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Comparison of threat and exploitation status in North-East Atlantic marine populations

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Summary

1. Threat listing of exploited marine species has been controversial because of the scientific uncertainty of extinction risk as well as the social, economic and political costs of management procedures that may be triggered by designation of species as threatened.

2. We applied three sets of threat criteria to 76 stocks (populations) of 21 exploited marine fish and invertebrate species. Two criteria sets were based on decline rates: World Conservation Union (IUCN A1) and the American Fisheries Society (AFS). The third set of criteria, based on population viability (IUCN E), was assessed using non-parametric simulation and two diffusion approximation methods.

3. We compared extinction risk outcomes (threatened or not) against the exploitation status of each stock as reported in fish stock assessments (inside or outside safe biological limits). For each combination of threat and exploitation we assessed the rate of hits, misses and false alarms.

4. Our analyses suggest that decline rate criteria provide risk categorizations consistent with population viability analyses when applied to exploited marine stocks. Nearly a quarter of the fish and invertebrate populations ($n = 18$) considered met one or more of the threat criteria.

5. None of the threat metrics produced false alarms, where sustainably exploited stocks were categorized as threatened. The quantitative IUCN E metrics produced higher hit rates than the decline rate metrics (IUCN A1 and AFS) and all of the metrics produced similar miss rates. However, the IUCN E methods could be applied to fewer stocks (12–14) compared with IUCN A1 decline rate and AFS criteria, both of which could be applied all 76 stocks.

6. *Synthesis and applications*. Threat criteria provide warnings of population collapse that are consistent with those provided in fisheries stock assessments. Our results suggest that scientists with different backgrounds and objectives should usually be able to agree on the stocks for which the most urgent management action is needed. Moreover, IUCN A1 decline rate metrics may provide useful indicators of population status when the information needed for full fisheries stock assessment is not available.

Key-words: CITES, extinction risk, marine fisheries, minimum viable population, Red List

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Introduction

The understanding of extinction risk in marine organisms is poor, partly as a result of the controversy over whether the methods, originally developed for assessing threat in mammals and birds, can be applied to exploited marine species (Musick 1999; Hutchings 2001a). A number of commercially exploited fishes, including Atlantic cod *Gadus morhua* L., southern bluefin tuna *Thunnus maccoyii* (Castelnau 1872) and Atlantic halibut *Hippoglossus hippoglossus* (L.), were listed as threatened in 1996 under the World Conservation Union (IUCN) Red List criteria

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because their observed decline rates exceeded the qualifying rate over a time period equal to three generations (Mace & Hudson 1999). Concern was expressed that such listings did not reflect true risk of extinction, especially if the decline was a consequence of management action focusing on maximizing the sustainable yield of a fishery (Mace & Hudson 1999; Powles *et al*. 2000). Moreover, it was argued that while the IUCN criteria were effective at flagging rapid short-term population changes, they may overestimate extinction risk for many exploited marine species (Matsuda, Yahara & Uozumi 1997; Musick 1999). This weakens the credibility of scientific advice and may result in unnecessarily high short-term political, social and economic costs (Punt 2000). In practice, and notwithstanding high-level political commitment to conserving biodiversity and adopting a precautionary approach, (WSSD 2002), the potentially high costs often weigh against action to conserve biodiversity, particularly when there is any element of uncertainty in the advice (Hilborn 2004; Jennings 2004).

Three studies have explored the validity and utility of threat criteria in marine fishes. Analyses of declines suggest threat criteria thresholds adequately reflect the observed recovery potentials of fish stocks and are consistent with the precautionary approach to fisheries management (Hutchings 2000, 2001a,b). In contrast, two studies suggest the IUCN Red List decline criteria may raise false alarms when applied to commercially exploited fishes. The southern bluefin tuna was listed as critically endangered by the IUCN in 1996, based on an inferred population decline ≥ 80% within three generations (Matsuda *et al*. 1998). The probability of reaching 500 adults within 100 years was estimated using population viability methods, and found to be 50–96% (Matsuda *et al*. 1998). It was concluded that southern bluefin tuna was unlikely to become extinct within three to five generations and the 80% decline threshold was too conservative for abundant marine fishes. Punt (2000) compared IUCN Red List decline criteria and fisheries' reference points using age-structured population projections of six Australian fishes, and found a substantial probability (9–40%) of incorrectly classifying sustainably exploited species as threatened.

We compared threat status and exploitation status for 76 stocks of commercially exploited fishes and invertebrates using three sets of criteria: World Conservation Union (IUCN) population decline criteria A1 and population viability criteria E, and American Fisheries Society, Bethesda, Maryland (AFS) threat criteria. Threat outcomes were compared with an independent assessment of stock exploitation status provided by the International Council for the Exploration of the Sea (ICES). In northern Europe the Advisory Committee on Fisheries Management (ACFM) of ICES categorizes the exploitation status of each stock as 'inside' or 'outside' safe biological limits by comparing estimated spawning stock biomass and fishing mortality with reference points (ICES 2001; Piet & Rice 2004). We compared threat and exploitation status using a hits, misses and false alarms framework (Rice 2003; Dulvy *et al*. 2004; Piet & Rice 2004). There are two types of hit: a true positive and a true negative, resulting in a 2×2 table of outcomes (Table 1). With this framework: (i) a true positive hit is where a stock is exploited unsustainably (outside safe biological limits) and the threat criteria are met; (ii) a miss is where the species is exploited unsustainably (outside safe biological limits) but does not meet the threat criteria; (iii) a true negative hit is where a stock is exploited sustainably (within safe biological limits) and the threat criteria are not met; and (iv) a false alarm is where the stock is exploited sustainably (within safe biological limits) but the threat criteria are (erroneously) met.

A good set of criteria should avoid false alarms and minimize misses according to signalling theory (Rice 2003). This analysis has wide-ranging implications because it tests for consistency between the advice on population status provided by fisheries and conservation scientists.

Methods

DATA SOURCES

Time series of numbers at age and spawner–recruit data for 76 commercially exploited North-East Atlantic stocks, comprising 21 species (62 stocks) of fish and three species (14 stocks) of invertebrates, were extracted from ICES stock assessments (www.ices.dk). Both sexes were combined in the assessments, except for the Norway lobster *Nephrops norvegicus* L., where only females were used. Life-history data and additional population information were extracted from published compilations and stock assessments (see Appendix 1). Adult numbers were calculated as the sum of numbers in age classes older than the age at 50% maturity. Body growth rate, k, and asymptotic size, L_∞, were estimated by fitting a von Bertalanffy function to mean lengths at age. Fecundity was calculated as the length at 50% maturity from length–fecundity relationships. Age at 50% maturity was calculated by logistic regression of the proportion

Table 1. Framework for assessing the performance of threat criteria in relation to the exploitation status of stocks reported by the Advisory Committee on Fisheries Management of the International Council for the Exploration of the Sea (adapted from Rice 2003; Dulvy *et al*. 2004)

	Stock meets threat criteria	Stock does not meet threat criteria
Exploited within safe biological limits	False alarm	Hit (true negative)
Exploited outside of safe biological limits	Hit (true positive)	Miss

of individuals mature at each age. Maximum age was taken as the maximum age used in stock assessments. Generation time was calculated as the midpoint between age at maturity and maximum age. The maximum population growth rate, r_{max}, was calculated using age at maturity and spawner production ($\tilde{\alpha}$, Myers, Mertz & Fowlow 1997, equation 8). Spawner production ($\tilde{\alpha}$) was estimated by maximum likelihood fit of a Ricker model to spawner–recruit time series (Hilborn & Walters 1992; Myers, Mertz & Fowlow 1997).

CATEGORIZING THREAT

Threat was categorized using three sets of criteria.

IUCN A1

The observed population decline rate was calculated over the greater of 10 years or three generations using time series of adult numbers. Declines of ≥ 90%, ≥ 70% or ≥ 50% meet the IUCN A1 criteria for Critically Endangered, Endangered and Vulnerable. These criteria are appropriate for populations 'where the causes of the reduction are clearly reversible and understood and ceased' (IUCN 2004). The A1 criteria have higher thresholds for decline than the A2–4 criteria. One might question how many stocks actually meet these assumptions, but these are the criteria under which most exploited marine fishes are currently assessed (e.g. Cavanagh *et al*. 2003).

American Fisheries Society

The AFS criteria were used to classify populations or stocks as vulnerable or otherwise using qualifying decline thresholds and a productivity index (Musick 1999). Productivity was categorized as high, medium, low or very low based on any of five life-history parameters (Musick 1999). The maximum population growth rate, r_{max}, was used to categorize productivity, if available. Otherwise the lowest productivity category calculated using the other life-history traits was used (Musick 1999). The observed population decline rate was calculated over the greater of 10 years or three generations for the corresponding productivity categories. The decline thresholds resulting in a vulnerable listing were 99%, 95%, 85% and 70% for high, medium, low and very low productivities, respectively.

IUCN E

The IUCN E 'quantitative analysis' criteria depend on the probability of extinction in the wild over a defined time frame. A population is categorized as Critically Endangered if it exhibits a ≥ 50% probability of extinction within 10 years or three generations, Endangered if it exhibits a ≥ 20% probability of extinction within 20 years or five generations, and Vulnerable if it exhibits a ≥ 10% probability of extinction within 100 years.

We used two population projection approaches to calculate extinction probabilities over the qualifying time frames: non-parametric simulation and diffusion approximation (Dulvy *et al*. 2004). The probability of extinction was calculated using a non-parametric simulation based on the observed distribution of the rate of change (Matsuda *et al*. 1998). The rate of change r_t was calculated from:

$$r_t = \ln(N_{t+1}/N_t) \qquad \text{eqn 1}$$

where N_t is the population size in year t. Under the assumption that r_t is an independent variable that reflects the probability distribution in each year, then a value for r_t can be used to project the current population forwards in time using $N_{t+1} = N_t \exp(r_t)$. This avoids assuming any underlying probability distribution of r_t (Matsuda *et al*. 1998). We calculated the empirical r_t values from each interval in time series of adult abundance and projected forward in time from the observed initial population size by bootstrapping r_t values with replacement at each time step. A slope was calculated by fitting a least-squares regression model between the simulated population sizes and the observed ones. This was repeated 5000 times and the average slope was calculated. If the average slope was between 0·75 and 1·25, the non-parametric model was judged to provide a reasonable fit to a time series. Stocks not meeting these criteria were not considered further. For the remaining stocks with reasonable model fits, extinction probabilities were calculated from 1000 replicate population projections beginning at the last observed population size. The probability of extinction was calculated as the number of projections attaining the extinction threshold as a proportion of the total number of projections.

Diffusion approximation methods assume that stochastic age-structured populations without density dependence behave as a stochastic discrete time model, where the finite rate of population increase, λ (Dennis, Munholland & Scott 1991, equation 64), is:

$$\lambda = \exp[\mu + (\sigma^2/2)] \qquad \text{eqn 2}$$

The parameters μ and σ^2 were estimated using two methods. The Dennis, Munholland & Scott (1991) method estimates μ and σ^2 from:

$$\mu = \text{mean}[\ln(N_{t+1}/N_t)] \qquad \text{eqn 3}$$

$$\sigma^2 = \text{variance}[\ln(N_{t+1}/N_t)] \qquad \text{eqn 4}$$

The Holmes (2001) method is relatively insensitive to sampling error, and μ and σ^2 are calculated from regression slopes of the mean or variance in running sums of adult abundance over time:

$$\mu = \text{slope of the regression of mean}[\ln(R_{t+1}/R_t)] \text{ vs. time} \qquad \text{eqn 5}$$

σ^2 = slope of the regression of variance$[\ln(R_{t+1}/R_t)]$ vs. time eqn 6

Running sums (R_t) of adult abundance (A) were calculated as:

$$R_t = \sum_{i=1}^{L} w_i \cdot A_{t-1} \quad \text{eqn 7}$$

where L is the number of census counts added to give a running sum at time t, and w_i is the weight given to each count. In practice it is difficult to obtain weightings, and so we used a weighting of $w_i = 1$ (Holmes 2001). We used a value of $L = 4$, which is the maximum possible for time series of the length used here (Holmes 2001). Population size at $t + 1$ was projected from the initial observed population size from:

$$N_{t+1} = N_t\lambda \quad \text{eqn 8}$$

using parameters estimated by both the Dennis (Dennis, Munholland & Scott 1991) and Holmes (Holmes 2001) methods. The probability of extinction was calculated only if the Dennis or Holmes methods provided parameter estimates resulting in a reasonable range of projections. Projections were judged to provide a reasonable fit if the average slope of the relationship between the observed and projected abundance was 0·75–1·25.

Extinction risk was calculated as the probability of declining from the last observed population size n_q to a lower threshold population size n_t from:

$$\pi = \left(\frac{n_t}{n_q}\right)^{\frac{2\mu}{\sigma^2}} \quad \text{eqn 9}$$

using equation 85 in Dennis, Munholland & Scott (1991).

Extinction probabilities were calculated for five future time periods (10 years, 20 years, 3 generations, five generations and 100 years) and six quasi-extinction thresholds (1, 50, 500, 5000, 50 000 and 500 000 individuals), recognizing the possibility that the effective population size is 10^2–10^5 times less than the census population size (Hutchinson *et al*. 2003; Rowe & Hutchings 2003; Dulvy *et al*. 2004).

CATEGORIZING EXPLOITATION STATUS

The exploitation status of 68 stocks was extracted from ICES ACFM reports (ICES 2001). Where possible, ACFM defines stocks as exploited 'inside' or 'outside' safe biological limits. A stock is reported to be 'outside safe biological limits' if mean recruitment is lower than if the stock were at its full reproductive capacity (ICES 2001). In order for stocks and fisheries exploiting them to be within safe biological limits, there should be a high probability (i) that the spawning stock biomass is above the threshold where recruitment is impaired (the threshold is defined as B_{lim}) and (ii) that the rate of fishing mortality F is below the threshold that will drive the spawning stock to B_{lim} (F_{lim}). We assumed that the Icelandic spring spawning herring was exploited outside safe biological limits (Jakobsson 1980), resulting in status classifications for a total of 69 stocks.

Results

IUCN A1 CRITERIA

The IUCN A1 criteria could be applied to all 76 stocks, and a total of 12 stocks declined at a sufficient rate to be classified as threatened (Fig. 1; see Appendix 2). The Icelandic spring-spawning herring declined by 99·9% in the last 10 years of the time series and was the only stock to meet the Critically Endangered criteria. Five stocks declined by ≥ 70% over the qualifying time period, meeting the Endangered criteria, including the West Galicia and North Portugal Norway lobster, North Sea cod, East Baltic cod, West Scotland and Ireland herring and Greenland halibut. Six stocks declined by ≥ 70% over the qualifying time period, meeting the Vulnerable criteria, including the Irish Sea cod, Norway coastal cod, West Scotland cod, Baltic sole, West Baltic herring and North Galicia Norway lobster (Fig. 1).

AFS CRITERIA

The AFS criteria could be applied to all 76 stocks but only one stock, the Icelandic spring-spawning herring, exhibited the combination of decline rate in adult numbers and productivity to qualify for the Vulnerable category (see Appendix 2). This stock met the vulnerable threshold for the following production metrics: k, fecundity, age at maturity and maximum age. Our results suggested that r_{max} did not necessarily produce the lowest productivity estimate. Production indices based on k, age at maturity and maximum age were low relative to r_{max}, and production assignments based on fecundity tended to be higher (see Appendix 2).

IUCN E CRITERIA

The non-parametric simulation provided reasonable model fits for 12 stocks. None had a high probability of reaching a population threshold $n_t \leq 1$ individual within any of the five time scales; hence none met the IUCN E threat criteria (Table 2). Four stocks had a high probability of reaching a population threshold of $n_t \leq 500\,000$ individuals within five generations or 100 years: Kattegat and Norwegian coastal cod, Iberian hake and the North Galician Norway lobster (Fig. 2 and Table 2). The North Galician Norway lobster stock had the highest extinction probability, $P = 0{\cdot}88$, of declining below a population threshold of $n_t \leq 500$ individuals within 100 years.

The diffusion approximation approach provided reasonable model fits for 12 stocks using the Dennis (Dennis, Munholland & Scott 1991) method and 14

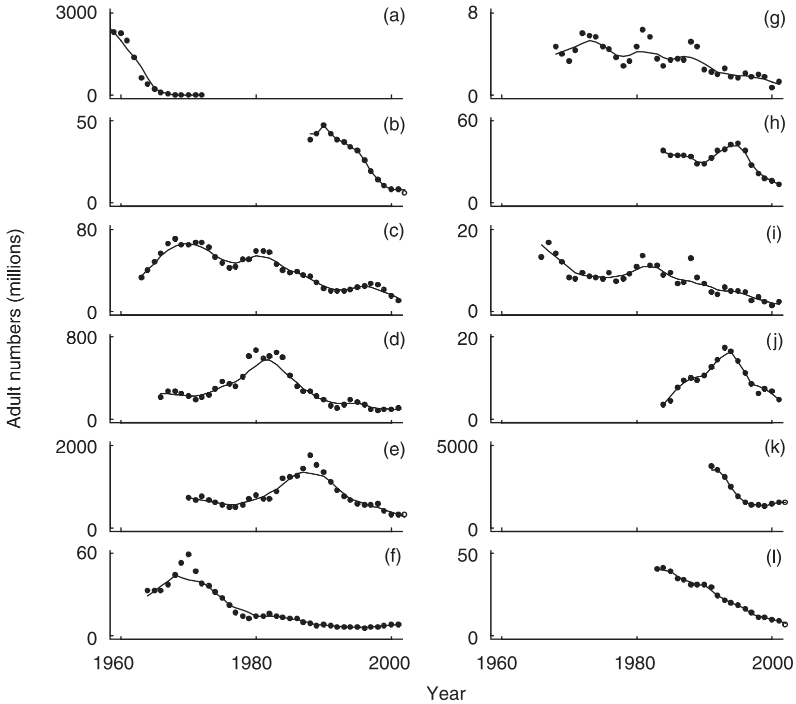

Fig. 1. Change in adult numbers over time for 12 stocks classified as threatened using the IUCN A1 decline criteria: (a) Icelandic spring spawning herring, Critically Endangered; (b) West Galicia and North Portugal Norway lobster, Endangered; (c) North Sea cod, Endangered; (d) East Baltic cod, Endangered; (e) West Scotland and Ireland herring, Endangered; (f) Greenland halibut, Endangered; (g) Irish Sea cod, Vulnerable; (h) Norwegian coastal cod, Vulnerable; (i) West Scotland cod, Vulnerable; (j) Baltic sole, Vulnerable; (k) West Baltic herring, Vulnerable; (l) North Galicia Norway lobster, Vulnerable. Note the *y*-axis scales vary. A smoothed loess line (span = 0·25) is presented to indicate the overall trend.

Table 2. Extinction probabilities of 12 stocks calculated using the non-parametric simulation method over five time periods and the highest quasi-extinction threshold population size n_t = 500 000. Y, year; G, generation

		Time span				
Species	Stock	10 Y	3 G	20 Y	5 G	100 Y
Anchovy	Bay of Biscay	0	0	0	0	0·002
Anglerfish	Celtic Sea, etc.	0	0	0	0	0
Atlantic cod	Kattegat	0	0	0·002	0·01	0·939
Atlantic cod	Norwegian coastal	0	0	0	0·05	1
Blue whiting	Northern ICES	0	0	0	0	0
Hake	Iberia	0	0	0	0	0·604
Herring	East Baltic	0	0	0	0	0
Herring	Iceland	0	0	0	0	0
Sole	East English Channel	0	0	0	0	0
Norway lobster	Botney Gut (North Sea)	0	0	0	0	0
Norway lobster	Farn deeps (North Sea)	0	0	0	0	0
Norway lobster	North Galicia	0	0·014	0·004	0·984	1

stocks using the Holmes (Holmes 2001) method. Both methods gave similar results when applied to the same stocks (Table 3). Five stocks met the critically endangered criteria, with an extinction probability, $P > 0·5$, of declining below a population threshold $n_t < 1$ within both 10 years and three generations using the Dennis method: Kattegat cod, Norway coastal cod, Iberian hake, East Baltic herring and North Galician Norway lobster (Table 3). Seven stocks met the critically endangered criteria, with an extinction probability, $P > 0·5$, of declining below a population threshold $n_t < 1$ within both 10 years and three generations using the Holmes method: Kattegat cod, Norway coastal cod, Northern and Iberian hake and three Norway lobster stocks (North Galician, South-West and South Portugal and Bay of Biscay; Table 3).

COMPARISON OF THREAT AND EXPLOITATION STATUS

A total of 46 (67%) stocks were outside safe biological limits (exploited unsustainably) and 23 (33%) were inside safe biological limits (exploited sustainably). The exploitation status was compared with IUCN A1,

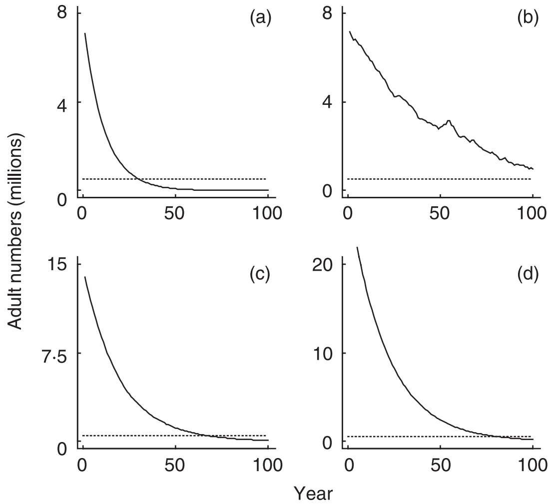

Fig. 2. Projected change in average adult numbers of four populations over 100 years using the non-parametric simulation approach: (a) North Galicia Norway lobster; (b) Kattegat cod; (c) Norway coastal cod; (d) Iberian hake. The horizontal dotted line represents a quasi-extinction threshold of 500 000 individuals.

AFS and IUCN E threat status. The non-parametric simulation approach was not considered here as none of the stocks assessed with this method met the IUCN E criteria.

None of the threat metrics produced false alarms, i.e. none of the stocks was found to be threatened but was exploited within safe biological limits (Table 4). The quantitative IUCN E metrics both produced higher hit rates and lower miss rates than the decline rate metrics (IUCN A1 and AFS) but could be applied to fewer stocks. The AFS criteria were the most conservative, with the lowest true positive hit rate and the highest true negative hit rate (Table 4).

Discussion

Threat criteria and fisheries stock assessments provide comparable information on the status of populations exploited by North-East Atlantic European fisheries. We found no evidence that the application of threat criteria would raise false alarms: none of the threatened stocks was classed as exploited within safe biological limits by ICES. These results lead to two conclusions. First, in every case stocks classified as threatened are unsustainably exploited 'outside safe biological limits'. Secondly, stocks identified as 'outside safe biological limits' may, in some cases, also be considered threatened. Thus management advice to reduce fishing mortality on stocks outside safe biological limits is consistent with the requirement to reduce the risk of extinction, and reductions in fishing mortality will meet the concerns of both fisheries and conservation interests.

Our findings differ from previous analyses suggesting decline criteria may be prone to false alarms when applied to exploited marine species (Matsuda *et al*. 1998; Punt 2000). Both previous studies evaluated the older IUCN A decline criteria, which had lower decline thresholds, where declines of 20%, 50% and 80% resulted in Vulnerable, Endangered and Critically Endangered categorizations. Also Punt (2000) defined 'extinction' as actual extinction, and Matsuda *et al*. (1998) defined a low quasi-extinction threshold (500 individuals), whereas we used higher quasi-extinction thresholds to account for very low effective population sizes in broadcast spawners (Hauser *et al*. 2002; Hutchinson *et al*. 2003; Hoarau *et al*. 2005). We suspect this change in decline thresholds may underlie the difference in the findings between this and the previous studies; however, we cannot rule out the possibility that the differences may be the result of the different species and modelling approaches used.

Table 3. Extinction probabilities for 18 stocks calculated using two diffusion approximation methods over five time periods and an extinction threshold of $n_t = 1$. Y, year; G, generation

		Dennis, Munholland & Scott (1991) method					Holmes (2001) method				
Species	Stock	10 Y	3 G	20 Y	5 G	100 Y	10 Y	3 G	20 Y	5 G	100 Y
Anglerfish	Celtic Sea, etc.	0	0	0	0	0	0	0	0	0	0
Atlantic cod	Kattegat	1	1	1	1	1	1	1	1	1	1
Atlantic cod	Norwegian coastal	1	1	1	1	1	1	1	1	1	1
Blue whiting	Northern ICES	0	0	0	0	0	0	0	0	0	0
Hake	Iberia	1	1	1	1	1	1	1	1	1	1
Hake	North Sea, etc.	–	–	–	–	–	1	1	1	1	1
Herring	Iceland	0	0	0	0	0	0	0	0	0	0
Herring	East Baltic	1	1	1	1	1	–	–	–	–	–
Herring	Gulf of Riga	–	–	–	–	–	0	0	0	0	0
Megrim	Celtic Sea, etc.	–	–	–	–	–	0	0	0	0	0
Sole	East English Channel	0	0	0	0	0	–	–	–	–	–
European oyster	Solent	–	–	–	–	–	0·09	0·08	0·08	0·07	0·04
Norway lobster	Botney Gut	0	0	0	0	0	–	–	–	–	–
Norway lobster	Farn deeps	0	0	0	0	0	0	0	0	0	0
Norway lobster	Firth of Forth	0	0	0	0	0	–	–	–	–	–
Norway lobster	North Galicia	1	1	1	1	1	1	1	1	1	1
Norway lobster	Bay of Biscay	–	–	–	–	–	1	1	1	1	1
Norway lobster	South-West and South Portugal	–	–	–	–	–	1	1	1	1	1

Both the decline rate criteria and quantitative criteria indicate that, in addition to the Icelandic spring-spawning herring, 17 overexploited fish and invertebrate stocks may be at increased risk of extinction. These stocks include six cod stocks (North Sea, Irish Sea, East Baltic, West Scotland, Kattegat and Norwegian coast), Iberian and Northern hake, Greenland halibut, Baltic sole, West and East Baltic herring, West Scotland and Ireland herring and four Norway lobster stocks (Bay of Biscay, North Galicia, West Galicia and North Portugal and South-West Portugal). Some of the stocks flagged here have undergone a rapid increase followed by a rapid decrease in abundance, for example Baltic sole and West Scotland and Ireland herring, and threat assessment may be less reliable for strongly fluctuating stocks. The quantitative methods applied here are purely statistical with no mechanistic basis, and therefore the estimate of threat assumes that past management regimes, environmental conditions, habitat availability and quality will also apply in the future.

There are a number of caveats that should be borne in mind when interpreting our results. Marine threat analyses will always involve a trade-off between the degree of threat faced by populations and the availability of data. There are usually few data available for assessing the most threatened marine populations; even when data are available power is too low to detect all except the steepest declines (Dulvy, Sadovy & Reynolds 2003; Maxwell & Jennings 2005). Consequently we were unable to include the most threatened European species in this study, such as populations of common sturgeon *Acipenser sturio* L., common skate *Dipturus batis* L., white skate *Rostroraja alba* (Lacepède 1803), long-nose skate *Dipturus oxyrhinchus* (L.), angel shark *Squatina squatina* (L.), tope *Galeorhinus galeus* (L.), largetooth sawfish *Pristis perotteti* Müller & Henle 1841, common sawfish *Pristis pristis* (L.) and Atlantic halibut *Hippoglossus hippoglossus* (L.) (Dulvy, Sadovy & Reynolds 2003; Baillie, Hilton-Taylor & Stuart 2004). We report results based on a well-understood and consistent marine data set with sufficient variance in threat and exploitation status: 67% of stocks were exploited unsustainably and 24% were threatened. The exclusion of the most threatened species does not undermine this study, indeed most of these threatened species are likely to score as true positive hits, being both unsustainably exploited and threatened.

Maximum age is likely to be underestimated as stock assessments often pool all individuals greater than a certain age, because of the uncertainty of ageing. This negative bias may result in less precautionary (lower) decline rates being measured over a reduced three-generation time window and more precautionary (lower) AFS productivity estimates. The quantitative methods also assume that the variance is solely the result of the population dynamics and demography (Dennis, Munholland & Scott 1991; Holmes 2001). These numbers at age data were derived from Virtual Population Analysis and may be subject to additional variation as a result of observation, process and model error, in addition to the intrinsic variance of the stocks (ICES 2001; Dulvy *et al*. 2004). Observation errors are known to influence the threat assessment of stocks (Punt 2000), but these are unimportant for our conclusions as the Holmes (2001) method corrects for observation error. However, the model fitting in the stock assessment process should already have eliminated much of the actual observation error in the numbers at age. The quantitative methods used here do not incorporate density dependence. Relatively few stocks can be described by the density-independent diffusion approximation methods, suggesting the dynamics of the other stocks may be better described with density-dependent population viability methods. It is not clear whether this omission leads to overestimation or underestimation of extinction risk. If stocks become more productive in small populations, perhaps because of less competition between individuals for food and space, then the extrapolations into the future will be too pessimistic. Conversely, there is also a very real likelihood of Allee effects (depensation) occurring in marine populations that would lead to greater risk of extinction than indicated by our simulations (Rowe & Hutchings 2003; Dulvy, Freckleton & Polunin 2004; Gascoigne & Lipcius 2004a,b; Rose 2004). Further analyses might explore the effects of model and process errors and incorporate age structure and a range of density-dependent possibilities (Staples, Taper & Dennis 2004).

Our analysis assumes that the ICES exploitation status determinations are 'correct'. While this may be true for some stocks it will not be true for others, as it is difficult to detect long-term declines in abundance and stock status based on relatively short times series. Threat metrics and ICES exploitation status are also based on the same data and are not strictly independent appraisals

Table 4. The proportion (%) of stocks meeting each of four possible outcomes (true positive hit, true negative hit, miss and false alarm) and the total number of stocks for which both stock status and threat status were available. IUCN E^D was calculated using the Dennis, Munholland & Scott (1991) method and IUCN E^H was calculated using the Holmes (2001) method

Threat criteria	False alarm	Hit (positive)	Hit (negative)	Miss	Number of stocks compared
IUCN A1	0	16	36	48	64
AFS	0	2	36	62	58
IUCN E^D	0	36	45	18	11
IUCN E^H	0	50	25	25	12

of status. We have only compared threat to assessments of exploitation status relative to safe biological limits as defined in a European fisheries management system (ICES 2001a), and there is further scope to compare threat to reference points used in other fisheries management systems.

This analysis supports suggestions that AFS criteria decline rate thresholds are overly conservative and may overlook threatened species (Hutchings 2001a). The AFS criteria assume high resilience in teleost fishes from high interannual variability and reproductive output, but these assumptions have no theoretical or empirical basis (Hutchings 2001a; Sadovy 2001; Denney, Jennings & Reynolds 2002; Hutchings & Reynolds 2004; Myers & Worm 2005). The new CITES criteria use the same decline thresholds as the AFS criteria but are combined with a different decline measure, resulting in even more conservative threat criteria. Marine species have to decline to 5–30% of virgin population sizes before they are considered threatened under the new CITES criteria (FAO 2002; Mace *et al*. 2002). Our findings suggest that a declines of > 50% over the greater of 10 years or three generations are far in excess of the point of maximum yield and the safe biological limits of many stocks (ICES 2001; Beddington & Kirkwood 2005). In contrast, to meet the new CITES criteria, species must decline even further, by 70–95% from virgin population size, not just over the greater of 10 years or three generations, to qualify as threatened. Consequently the new CITES criteria, like AFS criteria, risk postponing population and species threat listings until their probability of extinction is unduly high and their probability of recovery unduly low.

Concerns about the conservation status of fish stocks have led to the development of regional criteria for assessing extinction risk. For example, the Oslo–Paris Commission for the Protection of the Marine Environment of the North-East Atlantic (OSPAR) has recently adopted the use of a newly developed and relatively unknown set of criteria known as the Texel-Faial criteria (ICES 2001). While the Texel-Faial criteria include a decline criterion, the thresholds are undefined and hence could not be applied to the stocks considered here. Such criteria often have a weaker theoretical grounding than the existing IUCN criteria and, given the apparently reliable performance, international recognition and peer-review of the IUCN criteria, and the consistency with which IUCN criteria reflect ICES fisheries' management stock assessments, there is little reason to invent new threat criteria without a strong scientific case.

Complex stock assessment models and the IUCN decline criteria both indicate when stocks are exploited unsustainably and may be at risk of extinction. Stock assessments are a financial and logistical impossibility for many target and non-target species that have been depleted by fisheries exploitation, and our results suggest that IUCN criteria provide simple guidance on exploitation status. As this guidance is likely to be consistent with the guidance provided by stock assessments, at least in terms of recommending a direction of change in fishing mortality, the IUCN approach and stock assessment can be regarded as complementary and consistent with an ecosystem approach to fisheries management (Sinclair & Valdimarsson 2003). In management terms, the most important issue is not expected to be the difference between the approaches, but how to reduce fishing mortality effectively and to move exploitation towards sustainability.

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Supplementary material

The following supplementary material is available for this article online:

Appendix 1. Summary of study species, geographical location and their life-history traits

Appendix 2. Threat and exploitation status of 76 stocks

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