment	Age 0 excluded ($F < 0.04$)
AFWG	Haddock	NE Arctic (I, II)	1950–2008	1986–2008	3–11	4–7	5.79	0.381	0.166	0.618	2.534	B	No data. Discarding might present a serious problem	Ages 1, 2 reconstructed
NWWG	Haddock	Faroe (Vb)	1957–2008	1997–2008	1–10	3–7	3.04	0.358	0.160	0.157	1.286	A	Minimal discarding	–
WGCSE	Haddock	West of Scotland (Via)	1978–2008	2001–2008	1–8	2–6	1.99	0.556	0.592	0.441	1.196	A	Discards included in the assessment	–
WGCSE	Haddock	Rockall (VIb)	1991–2008	1999–2008	1–7	2–5	2.50	0.518	0.390	0.459	2.070	A	Discards included in the assessment	–
NWWG	Saithe	Faroe (Vb)	1961–2008	1987–2008	3–12	4–8	5.54	0.458	0.145	0.452	1.109	A	Little discarding	Ages 1, 2 reconstructed
WGNSSK	Saithe	North Sea (IV, IIIaN, VI)	1963–2008	1990–2008	3–10	3–6	4.75	0.376	0.219	0.609	1.337	A	Assumed to be a small problem	Ages 1, 2 reconstructed
AFWG	Saithe	NE Arctic (I, II)	1960–2008	1988–2008	3–11	4–7	5.16	0.285	0.246	0.560	3.925	A	Not considered a problem	Ages 1, 2 reconstructed
WGCSE	Whiting	Celtic Sea (VIIe–k)	1982–2008	2004–2008	1–8	2–5	1.16	0.793	0.139	0.062	1.264	B	Discarding a major feature of the fishery	–
WGNSSK	Norway pout	North Sea (IV, IIIaN)	1983–2008	2005–2008	0–4	1–2	1.08	0.100	0.479	0.536	0.831	A	Not mentioned	Age 0 excluded ($F < 0.03$), quarters transformed to years
WGHMM	Hake	Southern stock (VIIIc, IXa)	1982–2008	1997–2008	1–8	2–5	3.09	0.498	0.569	0.624	0.574	B	Discard rate considered to be high at young ages	–
WGHMM	Hake	Northern stock (IIIa, IV, VI, VII, VIIIa,b)	1978–2008	1994–2008	1–8	2–6	3.80	0.299	0.385	0.565	0.936	B	Underestimated F at younger ages	–

Continued

Table 1. *Continued*

ICES Working Group	Species	Area	Report time-series	Time interval for *ER*, *EP*	Age classes	Age classes for *ER*	A_{50} (years)	*ER* (year^{-1})	EP_1	EP_2	*CSS*	*AE*	Comments on discarding	Other comments
WGNSSK	Sandeel	North Sea (IV)	1983–2008	2003–2008	0–4	1–2	1.51	0.558	0.948	0.747	0.641	A	Not mentioned	Age 0 excluded ($F < 0.07$)
WGCSE	Plaice	Irish Sea (VIIa)	1964–2008	1997–2008	2–9	3–6	3.00	0.263	0.257	0.255	2.377	B	High levels of discarding	Age 1 reconstructed
WGCSE	Plaice	Celtic Sea (VIIf,g)	1977–2008	1998–2008	1–9	3–6	2.85	0.556	0.292	0.326	0.561	B	Heavy discarding of youngest age classes	–
WGNSSK	Plaice	North Sea (IV)	1957–2008	1999–2008	1–10	2–6	2.49	0.501	0.647	0.495	1.226	A	Discards included in the assessment	–
WGNSSK	Plaice	Eastern Channel (VIId)	1980–2008	1998–2008	1–10	3–6	2.86	0.743	0.379	0.485	0.524	B	Substantial discarding, up to 79%	–
WGCSE	Plaice	Western Channel (VIIe)	1976–2008	1998–2008	1–10	3–7	2.85	0.596	0.357	0.374	0.620	B	Variable discarding	–
WGCSE	Sole	Celtic Sea (VIIf,g)	1971–2008	1997–2008	1–10	4–8	3.05	0.428	0.316	0.307	1.446	A	Low discard rates (up to 5%)	–
WGCSE	Sole	Irish Sea (VIIa)	1970–2008	1999–2008	2–8	4–7	2.45	0.415	0.250	0.146	0.478	A	Low discard rates (0–8%)	Age 1 reconstructed
WGNSSK	Sole	North Sea (IV)	1957–2008	1999–2008	1–10	2–6	2.50	0.508	0.213	0.252	0.819	B	High discard rates (6–29%)	–
WGNSSK	Sole	Eastern Channel (VIId)	1982–2008	1999–2008	1–11	3–8	2.50	0.436	0.336	0.289	1.390	A	Discards not substantial	–
WGBFAS	Sole	Skagerrak and Kattegat (IIIa)	1984–2008	1999–2008	2–9	4–8	2.50	0.301	0.130	0.099	2.427	A	Negligible discarding	Age 1 reconstructed
WGHMM	Sole	Bay of Biscay (VIIa,b)	1984–2008	1999–2008	2–8	3–6	2.38	0.504	0.244	0.214	1.008	B	Discards may be important at age 1	Age 1 reconstructed
NWWG	Herring	Iceland (Va)	1986–2008	1995–2008	3–12	5–10	3.45	0.292	0.169	0.177	2.219	A	No discarding	Ages 1, 2 reconstructed
HAWG	Herring	Celtic Sea (VIIa(S),g,h,j,k)	1958–2008	2005–2008	1–6	2–5	1.00	0.281	0.129	0.050	1.132	A	Discarding not a feature of the fishery	–
HAWG	Herring	North Sea (IV, IIIa, VIId)	1960–2008	2002–2008	0–9	2–6	1.86	0.297	0.311	0.392	0.817	A	Discards included in the assessment	Age 0 excluded ($F < 0.03$)
WGWIDE	Herring	NE Atlantic (IIa,b, Va,b, XIVa)	1988–2008	1992–2008	0–15	5–14	4.29	0.143	0.057	0.132	2.390	A	Very low discarding	Age 0 excluded ($F < 0.001$)
WGWIDE	Mackerel	NE Atlantic (IIa, IIIa,b,d, IV, Va, Vb, VI, VII, VIIIa,b,d,e, XII, VIIIc, IXa, XIV)	1972–2008	2001–2008	0–12	4–8	2.00	0.327	0.250	0.147	1.083	A	Discards included in the assessment	Age 0 excluded ($F < 0.02$)
WGWIDE	Horse Mackerel	Western Stock (IIa, IIIa (W), IVa, Vb, VIa, VIIa–c, VIIe–k, VIIIa–e)	1982–2008	1995–2008	0–11	4–8	3.55	0.085	0.679	0.345	1.518	A	Discards included in the assessment	Age 0 excluded ($F < 0.01$)
WGWIDE	Blue Whiting	NE Atlantic (I–IX, XII, XIV)	1981–2008	2000–2008	1–10	3–7	2.35	0.424	0.385	0.247	2.648	A	Minor discarding	–

Comments on discarding are as expressed in respective ICES reports. A_{50}, age at 50% maturity; *ER*, mean exploitation rate ($\bar{F}$), Equation (2); EP_1, mean F_{imm}/F_{mat}, Equation (5); EP_2, mean proportion of immature fish in the catch, Equation (7); *CSS*, average B/B_{pa} in the last 3 years of every assessment, Equation (1), and *AE*, assessment effect. A indicates low bias and B higher bias.

Table 2. Evaluation of candidate models for *CSS* using either EP_1 or EP_2 to quantify exploitation pattern.

Model	Formula	Deviance explained (%)	AIC	Δi
m_1	$c + ER + EP_1 +$ factor (AE)	44.3	66.30	0.0
m_2	$c + ER + EP_1$	40.4	67.00	0.70
m_3	$c + ER{:}EP_1 + ER + EP_1 +$ factor (AE)	44.8	67.98	1.68
m_4	$c + ER{:}EP_1 + ER + EP_1$	40.5	68.97	2.67
m_5	$c + ER$	33.6	69.27	2.97
m_6	$c + ER +$ factor (AE)	35.4	70.22	3.92
m_7	$c + ER + EP_2$	33.7	71.20	4.90
m_8	$c + ER + EP_2 +$ factor (AE)	35.6	72.06	5.76
m_9	$c + ER{:}EP_2 + ER + EP_2 +$ factor (AE)	38.1	72.49	6.19
m_{10}	$c + ER{:}EP_2 + ER + EP_2$	34.6	72.67	6.37
m_{11}	$c + EP_1 +$ factor (AE)	24.6	76.37	10.07
m_{12}	$c + EP_1$	15.2	79.08	12.78
m_{13}	$c +$ factor (AE)	6.1	83.24	16.95
m_{14}	$c + EP_2 +$ factor (AE)	9.8	83.56	17.27
m_{15}	c	0	83.72	17.42
m_{16}	$c + EP_2$	4.2	83.97	17.68

AIC, Akaike's information criterion; Δi, Akaike's differences.

Table 3. Summary of the optimal model selected for *CSS* (m_1).

Coefficient	Estimate	s.e.	*t*-value	*p*-value
Intercept	1.240	0.210	5.913	1.12e−06
ER (year^{-1})	−1.232	0.344	−3.582	0.001
EP_1	−1.058	0.432	−2.450	0.020
Factor(AE) B	−0.294	0.190	−1.547	0.131

s.e., standard error.

Table 4. Summary of the second best model selected for *CSS* (m_2).

Coefficients	Estimate	s.e.	*t*-value	*p*-value
Intercept	1.157	0.218	5.320	6.09e−06
ER (year^{-1})	−1.360	0.342	−3.982	3.29e−04
EP_1	−0.883	0.444	−1.986	0.055

s.e., standard error.

Presumably, this is because EP_1 is a direct metric of exploitation pattern that takes into account the whole fish population, whereas EP_2 is a proxy that ignores the immature fish that are not captured and will potentially be allowed to spawn in future. Plotting EP_2 against EP_1 (Figure 1) reveals that the stocks appearing as outliers are those with a large proportion of immature fish in the catch in conjunction with low F_{imm} values relative to F_{mat}. The fact that all five outlier stocks are within precautionary limits ($CSS > 1$) despite the large proportions of immature fish in the catch also illustrates the greater sensitivity of EP_1.

The quantity $F_{\mathrm{imm}}/F_{\mathrm{mat}}$ could be used potentially as an indicator of fisheries sustainability for data-rich stocks where values of *F*-at-age are available. It has the main desirable characteristics for this purpose (Liu and Ou, 2007): (i) scientific validity; (ii) availability over broad scales of time and space; (iii) acceptability, simplicity, and relevance to fisheries; and (iv) ease of monitoring. $F_{\mathrm{imm}}/F_{\mathrm{mat}}$ could be implemented in fisheries management to describe exploitation pattern in the same way that $\overline{F}$ is used today to represent exploitation rate. Further investigation on a stock-by-stock basis would be needed though to suggest stock-specific limits of exploitation pattern analogous to those used

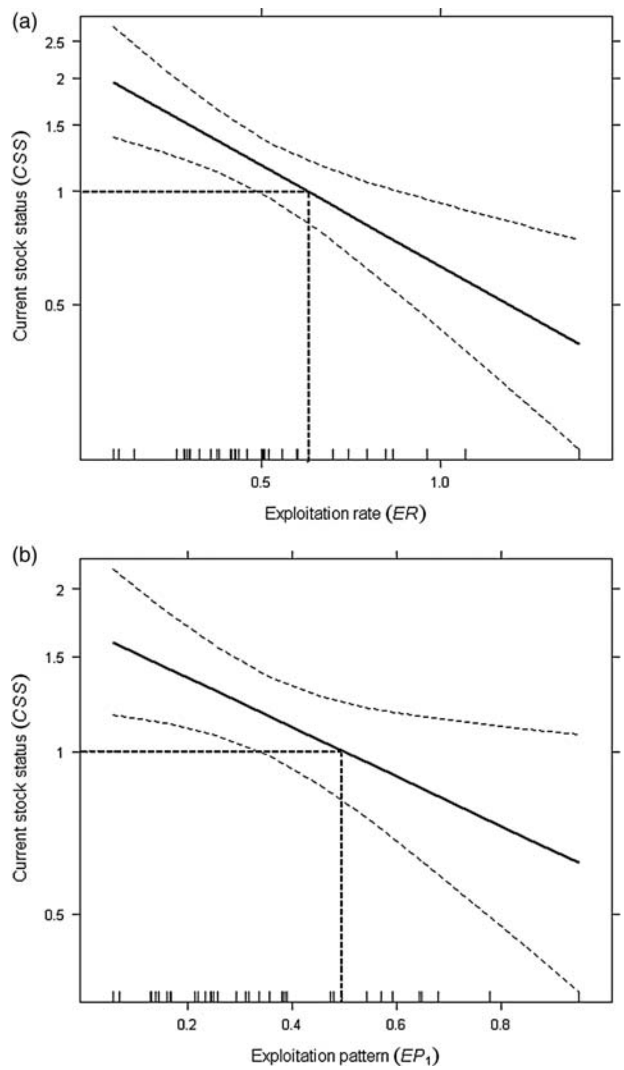

Figure 2. Effect displays for the continuous explanatory variables of model m_1. Curved dashed lines indicate the 95% confidence intervals. Straight dashed lines indicate ER and EP_1 values for $CSS = 1$. (a) Effect display of ER; (b) effect display of EP_1.

today for exploitation rate, so that exploitation pattern could become part of fisheries advice. For fish stocks where *F*-at-age is not available, EP_2 could be used as an indicator of sustainability only if it can be confirmed that it is representative of the pressure on the immature fish in the population, i.e. if there are not great numbers of immature fish that remain unfished.

Another conclusion that can be derived from this type of analysis is the existence of a trade-off between exploitation rate and exploitation pattern; the lower the exploitation rate, the less the importance of EP_1 and vice versa. Consequently, either a combination of a relatively high exploitation rate with a low relative exploitation of immature fish, or a combination of a high relative exploitation of immature fish with a low exploitation rate, could improve stock status. For example, the NE Arctic cod stock and the Celtic Sea whiting stock are in good condition despite being exploited at quite high rates owing to a very low relative exploitation of immature fish (Table 1). On the other hand, the saithe fisheries of the NE Arctic and North Sea exploit immature fish more

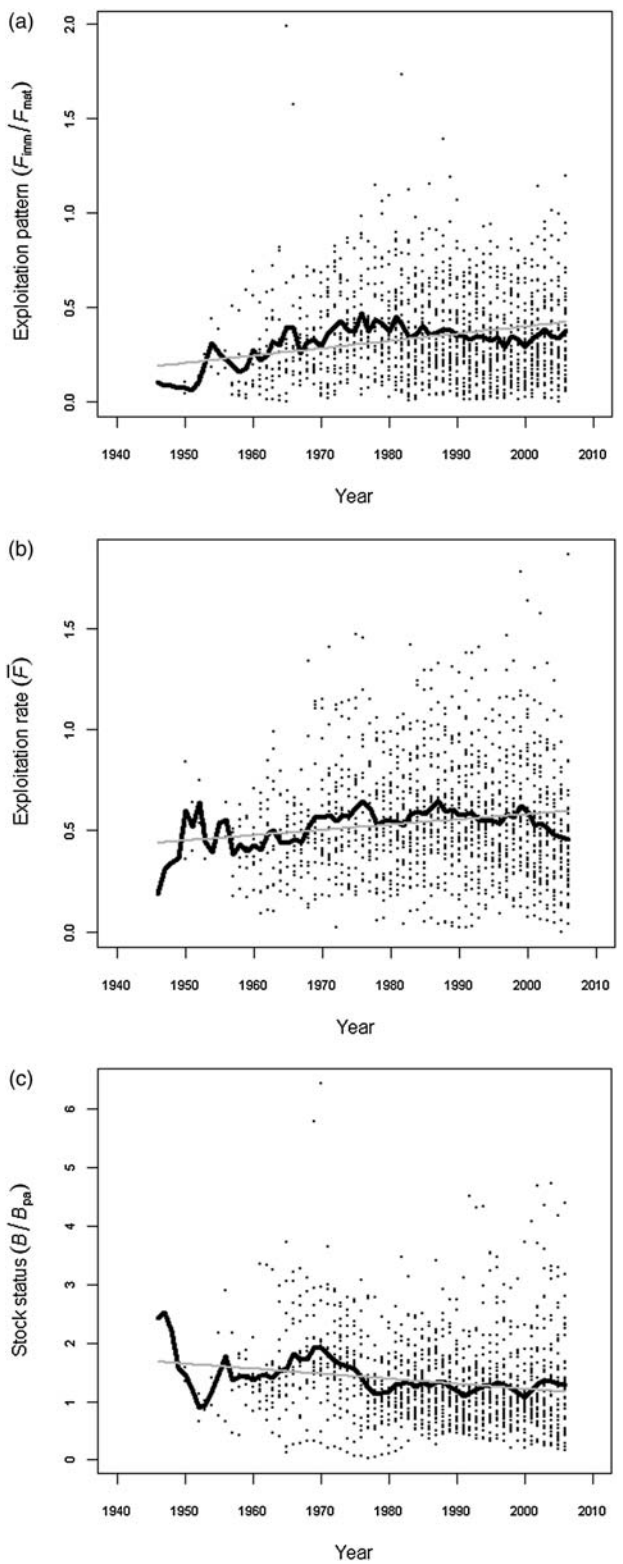

Figure 3. (a) Exploitation pattern, (b) exploitation rate, and (c) stock status for all 38 stocks over time. The black line reflects the mean values by year and the grey line the linear regression. One data point ($F_{imm}/F_{mat} = 4.6$ in 1978 for the North Sea herring stock) was omitted from (a) as an outlier.

heavily but good stock status is sustained by lower exploitation rates (Table 1). Moreover, the independent effects of exploitation rate and exploitation pattern on stock status should be taken into consideration in management plans where fishers are compensated for using more selective gears by being given more days at sea (e.g. the Scottish conservation credits scheme; Edwards, 2009). Protection of immature fish is not a panacea; if the cost of improving exploitation pattern is to increase exploitation rate, then the management plans might not fulfil their goals. This agrees with the perception that fisheries managers should not rely on measures regulating either exploitation rate or exploitation pattern, but both (Halliday and Pinhorn, 2002; Caddy and Agnew, 2004).

To evaluate the general status of the 38 fish stocks used in this study, the trends of F_{imm}/F_{mat}, $\overline{F}$, and B/B_{pa} of all stocks for the period 1946–2006 were also investigated. Estimates for the most recent 2 years were excluded because of their low precision (Pope, 1972). This analysis is directly comparable with the one conducted by Sparholt *et al.* (2007), who studied the trends of exploitation rate and stock status for the period 1946–2003 using a similar dataset of 38 stocks assessed by ICES. Froese and Proelss (2010) also conducted a similar meta-analysis for 54 fish stocks in the NE Atlantic. F_{imm}/F_{mat} has overall an increasing trend, but it appears to have been more or less stable in the past 20 years (Figure 3a). Meanwhile, there is a clear decreasing trend of exploitation rate in the last 8 years of the time-series (Figure 3b). The fact that this decreasing trend of exploitation rate has not resulted in a noticeable improvement in stock status (Figure 3c) possibly indicates that a twofold management approach aiming at reducing both exploitation rate and proportional exploitation of immature fish might be more effective. The trends of exploitation rate and stock status presented here (Figure 3b and c) are similar to those observed by Sparholt *et al.* (2007) and Froese and Proelss (2010).

The results of this study must, however, be interpreted with caution because they are a meta-analysis of data ($N_{a,y}$, $F_{a,y}$, B_y), which are outputs of other models (mostly VPA type) used by ICES to assess fish stocks and which accordingly include a certain amount of uncertainty. Moreover, the primary data ($C_{a,y}$, maturity ogives) of the stock assessment reports are also prone to misreporting and/or bias (Pinhorn and Halliday, 2001); for instance, there are stocks where the maturity ogive is updated every year (e.g. NE Arctic cod) and others where it has not been updated for decades (e.g. blue whiting). The exploitation pattern metrics used here also assume the same vulnerability to fishing for immature and mature fish within the same age classes, which might not be the case for species such as herring or blue whiting that are caught in spawning areas or when undergoing spawning migrations (Horbowy, 2005). Therefore, there is a need for stock assessment reports to provide more accurate information on the maturity ogives of exploited stocks and on the maturity stage of the fish caught in the future. Finally, defining current stock status based on B_{pa} is in accordance with ICES practice, but B_{pa} is not well defined and can be formulated differently between stocks. Nevertheless, this type of meta-analysis makes general trends and patterns clearer and can help to synthesize the results of individual stock assessments into good management strategies (Sparholt *et al.*, 2007).

The results of this study describe the independent effects of exploitation rate and pattern on fisheries sustainability. They do not consider important stock-specific features such as different levels of resilience to the same levels of exploitation rate and exploitation pattern and/or the role of environmental variability (e.g. Fréon *et al.*, 2005). Such effects are expected to account for the large amount of residual deviance that our models did not explain (ca. 56% for m_1). The results suggest that immature fish should be protected from fishing (e.g. through the implementation of more selective fishing gears) and that fisheries management could benefit by placing more emphasis on exploitation pattern, which is currently overlooked in favour of exploitation-rate-based approaches.

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