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Original Article

Estimating F_{msy} from an ensemble of data sources to account for density dependence in Northeast Atlantic fish stocks

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A new approach for estimating the fishing mortality benchmark F_{msy} (fishing pressure that corresponds to maximum sustainable yield) is proposed. The approach includes density-dependent factors. The analysis considers 53 data-rich fish stocks in the Northeast Atlantic. The new F_{msy} values are estimated from an ensemble of data sources: (i) applying traditional surplus production models on time-series of historic stock sizes, fishing mortalities, and catches from the current annual assessments; (ii) dynamic pool model (e.g. age-structured models) estimation for stocks where data on density-dependent growth, maturity, and mortality are available; (iii) extracts from multispecies and ecosystem literature for stocks where well-tested estimates are available; (iv) the "Great Experiment" where fishing pressure on the demersal stocks in the Northeast Atlantic slowly increased for half a century; and (v) linking F_{msy} to life history parameters. The new F_{msy} values are substantially higher (average equal to 0.38 year^{-1}) than the current F_{msy} values (average equal to 0.26 year^{-1}) estimated in stock assessments and used by management, similar to the fishing pressure in the 1960s, and about 30% lower than the fishing pressure in 1970–2000.

Keywords: density dependent, ecosystem, F_{msy}, fisheries, meta-analysis, management

Introduction

Overfishing has been, and still is, a major problem worldwide. In previous decades, when many stocks in the Northeast Atlantic were overexploited and fisheries managers became increasingly pressured to reduce effort, it became clear that management approaches had to be precautionary to promote rebuilding and limit the risk of collapses under sustained fishing pressure. More recently, this precautionary approach has been supplemented by

several strong international agreements and policies highlighting maximum sustainable yield (MSY) as a central reference point for management: United Nations Convention on the Law of the Sea (UN, 1982), United Nations Straddling Fish Stocks Agreement of 1995 (UN, 1995), the FAO Code of Conduct for Responsible Fisheries (FAO, 1995), and the Johannesburg Declaration of the World Summit on Sustainable Development (UN, 2002). All countries (some via the European Union) around the Northeast Atlantic have made corresponding policy and legal instruments (Anonymous, 1990, 2004, 2008, 2017; EU, 2013). Management interventions have, in many areas, led to a partial end to overfishing, with some fish stocks rebuilding and others already rebuilt (Hilborn and Ovando, 2014; FAO, 2018; Hilborn *et al.*, 2020; ICES, 2018). Where this is the case—notably in temperate parts of the world ocean where effective management is in place—it raises a new twist to the central question for fisheries management: how should fisheries for rebuilt populations be managed to obtain MSY?

The primary benchmark for comparing current fishing pressure is F_{msy}, the fishing pressure expected to result in MSY over the long term. The standard assessment approaches used in the Northeast Atlantic area for estimating F_{msy} do not account for density-dependent processes other than recruitment. Density can also affect growth, maturation, and natural mortality; failing to account for these processes in stock assessments can result in biased estimates of F_{msy} (see Lorenzen, 2016 for a review). As populations rebuild, interactions such as predation and food competition strengthen, leading to higher mortality and slower growth, basic elements of ecosystem dynamics that determine ecosystem carrying capacity and density-dependent mechanisms.

Ecosystem and multispecies modelling indicate that the downward bias in F_{msy} estimates, from failing to account for these interactions, could be substantial (Gislason, 1999; Collie *et al.*, 2003; ICES, 2012, 2013). This is expected to result in forgone sustainable yield. At the same time, the global human population is demanding more food. Fish products are healthy and have a low carbon footprint compared to most meat produced on land, and well-managed fisheries are relevant for as many as 10 of the 17 United Nations Sustainable Development Goals for 2030 (FAO, 2018).

As early as the 1970s, the "North Sea model" (Andersen and Ursin, 1977) demonstrated the importance of both predation and food competition, but it has proven difficult to use such knowledge in fisheries management. In addition, there have been challenges due to a lack of clear management objectives, lack of capacity to address trade-offs between competing fisheries, and structural problems on the scientific side with a gap between the science available on ecosystem functioning and that of management advice for individual stocks.

Scientific evidence is accumulating that aiming for overall maximum MSY (in terms of tonnes extracted) from a marine ecosystem would involve fishing higher trophic levels at higher rates than indicated by single-species F_{msy} because the forgone catch from higher trophic levels will be more than compensated by increases in catches from lower trophic levels (see e.g. Gislason, 1999; Pope *et al.*, 2006; Andersen *et al.*, 2015; Szuwalski *et al.*, 2017; Zhou *et al.*, 2019). Overfishing the predators, however, may result in predator biomass declining below the biomass limit for management (B_{lim}) and is rarely acceptable to management and the public, or in line with biodiversity goals by the UN Convention on Biological Diversity (CBD) (UN, 1992). The approach suggested here does not take the route of aiming for an overall maximum MSY from the marine ecosystem; rather, we consider the maximum sustainable yield from each individual species, considering density-dependent (DD) effects. The current management in the Northeast Atlantic implies a very low risk for stocks getting below B_{lim} by reducing F when the stock size decreases below a buffer biomass (in ICES called B_{pa} or $MSYB_{trigger}$) well above B_{lim} and, therefore, can be considered to living up to the CBD goals.

For a few very data-rich stocks, it is possible to directly include density dependence in the current cohort-based management strategy evaluation analysis of harvest control rules (HCR) and their reference points. This has been done, for example, for Northeast Arctic cod (*Gadus morhua*) (ICES, 2016), and this represents a viable method for including density dependence in reference point calculations. However, for most stocks, scientific knowledge about each DD factor is not yet available, and we, therefore, propose a simpler, pragmatic approach to include density dependence in the existing management reference points where the more detailed version is not feasible.

We propose an approach that is simple, scientifically sound, builds on the existing stock assessment framework, and removes some known biases in the current methodology. An ensemble approach is applied that builds on the existing single-species stock assessment framework. We aim for alternative estimates of F_{msy} that implicitly consider density-dependent recruitment, growth, maturity, and cannibalism. The approach does not explicitly include multispecies interactions and hence does not address trade-offs between stocks in how the proposed set of F_{msy} values could be used for management.

We evaluated the impact of considering density dependence for Northeast Atlantic fisheries, FAO Area 27 (FAO, 2018), which currently account for about 9 million tonnes of catch annually. This catch corresponds to 11% of global capture fisheries. The annual catch in the area increased in the 1950s and 1960s to reach a maximum of 15 million t in the mid-1970s (Supplementary Figure S1). We focused on the 53 most data-rich and commercially important fish stocks in the area representing an annual catch currently around 5.5 million tonnes. We did not include short-lived forage fish in these analyses because their management in the Northeast Atlantic is not based on F_{msy}.

An ensemble approach was taken for estimating F_{msy}, drawing from a variety of data sources. For Northeast Atlantic stocks, the approach relies mostly on surplus production models, but we also consider other sources of F_{msy} estimates to place the results into a wider context. We applied up to five approaches to estimate F_{msy} for each of the 53 fish stocks: (i) surplus production models (Schaefer, 1957; Pella and Tomlinson, 1969), using time-series of catch, fishing mortality, and stock biomass from stock assessments (ICES, 2018); (ii) extraction of F_{msy} from the literature on ecosystem and multispecies analysis; (iii) direct calculations based on submodels for density dependence of growth, reproduction, and cannibalism; (iv) the "Great Experiment" where fishing pressure on the demersal stocks in the Northeast Atlantic slowly increased over half a century, and catches initially increased, but then decreased as fishing pressure crossed the boundary to overfishing (Sparholt and Cook, 2010); and (v) generalized linear regression linking F_{msy} from approaches (i)–(iii) to life history parameters. Surplus production models are often used in data-poor situations, but we use them here with abundant

stock assessment data because they implicitly include density-dependent effects.

Material and methods

The stocks included in the present study are the so-called "data-rich" stocks named by ICES as "Category 1" stocks (ICES, 2018). Some of the ICES Category 1 stocks were, however, found unfit for our methodology and analysis, and they were excluded. The excluded stocks are: (i) short-lived, forage fish like sandeel (*Ammodytes marinus*), capelin (*Mallotus villosus*), and Norway pout (*Trisopterus esmarkii*) because they have a management control rule where F_{msy} is not relevant; (ii) ill-defined stock units where the separation between neighbouring stocks is very uncertain; (iii) stocks with relative rather than absolute fishing mortality estimates; (iv) all shellfish and elasmobranch stocks as they have very different population dynamics compared with teleost fish species; (v) stocks where most of the catch data are estimated rather than sampled; and (vi) stocks that mostly have experienced fishing mortalities much lower than natural mortalities and, therefore, have a stock-size development over time that is a result of natural variability rather than variability in fishing pressure. This left us with the 53 category 1 stocks listed in Table 1.

The surplus production modelling (SPM) used time-series of catch, fishing mortality, and stock biomass taken from routine assessments by ICES (2018). Data from Froese *et al.* (2016) on F/F_{msy} were used in combination with ICES time-series of F to get F_{msy} in the ICES "currency" of F (i.e. an average F over age groups specific for each stock). Multispecies and ecosystem peer-reviewed literature of robust and well-tested models were used to extract F_{msy} values for six stocks (Collie *et al.*, 2003; ICES, 2012, 2013). For dynamic pool models, density dependence in recruitment, growth, maturity, and natural mortality were taken from ICES Benchmark Workshops (see references in ICES, 2018) to calculate F_{msy} values for five stocks where such information was available. Life history parameters were taken from ICES fish stock assessment working group reports (see references in ICES, 2018).

The central method employed is the surplus production model to estimate stock trends and F_{msy} for each of the 53 stocks in our dataset. These F_{msy} estimates will not be directly comparable with the ICES estimates, which were derived from different models. This is because different models will generally agree more closely on trends in the stock dynamics than they do on absolute biomass levels. Furthermore, the ICES F_{msy} estimates themselves will have been derived from different models for different stocks. We, therefore, do not use the SPM-derived estimates directly. Rather, we consider the ratio of F/F_{msy} that gives a measure of how much above or below F_{msy} the current fishing is. We then apply that ratio to the ICES F value to produce an estimate of the F_{msy}, which is consistent with the model used for that stock. This procedure translates the F_{msy} from the SPM to the assessment model and, in doing so, avoids the issue of absolute level of the variables.

SPM-based approach

We use the approach of the model-extended RAM Legacy Stock Assessment Database [RAMLDB; RAM Legacy Stock Assessment Database, vers. 4.44 (2019). http://doi.org/10.5281/zenodo.2542919]. RAMLDB is a compilation of assessment time-series for commercially exploited marine fish populations from around the globe (Ricard *et al.*, 2012). SPMs are used to estimate F/F_{msy} and the annual surplus production (SP_{t_obs}) observed in each year, t, as the sum of the change in stock biomass (B) and the catch (C):

$$SP_{t_{obs}} = B_t - B_{t-1} + C_t.$$

The predicted annual surplus production SP_{t_pred} is calculated based on the Pella–Tomlinson model (Pella and Tomlinson, 1969):

$$SP_{t_{pred}} = \left[\left(\frac{\varphi}{\varphi - 1}\right) \times B_t \times ER_{MSY}\right] - \left[\frac{ER_{MSY} \times B_t{}^{\varphi}}{(\varphi - 1) \times B_{MSY}{}^{(\varphi-1)}}\right],$$

where φ is the shape parameter for the production curve. Values φ of 2 and 1.736 correspond to the Schaefer curve and to the mean of 141 stocks in a meta-analysis by Thorson *et al.* (2012), respectively. ER_{MSY} is the exploitation rate at MSY (like F_{msy}, but treated as an annual fraction harvested), where ER is catch biomass divided by stock biomass. In the analysis, it is important that the stock biomass metric is relevant for SPMs, i.e. that it is exploitable biomass and not other measures of biomass such as spawning-stock biomass (SSB). Where necessary, a conversion from SSB to exploitable biomass was performed by a General Linear Mixed Model analysis linking the ratio of exploitable biomass to SSB to life history parameters (see Supplementary material).

A robustness analysis was performed to evaluate which of several alternative SPMs performed best in terms of cross-validations with assessment-estimated reference point values. The Schaefer (1959) model ($\varphi = 2$), the "general Thorson *et al.* (2012)" model ($\varphi = 1.736$), and the "taxa-based Thorson *et al.* (2012)" model (Pleuronectiformes $\varphi = 1.406$, Gadiformes $\varphi = 2.027$, Perciformes $\varphi = 0.799$, Clupeiformes $\varphi = 1.427$, Scorpaeniformes $\varphi = 3.377$, others $\varphi = 1.026$) were the three best models overall and were used in the present study (see Supplementary material for details).

The 53 fish stocks were subjected to a series of filters (Table 2) before the analysis was conducted, and, if failed, the estimates were rejected. Five stocks failed due to the time-series length, and two stocks failed due to the other criteria. The observed annual production against exploitable biomass for the stocks (normalized to MSY and k, respectively, from the "general Thorson *et al.*, 2012 model") that passed the filter are shown in Figure 1. Here, the large variability in the production in different years is obvious, but it is also obvious that there is a clear dome-shaped relationship between surplus production and stock size, which is consistent with the classic surplus production model curves. This indicates that SPMs are reflecting observed fish population dynamics.

The outcome of this analysis was the ratio F/F_{msy} for each stock and year in the time-series, and this ratio was linked to the ICES time-series of F to obtain F_{msy} in the ICES F-"currency", which is a mean over some age groups and based on numbers rather than biomass. For a year where the SPM estimate $F/F_{msy} = 1$, the fishing pressure that year obviously is estimated to be equal to F_{msy}. From the ICES time-series of F, we have an estimate of F that year, and this F value must then be F_{msy} in the ICES "currency" of F. For all other years where F/F_{msy} is not equal to 1, we can similarly estimate F_{msy}, e.g. if F/F_{msy} in a given year is 1/1.3, F_{msy} in the ICES F-"currency" must then be 1.3 times the F value from the ICES time-series for that year. F_{msy}

Table 1. Estimates of F_{msy} by stock and method.

#	Column identifier / Stock name—short	a ICES 2018	b Froese *et al.* SPM	c RAM Legacy Database, Schaefer	d RAM Legacy Database, Thorson "Taxonomic"	e RAM Legacy Database, Thorson "general"	f Ecosystem model	g Dynamic pool models, e.g. PROST	h Average of b, average (c–e), f and g	i GLM of h, based on life history parameters	j Final recommended F_{msy} values—column i unless there are ecosystem or dynamic pool estimates and then a mean of column h and i	Full stock name (truncated to save space)
1	reb.27.1-2		0.06	0.14	0.20	0.15			0.11	0.13	0.13	Beaked redfish in subareas 1 and 2 (Northeast Arctic)
2	bli.27.5b67	0.12	0.11						0.11	0.22	0.22	Blue ling in subareas 6–7 and division 5.b (Celtic Seas, English . . .
3	whb.27.1-91214	0.32	0.37	0.31		0.28			0.33	0.44	0.44	Blue whiting in subareas 1–9, 12, and 14 (Northeast Atlantic and . . .
4	cod.27.5a		0.63	0.45	0.39	0.44		0.70	0.59	0.43	0.51	Cod in division 5.a (Iceland grounds
5	cod.27.7a	0.44	0.95	0.75		0.66			0.83	0.76	0.76	Cod in division 7.a (Irish Sea)
6	cod.27.7e-k	0.35	0.56	0.51		0.47			0.52	0.63	0.63	Cod in divisions 7.e–k (eastern English Channel and southern . . .
7	cod.27.47d20	0.31	0.70	0.73	0.41	0.68	0.87	0.70	0.72	0.71	0.71	Cod in subarea 4, division 7.d, and subdivision 20 (North Sea, . . .
8	cod.27.1-2	0.40	0.55	0.51	0.46	0.50		0.60	0.55	0.38	0.47	Cod in subareas 1 and 2 (Northeast Arctic)
9	cod.27.5b1	0.32	0.36	0.57	0.52	0.57			0.46	0.60	0.60	Cod in subdivision 5.b.1 (Faroe Plateau)
10	cod.27.22-24	0.26	0.62						0.62	0.51	0.51	Cod in subdivisions 22–24, western Baltic stock
11	ldb.27.8c9a	0.193	0.33	0.33	0.24	0.32			0.31	0.44	0.44	Four-spot megrim in divisions 8.c and 9.a (southern Bay of Biscay . . .
12	reg.27.1-2	0.0525	0.10						0.10	0.14	0.14	Golden redfish in subareas 1 and 2 (Northeast Arctic)
13	reg.27.561214	0.097	0.14	0.11	0.08	0.10			0.12	0.14	0.14	Golden redfish in subareas 5, 6, 12, and 14 (Iceland and Faroes . . .
14	had.27.5a		0.47	0.33		0.31			0.40	0.38	0.38	Haddock in division 5.a (Iceland grounds)
15	had.27.5b	0.165	0.28	0.39	0.36	0.39			0.33	0.46	0.46	Haddock in division 5.b (Faroes grounds)
16	had.27.6b	0.20	0.31						0.31	0.39	0.39	Haddock in division 6.b (Rockall)
17	had.27.7a	0.27	0.41						0.41	0.43	0.43	Haddock in division 7.a (Irish Sea)
18	had.27.7b-k	0.40	0.87						0.87	0.67	0.67	Haddock in divisions 7.b–k (southern Celtic Seas and English . . .
19	had.27.46a20	0.19		0.47	0.71	0.51	0.58		0.57	0.35	0.46	Haddock in subarea 4, division 6.a, and subdivision 20 (North Sea, . . .
20	had.27.1-2	0.35	0.43	0.30	0.24	0.29			0.35	0.26	0.26	Haddock in subareas 1 and 2 (Northeast Arctic)
21	hke.27.8c9a	0.25	0.59	0.51	0.43	0.50			0.54	0.65	0.65	Hake in divisions 8.c and 9.a, Southern stock (Cantabrian Sea and . . .
22	hke.27.3a46-8abd	0.28	0.82	0.42	0.28	0.40			0.59	0.64	0.64	Hake in subareas 4, 6, and 7, and divisions 3.a, 8.a–b, and 8.d, . . .

Continued

Table 1. continued

#	Column identifier / Stock name—short	a ICES 2018	b Froese *et al.* SPM	c RAM Legacy Database, Schaefer	d RAM Legacy Database, Thorson "Taxonomic"	e RAM Legacy Database, Thorson "general"	f Ecosystem model	g Dynamic pool models, e.g. PROST	h Average of b, average (c–e), f and g	i GLM of h, based on life history parameters	j Final recommended F_{msy} values—column i unless there are ecosystem or dynamic pool estimates and then a mean of column h and i	Full stock name (truncated to save space)
23	her.27.5a	0.22	0.23	0.25	0.29	0.26			0.25	0.28	0.28	Herring in division 5.a, summer-spawning herring (Iceland grounds)
24	her.27.nirs	0.27	0.43	0.57	0.66	0.58			0.52	0.32	0.32	Herring in division 7.a North of 52°30′N (Irish Sea)
25	her.27.irls	0.26	0.34	0.30	0.41	0.32			0.34	0.40	0.40	Herring in divisions 7.a South of 52°30′N, 7.g–h, and 7.j–k (Irish Sea, . . .
26	her.27.3a47d	0.26	0.58	0.23	0.29	0.24	0.50		0.45	0.32	0.38	Herring in subarea 4 and divisions 3.a and 7.d, autumn spawners . . .
27	her.27.1-24a514a	0.157		0.16	0.13	0.16			0.15	0.23	0.23	Herring in subareas 1, 2, and 5 and divisions 4.a and 14.a, Norwegian . . .
28	her.27.28	0.32	0.34	0.53	0.52	0.53			0.43	0.31	0.31	Herring in subdivision 28.1 (Gulf of Riga)
29	her.27.20-24	0.31	0.33	0.29		0.27			0.30	0.30	0.30	Herring in subdivisions 20–24, spring spawners (Skagerrak, . . .
30	her.27.25-2932	0.22	0.21	0.18	0.15	0.18	0.35		0.24	0.25	0.25	Herring in subdivisions 25–29 and 32, excluding the Gulf of Riga . . .
31	her.27.3031	0.21		0.19	0.17	0.19			0.19	0.30	0.30	Herring in subdivisions 30 and 31 (Gulf of Bothnia)
32	lin.27.5a	0.286	0.34	0.43					0.39	0.32	0.32	Ling in division 5.a (Iceland grounds)
33	mac.27.nea	0.21	0.36	0.37	0.39	0.37		0.40	0.38	0.39	0.39	Mackerel in subareas 1–8 and 14 and division 9.a (the Northeast . . .
34	meg.27.7b-k8abd	0.191	0.37	0.35	0.34	0.35			0.36	0.33	0.33	Megrim in divisions 7.b–k, 8.a–b, and 8.d (west and southwest of . . .
35	meg.27.8c9a	0.191	0.15	0.18					0.17	0.34	0.34	Megrim in divisions 8.c and 9.a (Cantabrian Sea and Atlantic . . .
36	ple.27.7a	0.169	0.21	0.42	0.57	0.45			0.35	0.29	0.29	Plaice in division 7.a (Irish Sea)
37	ple.27.7d	0.25	0.27						0.27	0.29	0.29	Plaice in division 7.d (eastern English Channel)
38	ple.27.420	0.21	0.47	0.36	0.30	0.35			0.40	0.35	0.35	Plaice in subarea 4 (North Sea) and subdivision 20 (Skagerrak)
39	ple.27.21-23	0.37	0.55						0.55	0.28	0.28	Plaice in subdivisions 21–23 (Kattegat, Belt Seas, and the Sound)
40	pok.27.5a		0.31	0.19		0.17			0.25	0.31	0.31	Saithe in division 5.a (Iceland grounds)
41	pok.27.5b	0.30	0.37	0.34	0.25	0.32			0.34	0.34	0.34	Saithe in division 5.b (Faroes grounds)
42	pok.27.1-2		0.49	0.32	0.30	0.32			0.40	0.32	0.32	Saithe in subareas 1 and 2 (Northeast Arctic)
43	pok.27.3a46	0.36	0.54				0.33		0.44	0.33	0.38	

Continued

Table 1. continued

#	Column identifier / Stock name—short	a ICES 2018	b Froese *et al.* SPM	c RAM Legacy Database, Schaefer	d RAM Legacy Database, Thorson "Taxonomic"	e RAM Legacy Database, Thorson "general"	f Ecosystem model	g Dynamic pool models, e.g. PROST	h Average of b, average (c–e), f and g	i GLM of h, based on life history parameters	j Final recommended F_{msy} values—column i unless there are ecosystem or dynamic pool estimates and then a mean of column h and i	Full stock name (truncated to save space)
												Saithe in subareas 4 and 6 and division 3.a (North Sea, Rockall and . . .
44	sol.27.7a 1.2	0.20	0.18	0.27	0.17	0.26			0.21	0.36	0.36	Sole in division 7.a (Irish Sea)
45	sol.27.7d	0.256	0.48	0.63		0.68			0.57	0.34	0.34	Sole in division 7.d (eastern English Channel)
46	sol.27.7e	0.29	0.26	0.21		0.20			0.23	0.33	0.33	Sole in division 7.e (western English Channel)
47	sol.27.7fg	0.27	0.31	0.44	0.60	0.47			0.41	0.31	0.31	Sole in divisions 7.f and 7.g (Bristol Channel, Celtic Sea)
48	sol.27.8ab	0.33	0.43	0.38	0.27	0.36			0.39	0.32	0.32	Sole in divisions 8.a–b (northern and central Bay of Biscay)
49	sol.27.4	0.20	0.38	0.40	0.38	0.40			0.39	0.32	0.32	Sole in subarea 4 (North Sea)
50	sol.27.20-24	0.23	0.38	0.28	0.22	0.27			0.32	0.32	0.32	Sole in subdivisions 20–24 (Skagerrak and Kattegat, western Baltic . . .
51	spr.27.22-32	0.26	0.42	0.30	0.34	0.31	0.40	0.45	0.40	0.38	0.39	Sprat in subdivisions 22–32 (Baltic Sea)
52	mon.27.78abd	0.28	0.41						0.41	0.30	0.30	White anglerfish in subarea 7 and divisions 8.a–b and 8.d (Celtic . . .
52	mon.27.8c9a	0.24	0.63	0.27	0.21	0.26			0.44	0.30	0.30	White anglerfish in divisions 8.c and 9.a (Cantabrian Sea and . . .

Stock names from ICES Stock Assessment Database (19 November 2019), http://standardgraphs.ices.dk.

Table 2. Criteria for filtering out stocks in the surplus production modelling in the RAM Legacy Stock Assessment Database approach.

Stocks were filtered out if any of the following criteria was met
• More negative than positive surplus productions (SP) in the two middle quartiles of stock biomass (B) between 0 and carrying capacity k—*and*—the sum of SPs in the middle quartiles of B is smaller than 0
• ER_{MSY} estimated to be <0.005
• ER_{MSY} estimated to be >0.9
• B_{MSY} estimated to be $<0.05 \times B_{max\ observed}$
• B_{MSY} estimated to be $>2 \times B_{max\ observed}$
• Time-series <25 years
• If B_{MSY} was provided in the assessment, and estimated SP at B_{MSY} was negative
• The surplus production model fit is worse (higher Akaike Information Criterion with a correction term for small sample sizes (AICc) value) than those of three possible linear fits, $SP = m \times B + b$, $SP = m \times B$, and $SP = b$, where m and b are parameters estimated in the fits

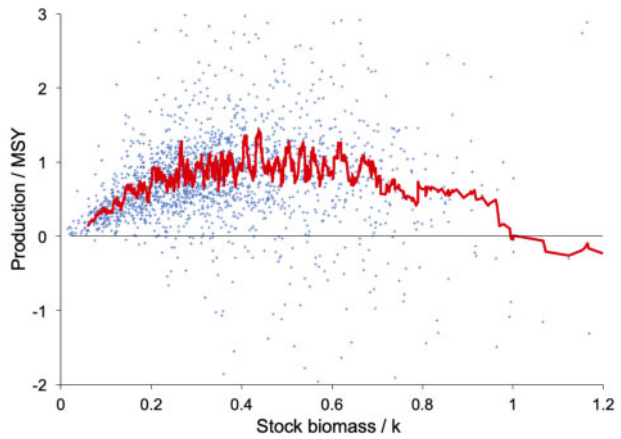

Figure 1. Stock production vs. stock biomass, normalized to MSY and k (carrying capacity), respectively, for 53 data-rich stocks, excluding five stocks filtered out in the RAM Legacy Stock Assessment Data Base analysis. For clarity, 34 out of 1901 data-pairs were not included because they were outside the intervals on the y-axis but were quite evenly spread around the general pattern. The red line is a running mean of 25 points. The "general Thorson *et al.* (2012)" model ($\varphi = 1.736$) was used to get MSY and k by stock.

estimates in the ICES F-"currency" for individual years should ideally be equal for a given fish stock, but they often differ slightly between years due to different basic model structures between ICES models and the SPMs. We use the mean of the estimates of F_{msy} for 2000–2012 as the final F_{msy} estimate from this method.

Regime shifts are a challenge for SPMs, which depend on long time-series with a good dynamic range to estimate F/F_{msy} with reasonable precision. Thus, cutting the existing time-series to reflect a regime shift will make the SPMs less useful. We did a sensitivity analysis using the Faroese stocks of cod, haddock (*Melanogrammus aeglefinus*), and saithe (*Pollachius virens*) to investigate regime shifts and their influence on the SPM estimates of F_{msy}. A 39-year time-window was moved in steps of 10 years, and we performed the above SPM method to estimate F_{msy} for each time-window (for more details, see the Supplementary material).

Froese *et al.* (2016) estimation of F/F_{msy} in combination with ICES F time-series

Froese *et al.* (2016) used the catch and SSB time-series (corrected to exploitable biomass by a so-called catchability factor estimated as part of the modelling process) from ICES assessments and a Schaefer SPM:

$$B_{t+1} = B_t + r\left(1 - \frac{B_t}{k}\right)B_t - C_t,$$

where B_{t+1} is the exploited biomass in the subsequent year $t+1$, B_t is the current biomass, r is the maximum intrinsic rate of population increase, k is carrying capacity, and C_t is the catch in year t. To account for depensation or just the reduced recruitment at severely depleted stock sizes, Froese *et al.* (2016) used a linear decline of surplus production if biomass fell below $_1/_4k$:

$$B_{t+1} = B_t + 4\frac{B_t}{k}r\left(1 - \frac{B_t}{k}\right)B_t - C_t.$$

The term $4B_t/k$ assumes a linear decline of recruitment below half of the biomass that on average produce MSY, as B_{msy} is equal to $_1/_2k$ in the Schaefer type SPMs.

The outcome of this analysis was the ratio F/F_{msy} for each stock and year in the time-series, and this ratio was linked to ICES time-series of F to obtain F_{msy} in the ICES F-"currency", as described above. We used the mean of the estimates of F_{msy} for 2000–2012 as the final F_{msy} estimate from this method. The *C.V.* of this mean value was, on average, quite small (0.06) when accounting for the variation in estimated F_{msy} by year.

Literature F_{msy} estimates from multispecies and ecosystem models

We extracted F_{msy} estimates from peer-reviewed publications of well-established multispecies and ecosystem modelling. We focused on models that multiple scientists have worked on for several years and for which the results have stood the test of time. From these models, we selected the analysis where the balance, in terms of stock biomass composition across species, has been about where it is at present and where fishing pressure has varied up or down simultaneously across stocks. This was done to mimic the current management approach with HCR that keeps all stocks at healthy stock sizes (defined as capable of producing unimpaired recruitment, ICES, 2018). We did not consider the Barents Sea ecosystem, because the main part of the multispecies interaction is already included in the way the current F_{msy} values are calculated for the fish stocks in this area (ICES, 2018). The references to which publications are used can be found in the Supplementary material for individual stocks.

Dynamic pool models

Dynamic pool models track the numbers in each age group separately, rather than lumped together as aggregate biomass. They are typically based on the results of Statistical Catch at Age (SCAA) models (Shepherd and Pope, 2002). Dynamic pool models can account for variable growth, sexual maturation, natural

mortality, and recruitment due to density dependence. This is the approach most often used by ICES, except that density dependence in growth, maturation, and mortality usually are missing. Here, we include density dependence in these factors.

Stochastic projections of an age-structured, dynamic pool population model were done using the Java software PROST (Åsnes, 2005). PROST has been used by ICES for the Northeast Arctic cod stock (Kovalev and Bogstad, 2005) to obtain the currently used F_{msy} values in the annual assessment and advice to management. PROST, or a similar approach, can be used for any stock to make single-species, single-fleet, single-area projections, incorporating density dependence in recruitment, growth, mortality, and maturity. This method was used in the present study for North Sea cod and Northeast Atlantic mackerel (*Scomber scombrus*). Input data are provided in the Supplementary material.

The model NE_PROST from ICES (2017a) has the same basic functionalities as PROST but is based on Excel and Visual Basic. This model was used for Northeast Arctic cod and cod at Icelandic grounds in the current study. Input data are from ICES (2017a) for Northeast Arctic cod. For cod at Icelandic grounds, input data are from ICES (2017b), and (i) B_{lim} was set to 207 000 tonnes based on a segmented regression analysis ("Hockey stick" model), (ii) B_{pa} was set to 330 000 tonnes, (iii) DD growth based on Danielsson *et al.* (1997), and (iv) cannibalism was set as for the Barents Sea cod 1970–1985 based on Bogstad *et al.* (1994) and ICES (2015, 2017a). Links to input data are provided in the Supplementary material.

Horbowy and Luzeńczyk (2017) used a tailor-made code in VisualBasic to obtain dynamic pool model estimates of F_{msy} with density-dependent growth and predation mortality for Baltic sprat (*Sprattus sprattus*). Cod is by far the most important predator on sprat in the Baltic Sea, and the cod stock biomass was assumed constant at each model run. Runs with cod stock biomass of 100, 200, and up to 600 kt were done. The predation mortality is then only dependent on the biomass of sprat for each level of cod biomass, and there is a minor negative relationship between predation mortality (not cannibalism here, but cod predation on sprat) and sprat biomass, when the cod biomass is kept constant. We used a cod biomass value of 200 kt to represent the present stock situation. The analysis was only sensitive to higher cod biomass. If the cod stock is rebuilding sometime into the future, this choice of cod biomass will need to be revised. The cod stock biomass has previously been over 600 kt.

ICES default HCRs have been applied in all of the above dynamic pool model runs, with the ICES biomass trigger points (MSY Btrigger) at the values from ICES (2018) or if these were missing, at B_{pa}, which also can be found in ICES (2018).

F_{msy} and life history parameters

F_{msy} has often been linked to life history parameters such as natural mortality and growth rate. We used general linear models (GLM) coded in R, for this purpose. We tested a set of relevant life history parameters (age-at-50% maturity—"a50mat", natural mortality of mature fish—"natM", $L_\infty \times K$ from the von Bertalanffy growth models—"Linf_K", preferred temperature—"prefT", trophic level of adult fish—"troph") against the F_{msy} values obtained from the methods mentioned above. The parameter values were based on ICES current input data to fish stocks assessments (ICES, 2018 and reference therein) supplemented with data from FishBase (Froese and Pauly, 2018). We tested a few relevant groupings of species and found that a five-category grouping of species "taxg3" [cod and hake (*Merluccius merluccius*), other gadoids, flatfish, herring (*Clupea harengus*), and sprat, and others] worked well with the model. Only a few parameters can be included in the model as we only have 53 F_{msy} "observations". We tested several relevant GLM (see Supplementary material for detailed information). Across most of the models, we found (i) a positive influence on F_{msy} of "natM" and, to a lesser degree, of "Linf_K"; (ii) a negative influence on F_{msy} of "a50mat" and, to a lesser degree, of "prefT"; and (iii) "troph" was correlated with both "a50mat" and "Linf_K" and did not add much to the model when both of these were included. "Linf_K" was preferred to "natM" because it is easier to estimate with good precision for most stocks. The final GLM was:

$$\log(F_{msy}) = \log(\text{a50mat}) + \log(\text{Linf}_K) + \text{taxg3}.$$

It was assumed that F_{msy} is log-normally distributed. The above GLM were fitted to F_{msy} estimates, one datapoint for each stock obtained as the mean by stock from the SPMs, ecosystem, multispecies, and dynamic pool models (column "i" in Table 1). We used the predicted values of F_{msy} from this GLM modelling (column "j" in Table 1) as the final set of best estimates of F_{msy} to use in management of the individual fish stocks. However, for those nine stocks where ecosystem, multispecies, or dynamic pool models were also available, we used a mean of column "h" and "i" to put more weight on the non-GLM estimates of F_{msy} for these stocks, due to the availability of extra information from the ecosystem, multispecies, or dynamic pool models. Weighting is a difficult task, and we used our expert knowledge to make the choice of weighting. This is a standard approach when valid *C.V.* estimates are not available. In many situations, the choice of weighting could influence the results, and other weighting schemes could give different results. Fortunately, in this case, all our model estimates of F_{msy} for a given stock agree quite well, and, therefore, the results are not sensitive to the choice of weighting.

The "Great Experiment"

According to Sparholt and Cook (2010), the slow gradual increase in fishing mortality for the 28 most important and data-rich stocks in the Northeast Atlantic (Figure 2) from well below to well above F_{msy}, combined with corresponding yield in the period 1950–2005 offered, unintentionally, a natural experiment, here called the "Great Experiment", to estimate an aggregate F_{msy}. All 28 stocks are also included in the present analysis.

Sparholt and Cook (2010) found an aggregate F_{msy} of $0.46\,year^{-1}$. The present analysis gives an average F_{msy} of $0.43\,year^{-1}$ for the same set of stocks. We did not make any attempt to correct the F_{msy} values obtained from the methods described above because they are so similar.

The current F_{msy} values used by ICES are typically a mean F over some age groups. These age groups differ by stock. We investigated whether there was a need for standardizing F_{msy} values to make them comparable between stocks. One of the most extreme differences is between the F for North Sea cod (ages 2–4) and F for cod at Icelandic grounds (ages 5–10). We tested the need for standardization by converting the values for each stock to the F metric (1 − Spawner Per Recruit (SPR)) as explained below because this is a metric that relates directly to the impact on the

stock. For a given fishing pattern, SPR(F) is the ratio of SSB per recruit, when fishing at an intensity of F, divided by the SSB per recruit with no fishing (Goodyear, 1993; Cordue, 2012). The final metric is calculated as 1 minus SPR(F) [called (1 − SPR)] because it then is an increasing function of fishing intensity. Thus, (1 − SPR) is "1 − ratio of (SSB/R at F) to (SSB/R at $F=0$)".

Results

The GLM based on life history parameters explained 59% of the variation in the F_{msy} values. A model without the "taxg3" factor was almost as good, explaining 46% of the variation, while requiring only two parameters (see Supplementary material). However, the AICc was higher (50.9 vs. 45.8) than for the model including "taxg3". Linf_K was not significant at the 5% level, but leaving it out gave higher AICc scores (47.0), and the above-mentioned two-parameter model gave highly significant effects of Linf_K, indicating that it was an influential parameter. Diagnostics from the run are found in Table 3. Plots of model-predicted estimates of F_{msy} vs. "observed" F_{msy} and residuals vs. "observed" F_{msy} are presented in Figure 3.

F_{msy} estimates from all individual approaches are given in Table 1. For each stock, up to four of these approaches with available estimates are averaged and then fed into the GLM described above. Predicted values from the GLM represent an ensemble, mean F_{msy} estimate. Across all 53 stocks, the mean value of the "ensemble F_{msy}" estimates is 0.38 year^{-1}. For the 48 stocks for which there are F_{msy} estimates by ICES (2018), the average F_{msy} is also 0.38 year^{-1}. This is nearly 50% greater than the mean of the ICES estimates of 0.26 year^{-1} (a two-sided, pair-wise t-test was highly significant, with a p-value of <0.001). There is, however, considerable variation between stocks: for five stocks, the

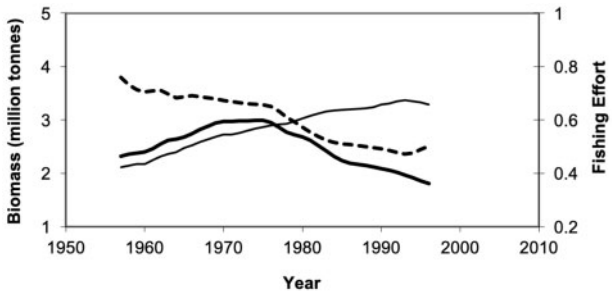

Figure 2. The "Great Experiment". Catch (thick line) and mean F (thin line) for 28 data-rich groundfish stocks in the Northeast Atlantic by year. F gradually increased over the time considered, and the catch followed the increasing path to start with, but around the mid-1970s took a decreasing path, indicating that the F_{msy} point had been surpassed. Stock biomass (spawning) is also shown (punctuated line). Biomass of both catch and stock refers to the left y-axis. From Sparholt and Cook (2010), where F (mean F by year across stocks) is called "fishing effort", which means it is not effort measured directly in terms of fishing days, for example but measured indirectly in terms of fishing mortality.

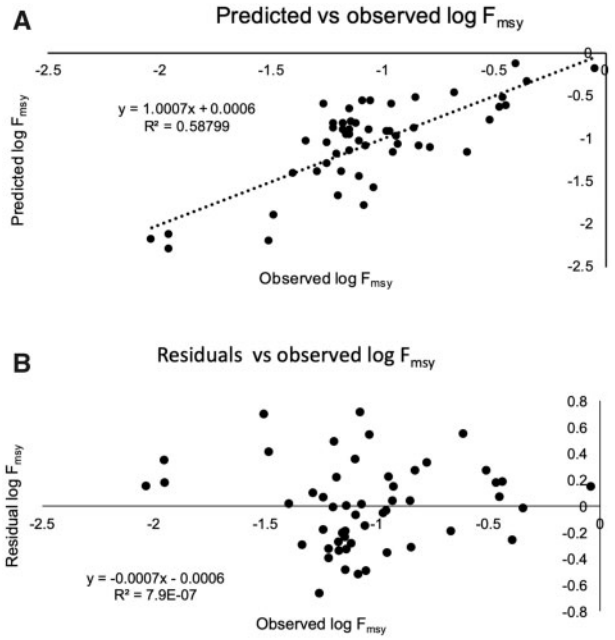

Figure 3. Model-predicted log(F_{msy}) vs. "observed" log(F_{msy}) from a GLM: log(F_{msy}) = log(a50mat) + log(Linf_K) + taxg3, used to link life history parameters to F_{msy} (a), and residual vs. "observed" log(F_{msy}) values (b).

Table 3. Diagnostics of the GLM log(F_{msy}) = log(a50mat) + log(Linf_K) + taxg3, used to link life history parameters to F_{msy}.

Variable name	Coefficient	Standard error	t-Value	p-Value
Intercept	−0.3807	0.3881	−0.981	0.3318
taxg3 (flatfish)	−0.6295	0.1906	−3.302	0.0019**
taxg3 (forage fish)	−0.7003	0.1880	−3.724	0.0005***
taxg3 (other gadoids)	−0.3984	0.1513	−2.634	0.0115*
taxg3 (other taxonomic groups)	−0.5154	0.2258	−2.258	0.0271*
Linf_K	0.2091	0.1145	1.826	0.0744
a50mat	−0.5800	0.1125	−5.156	0.0000***
Null deviance		12.7648 on 52 degrees of freedom		
Residual deviance		5.2618 on 46 degrees of freedom		
Akaike Information Criterion		43.987		
AICc		45.813		

Significance codes:
*<0.05,
**<0.01, and
***<0.001.

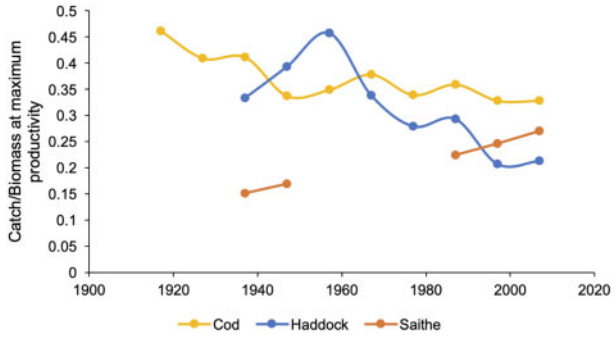

Figure 4. Estimates of catch divided by exploitable biomass at maximum productivity (B_{msy}) of cod, haddock, and saithe in Faroese waters (i.e. F_{msy}). Based on SPMs applied for intervals of 39 years that were moved in steps of 10 years represented by the dots. Smoother curves between the dots were applied.

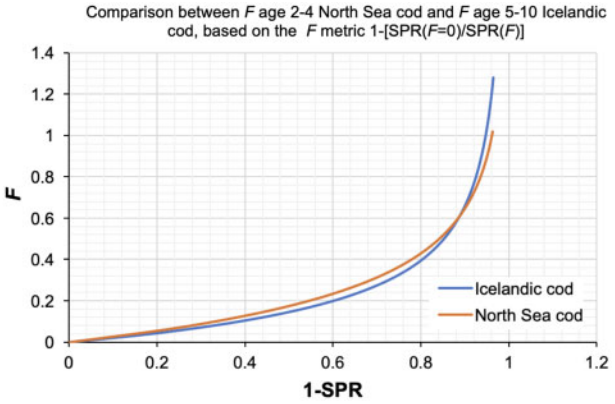

Figure 5. Comparison of the relationship between fishing mortality in ICES *F*-"currency" and the *F* metric (1 − SPR) for North Sea cod and cod at Iceland. In spite of great differences between the two cod stocks in terms of exploitation pattern, the impact on the stock for a given numerical ICES *F*-"currency" is quite similar.

ensemble F_{msy} estimates are lower than the ICES F_{msy} values; while for 19, 17, and 7 stocks, the ensemble F_{msy} estimates are 1–1.49 times, 1.5–1.99 times, and >2 times greater than ICES F_{msy} estimates, respectively. As expected, there is a tendency for the new ensemble F_{msy} values to be much higher than the current F_{msy} values for cannibalistic species like hake and cod.

Sensitivity tests of the SPM vs. regime shifts for three Faeroese stocks showed that F_{msy} (expressed as catch/biomass at maximum productivity) in the past century has been relatively stable, especially for cod (Figure 4). For haddock and saithe, it has fluctuated by a factor of about 2; however, much less for each half-century.

The fishing mortality F for North Sea cod is the mean over ages 2–4, while for cod at Icelandic grounds, it is over ages 5–10. In spite of the great difference between the two cod stocks in terms of exploitation pattern, the impact on the stock for a given numerical ICES *F*-"currency" is quite similar (Figure 5). Hence, there is no apparent need to correct F values to make them further comparable; this verification is important for our GLM analysis where we linked life history parameters to F_{msy} values.

Discussion

A simple and pragmatic approach is proposed to bridge a gap between the enormous amount of science available on ecosystem functioning from the past four decades, and scientific advice/management of fish stocks in the Northeast Atlantic. This gap has hitherto proven difficult to bridge and has resulted in many reorganizations of the science pillar of ICES work over the past two decades in attempts to facilitate the bridge. A central point in our approach has been to link the results of the well-established surplus production modelling framework to the time-series of fishing mortality from the age-based annual assessments. SPMs give the ratio of F/F_{msy} and the assessments give F in the same time interval. Dividing F from the assessments with the ratio F/F_{msy} from SPMs gives F_{msy}. This innovative linking approach has, to our knowledge, never been done before.

Translating SPM F/F_{msy} to number and cohort-based F_{msy}

The approach used is to obtain F/F_{msy} from SPM and translate them to number and cohort-based F_{msy} arising from each individual stock assessment, first so that they are on the same scale and comparable, and second so they can be used in the current advice process (with a focus on ICES).

The ratio of F/F_{msy} from SPM models (based on biomasses) is almost linearly related to F/F_{msy} based on numbers and cohorts, when F is close to or lower than F_{msy}. When F is much higher than F_{msy}, the ratio of F/F_{msy} from SPMs is lower than the ratio of F/F_{msy} from the number-based models. This is due to the exponential decay submodels used in the number-based models. Therefore, the new F_{msy} values could be underestimates in those cases where F, for several years in 2000–2012, has been much higher than F_{msy}. Correcting for this only meant a small change in F_{msy} (less than a few percentages even for the stocks with the highest F well above F_{msy}) and, therefore, this addition to the complexity of the calculations of the new F_{msy} values was omitted.

The new proposed F_{msy} values are, on average, less than half the maximum F values experienced historically by the stocks according to the current assessments (ICES, 2019). Only for beaked redfish (*Sebastes mentella*) in areas 1 and 2 (reb.27.1–2) it is higher (F_{msy} = 0.13 year^{-1}, maximum F = 0.05 year^{-1}). However, for this stock, the time-series in ICES (2019) is short, going back only to 1992. Before that, Virtual Population Analyses done by ICES (1985) indicate that F previously was much higher. It can, therefore, be concluded that all the 53 stocks dealt with in the present study have experienced higher F than the new proposed F_{msy} values, and, on average, twice as high.

F_{msy} for cannibalistic stocks

Generally, we find that cannibalistic stocks have the highest F_{msy} values. Hake is most cannibalistic, with juvenile hake constituting about 10% of the diet (Mahe *et al.*, 2007), cod second with about 5% (ICES, 1997; Link *et al.*, 2008), and haddock (ICES, 1997) and blue whiting (*Micromesistius poutassou*) (Dolgov and Prokopchuk, 2013) third with 1–2%. The new F_{msy} values are reflecting this, as F_{msy} is highest for hake, intermediate for cod, and lower for haddock and blue whiting. One implication of this result is that ignoring cannibalism when estimating F_{msy} can be a serious error for stocks where this is a significant driver of stock dynamics (Collie and Gislason, 2001).

Regime shifts

The SPM used time-series of catch and biomass to estimate F/F_{msy} by year. The basic assumption is that the productivity of a stock is constant over the time considered. Ecosystem regime shifts could, therefore, result in unreliable estimates of F_{msy}. To address the potential issue of regime shifts leading to unreliable SPM F_{msy} estimates, we performed a sensitivity analysis using the Faroese stocks of cod, haddock, and saithe. This analysis showed that, on a century scale, variations in productivity changed F_{msy} estimates from SPMs. However, it also showed that this effect was less pronounced on a half-century basis. The Faroe marine ecosystem is known for its large variation in productivity with time, so this is a comforting result.

Analysis presented in the Supplementary material and references therein showed that long (at least 25 years) time-series are needed to estimate F_{msy} with a useful precision for management. This observation supports the notion that it is most appropriate for the time being to only consider obvious regime shifts. A regime shift is obvious for cod in the Eastern Baltic (SD 25–32). A liver parasite pandemic started around 2010 and coincides with a much-reduced growth and condition and almost no survival to large sizes (ICES, 2018). Therefore, we discarded this stock in our set of 53 stocks.

Another case is sprat in the Baltic. If the Eastern Baltic cod stock (SD 25–32) recovers to former stock sizes, it will mean a high predation mortality on sprat and thus a reduced F_{msy} for sprat. Therefore, we suggest that our F_{msy} estimates should be revised every 5 years or so, and probably as part of the periodic reviews of the stock assessment methodology.

In cases where the reason for a regime shift is less obvious, it may be better to use the GLM-type approach mentioned above for estimating F_{msy}, rather than trying to estimate F_{msy} based on a surplus production model or dynamic pool model on too short time-series.

In general, strong evidence of regime shifts within the model-tuning series of a given stock could indicate that this methodology may not be appropriate for that stock and, at the least, would require more investigation.

Independent estimates of F_{msy}

Although we focus on SPM and the F/F_{msy} ratio translation to obtain F_{msy} estimates, we also present several other methods. The fact that our F_{msy} estimates are quite similar across methods for those stocks where several methods could be applied gives greater confidence in the main results presented here. The SPM estimates of F_{msy} were in line with the multispecies F_{msy} estimates for the stocks where these were available (multispecies 0.51 $year^{-1}$ vs. SPMs 0.46 $year^{-1}$). Furthermore, for the stocks where the dynamic pool F_{msy} estimates were available, these were also in line with our SPM values (dynamic pool 0.57 $year^{-1}$ vs. SPMs 0.49 $year^{-1}$). The aggregate F_{msy} (0.46 $year^{-1}$) of the 28 data-rich stocks in the "Great Experiment" analysis (Sparholt and Cook, 2010) is consistent with the mean of the individual F_{msy} values (0.43 $year^{-1}$) for these 28 stocks from the present analysis.

Density dependence

Density dependence (DD) in fish population dynamics was included from the beginning of this field of science (Baranov, 1918). ICES held a symposium in 1947 to consider how important DD was when fish stocks were left practically unfished during World War II (Graham, 1948). The seminal book by Beverton and Holt (1957) includes many concrete case studies with effects of DD on fish population dynamics. There seems to be a renewed attention in the marine science community on DD effects on growth, maturity, and cannibalism in recent years, probably because of the ending of overfishing in many areas. DD is important because it influences the estimations of biological reference points as shown above and by many others (see Lorenzen, 2016 for an overview). Since the 1950s, when fishing increased steadily year by year and overfishing became a general phenomenon, DD was not very important for management, because it was clear early on that it was not strong enough to counteract the increased fishing (see Graham, 1948). Brook and Bradshaw (2006) did a meta-analysis of 1198 species, including many fish and concluded: "…we show that density dependence is a pervasive feature of population dynamics … and that this holds across widely different taxa". Henderson and Magurran (2014) used a 33-year monthly time-series following the dynamics of 81 species of fish in the Bristol Channel. They found density dependence common and mostly so in species most abundant in the ecosystem. That DD is most pronounced in stocks most abundant in the ecosystem seems sensible to expect for ecosystems in general and might be relevant for the fish stocks analysed in the present study as these generally are dominant species in their respective ecosystems.

Zimmermann et al. (2018) analysed 70 stocks from ICES routine assessment work and concluded that: "The results reaffirm the importance of density-dependent recruitment in marine fishes, yet they also show that density dependence in somatic growth is not uncommon". Furthermore, as they state: "…this comparison is only approximate because it ignores the effects of growth on survival and maturation" and adding these effects to that of growth might bring the total effect above that of DD in recruitment. This result fits with Morgan *et al.* (2016) who found, by looking at R/SSB and SSB/R in a meta-analysis of Northwest Atlantic stocks, that the effect from the DD factors other than DD in recruitment contributed about as much as that from DD in recruitment to total DD in the population dynamics of the stocks. Zimmermann *et al.* (2018) give a very good overall description of the state of the art in DD research and they state: "Collection of population abundance data for commercial fish has almost always started after fishing had already depressed population abundance, and hence, high population sizes near carrying capacity are underrepresented or absent. Consequently, our parameter estimates might be biased." It is important to have a large dynamic range in stock biomass over time to be able to detect DD from data noise. Generally, the models hitherto used to estimate F_{msy} are based only on DD in recruitment, and they often result in estimated stock size at low F that are an order of magnitude larger than the largest ever observed stock size (ICES, 2008). ICES has generally accepted this problem with the models, and, therefore, ICES does not provide B_{msy} values for any stock, but only F_{msy} values (ICES, 2013). However, there are examples of stocks in other parts of the world than the Northeast Atlantic where data exist from before a stock was exploited, and it would be a good test of DD to analyse such data with the approach presented here.

Size spectrum-based ecosystem models suggest that DD is likely to be most important at the individual fish size where the cohort biomass (stock number times individual weight) is at its maximum because this is the point where the cohort makes up the largest part of the biomass in the ecosystem biomass spectrum (Andersen, 2017). This usually happens when the cohort starts to

mature (Charnov *et al.*, 2013) and it fits with the notion that DD in growth of post-recruits is an important element of the population dynamics of a fish stock.

Linking F_{msy} to life history parameters

The GLM approach to link F_{msy} values with life history parameters showed relatively strong explanatory power, explaining 59% of the variation in log(F_{msy}) among stocks. This relationship was used in the present study to reduce the influence of random data noise on the final F_{msy} values. However, as a spin-off of the present study, it can also be used on data-limited stocks not included in the present study to get F_{msy} if just some simple life history parameters are known. This regression approach could also be used to generate a prior for F_{msy} estimation for data-rich stocks. The parameter coefficients presented in Table 3 can be applied in such calculations.

Implications for management

The new proposed F_{msy} values are based on an approach that removes some of the known biases in the current methodology, and they have their basis in the science available on ecosystem functioning, ecosystem models, and multispecies models. They, therefore, live up to the core values of science underpinning fisheries management as stated, for example in the ICES Copenhagen Declaration (ICES, 2014): "… to give unbiased, sound, reliable, and credible scientific advice on human activities affecting, and affected by, marine ecosystems …" and" … advice in relation to fisheries management, giving full consideration to the ecosystem context".

The new F_{msy} values have been scaled to the current assessment models so that they could be used in connection with the forecast table presented in the annual advice sheets produced by ICES each year. For each stock, the total allowable catch (TAC) advice can be set at the catch corresponding to the fishing pressure equal to the new F_{msy}. In cases where SSB is below $MSYB_{trigger}$ (ICES, 2018), the TAC should be set so that F is reduced according to the ICES default HCR. The current biomass reference points B_{lim} and B_{pa} (ICES, 2018) can be maintained. These are based on historic stock–recruitment relationships from age-based models, where annual variables weight-at-age, maturity-at-age, and often also natural mortality-at-age are included, based on observations, and, therefore, DD is accounted for. $MSYB_{trigger}$ is often set based on B_{pa} and, therefore, can be maintained as well, but there is an issue if they are based on management strategy evaluations. These typically do not account for DD in growth, maturity, and natural mortality and thus suffer the same bias as is the central topic of the present study. One pragmatic solution could, in those cases, be to set $MSYB_{trigger}$ at B_{pa} and reduce F further if SSB surviving the TAC year has a risk of >5% for being below B_{lim}. Obviously, a better approach would be to directly include the DD process in the Operating Model using the management strategy evaluations (as was done in the Northeast Arctic cod example quoted above). Note here that we focus on the SSB surviving the TAC year instead of the current practice, where it is SSB at the start of the TAC year (ICES, 2018). We think that our focus is logical because the surviving SSB depends on which TAC is set, and the SSB at the beginning of the TAC year does not.

However, no single modelling approach is perfect, and it is right to acknowledge the limitations here. Although the translation between F/F_{msy} to the assessment model avoids the issue of transferring absolute values between models, it is not perfect. There is an underlying assumption that the trends are similar between the two models. Where the trends in the SPM and the assessment are markedly different, we would, therefore, advise caution in making the adjustment described here. A second limitation concerns the nature of recruitment modelling in SPMs. These models are poorly suited to stocks, which undergo periods of prolonged recruitment failure. It is prudent when applying the new ensemble estimates of F_{msy} in the scientific advice for a given stock that they are very carefully validated against the current F_{msy} value taken into account that the current estimate is likely underestimated because some DD factors are ignored. In this validation, it is also important to consider the new F_{msy} estimates of similar stocks; for example for the sole (*Solea solea*) stocks, which have almost identical growth and age-at-50% maturity, one would not expect large differences in F_{msy} values. Although the average yield may be highest at the quoted F_{msy} values, fishing at this level could give rise to unacceptably high chances of stock collapse during periods of poor recruitment. It would also make sense to concentrate on stocks, which show evidence of density dependence at current stock sizes. Finally, from purely pragmatic considerations, it would not seem sensible to increase fishing pressure on any stock close to or below biomass levels where recruitment could be impaired. This is not specific to the modelling conducted here; increasing fishing mortality on a stock just above its recruitment overfishing threshold is unlikely to be wise (or precautionary) regardless of the modelling performed.

We, therefore, think that our new F_{msy} values are especially relevant for the management of fisheries in periods of high stock sizes. From a harvesting point of view, if a stock is above the carrying capacity, its surplus production is negative, and it makes sense to reduce the stock size. When a stock is depleted, neither our new F_{msy} values nor the current ones should be used, but something much lower or even no fishing at all. The current ICES default HCR takes account of this to some extent, but a close consideration of this HCR is needed on a case-by-case basis. We would also not expect the values proposed here to be used for stocks only slightly above depleted levels and would urge caution for stocks that are known to experience prolonged periods of poor recruitment. One method to resolve these issues is the approach taken for Northeast Arctic cod, whereby two different target fishing levels are employed, with a higher level being permitted at high stock sizes (ICES, 2016). The Northeast Arctic cod management employs an approach whereby the target fishing mortality increases to a higher target F at $3B_{pa}$. In principle, the break points of the "double hockey stick" HCR employed should be evaluated with an Management Strategy Evaluation—analysis for each stock. This could be a default template that would allow the benefits of higher target Fs incorporating density dependence, as proposed here, while retaining the full precautionarity associated with the current F_{msy} targets.

We focused on a single-stock approach because we think this is most useful for current management until priorities between fisheries have been more clearly formulated by managers. However, the stocks live in an ecosystem and interact with each other. Some of this interaction is mimicked in our approach by DD mechanisms, estimated based on historical time-series. All the major commercially important fish stocks in the Northeast Atlantic have been exposed to about the same historical development in fishing pressure, driven by the same basic time-trend in fishing capacity of the fishing fleet. Therefore, we think that our

results are valid if our new F_{msy} values are implemented widely and for many stocks (except the ones at low stock levels). Due to reduced predation and food competition, each stock will benefit from the reduction in biomass of all the other stocks. Alternatively, if the new F_{msy} values are implemented on only one stock, this benefit will not fully materialize. Sparholt and Cook (2010) showed that the fishing intensity around 1960 gave MSY, and the fishing mortality was, at that time, similar to the mean of our proposed F_{msy} values. For the areas where ecosystem models are available, these give F_{msy} values consistent with our F_{msy} values. Although both the goals and methods of this study are different, the overall approach of attempting to incorporate increased ecosystem realism into the single-species F targets has similarities to the proposed methods in ICES (2020).

In summary, we have shown that excluding density dependence in growth, maturation, and natural mortality can give rise to biased estimates of F_{msy}. We propose two different methods for addressing this. Where the knowledge and resources exist, a full management strategy evaluation-style analysis can directly incorporate these processes into the HCR reference points. Where such an involved analysis is not possible, we propose a simpler method, using the F/F_{msy} ratio from surplus production models to produce an F_{msy} estimate compatible with the existing stock assessment model. We would anticipate that these revised estimates would be of greatest use for stocks currently at high stock level, where density dependence could be expected to be significant and where raising F target levels would not be unprecautionary. Given these caveats, we believe that the methods presented here would allow for increased yield and improve the underlying scientific basis for fisheries advice by avoiding the potential serious errors involved in ignoring density dependence. We also recommend that the new F_{msy} values are only used when HCR are in place, with biomass reference points defined, and where fishing pressure is reduced when a stock gets below reference points. Such HCR have worked well since they were introduced for stocks in the Northeast Atlantic about a decade ago and have effectively prevented stocks from collapse.

Supplementary data

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

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Author contributions

A complete list of contributions to the paper using the CRediT model is found in the Supplementary material. All co-authors have participated and contributed to the work during the 2-year project period and afterwards when further analysing data and very extensively discussing the results. Two open international scientific conferences were held in 2018 and 2019 just discussing the approach and results presented here with other scientists, managers, and stakeholders. Most of the co-authors participated.

Data and materials availability

See Supplementary material.

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