Threat and decline in fishes: an indicator of marine biodiversity

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Abstract: Recent policy commitments aim to reduce biodiversity loss and integrate environmental concerns into fisheries management. However, there are few operational indicators for reporting biodiversity trends and judging progress in relation to management objectives. Here we develop a threat indicator based on the population status of a suite of 23 North Sea fishes from 1982 to 2001 estimated using World Conservation Union (IUCN) Red List decline criteria. The composite indicator was calculated from the weighted average of the threat scores of individual species in each year and varies from 0 to 3, where a score of 3 is equivalent to each species qualifying as "critically endangered". The proportion of threatened fishes, their degree of threat, and the composite indicator value increased steadily over time. The composite indicator value has been >1 since the late 1990s, equivalent to all species meeting the "vulnerable" criterion. A suitable reference trajectory, consistent with the World Summit on Sustainable Development commitment to "achieve by 2010 a significant reduction of the current rate of biodiversity loss" would be a significant reduction in the rate of increase in this indicator before 2010, a limit reference point could be 1 (all species vulnerable) and a target reference point could be 0 (no threatened species).

Résumé : Les engagements récents de la politique de gestion des pêches visent à réduire la perte de biodiversité et à intégrer les préoccupations environnementales dans la gestion. Il existe cependant peu d'indicateurs opérationnels pour décrire les tendances de la biodiversité et pour juger des progrès en fonction des objectifs de gestion. Nous développons ici un indicateur de menace basé sur le statut démographique d'une série de 23 poissons de la mer du Nord de 1982–2001 et estimé à partir des critères de déclin de la liste rouge de l'Union internationale de la conservation de la nature (IUCN). L'indicateur composé se calcule à partir des moyennes pondérées des valeurs de la menace pour chaque espèce individuelle pour chaque année et varie de 0–3; une valeur de 3 équivaut à chacune des espèces atteignant les critères de classification d'espèce gravement menacée. La proportion d'espèces en danger, le degré de menace contre elles et les valeurs de l'indicateur composé augmentent régulièrement au cours des années. L'indicateur composé a atteint une valeur >1 depuis la fin des années 1990, ce qui équivaut à toutes les espèces ayant atteint les critères de classification d'espèce vulnérable. Une trajectoire de référence appropriée, en accord avec les engagements du Sommet mondial sur le développement durable d'« atteindre avant 2010 une réduction significative du taux actuel de perte de biodiversité », serait une réduction significative du taux de croissance de cet indicateur avant 2010; un point de référence limite pourrait être une valeur de 1 (toutes les espèces vulnérables) et un point de référence cible pourrait être 0 (aucune espèce menacée).

[Traduit par la Rédaction]

Introduction

There are a number of national, regional, and international policy commitments to halt or reduce the rate of marine biodiversity loss by 2010 (Sainsbury and Sumaila 2003; Rogers and Greenaway 2005). The main threat to marine biodiversity is exploitation, which has resulted in population declines and extinctions, habitat degradation, and ecosystem changes (Jackson et al. 2001; Hutchings and Baum 2005; Myers and Worm 2005). Because fishing is a recognised cause of biodiversity loss, indicators of marine biodiversity are needed to support conservation and fisheries management (Rice 2003; Cury and Christensen 2005; Hutchings and Baum 2005). One option for developing an indicator of the biodiversity status of marine fishes is to tally up the number of species or populations that are listed as threatened, by all threat-listing schemes or one specific scheme, in any given region and year. This is the basis of the latest terrestrial threat indicators. The state and rate of change in terrestrial biodiversity is provided by World Conservation Union (IUCN) Red List indicators for birds, mammals, and amphibians (Baillie et al. 2004; Butchart et al. 2004). The development of a similar Red List indicator for marine species is not well advanced because the threat status of all marine species has not been comprehensively surveyed. Although this is due mainly to the magnitude of the task, fishes alone comprising

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Table 1. Age of maturity of the species included in the analysis.

Common name	Species	Age at maturity	Reference
Starry ray	*Amblyraja radiata*	5†	Walker 1998
Wolf fish	*Anarhichas lupus*	6*	Maxwell and Jennings 2005
Grey gurnard	*Eutrigla gurnardus*	3*	Maxwell and Jennings 2005
Cod	*Gadus morhua*	3.8*	Maxwell and Jennings 2005
Tope	*Galeorhinus galeus*	9*	Maxwell and Jennings 2005
Witch	*Glyptocephalus cynoglossus*	5.5†	Muus and Neilsen 1999
Long rough dab	*Hippoglossoides platessoides*	2.6*	Maxwell and Jennings 2005
Cuckoo ray	*Leucoraja naevus*	8*	Maxwell and Jennings 2005
Dab	*Limanda limanda*	2.3*	Maxwell and Jennings 2005
Angler fish	*Lophius piscatorius*	4.5*	Maxwell and Jennings 2005
Haddock	*Melanogrammus aeglefinus*	2.5*	Maxwell and Jennings 2005
Whiting	*Merlangius merlangus*	1.5*	Maxwell and Jennings 2005
Hake	*Merluccius merluccius*	3.8*	Maxwell and Jennings 2005
Lemon sole	*Microstomus kitt*	4.5†	Muus and Neilsen 1999
Common ling	*Molva molva*	5.5†	Cohen et al. 1990; Deniel 1990
Plaice	*Pleuronectes platessa*	2.5*	Maxwell and Jennings 2005
Saithe	*Pollachius virens*	4.7*	Maxwell and Jennings 2005
Thornback ray	*Raja clavata*	10*	Maxwell and Jennings 2005
Spotted ray	*Raja montagui*	8*	Maxwell and Jennings 2005
Turbot	*Scophthalmus maximus*	6*	Maxwell and Jennings 2005
Brill	*Scophthalmus rhombus*	5†	Froese and Pauly 2005
Sole	*Solea solea*	2.5*	Denney et al. 2002
Spurdog	*Squalus acanthias*	6.5*	Maxwell and Jennings 2005

*Female age at 50% maturity.
†Sex and details of maturity stage were unknown.

half of all vertebrate species, the process may also have been delayed by ongoing debates about the validity of applying threat criteria to marine species (Hutchings 2001; Dulvy et al. 2003; Baillie et al. 2004).

Threat listings of terrestrial species, particularly birds and mammals, have been in place for some time, providing a time series that can be used to assess rate of change in aspects of terrestrial biodiversity (Butchart et al. 2004). Even if a comparable marine threat listing process were put in place now, it would take time to apply it comprehensively, and it would only provide the first data point in the time series necessary to assess the change in marine biodiversity. In the absence of capacity to assess the status of all marine species, or a suitable random sample, one option is to focus solely on fish. There are good reasons for focussing on fish: they comprise a large proportion of the biomass in marine ecosystems, their patterns of diversity are representative of other taxa, they provide ecosystem services to humans, and they show clear responses to fishing (Callaway et al. 2002; Rice 2003).

One possible approach to developing a biodiversity indicator is to assess threat for an assemblage of marine species for which time-series abundance data already exist. Fish species-abundance data are routinely collected to support existing fisheries management activities and can be used to provide descriptions of threat for marine fishes (Dulvy et al. 2005; Maxwell and Jennings 2005). Although a threat index derived from species-abundance data collected on trawl surveys will not provide a representative measure of marine biodiversity in the survey area, it does capture aspects of biodiversity about which society is concerned (Jennings 2004, 2005). This is broadly the sense in which the Red List indices for birds, mammals, and amphibians are already used to report on the state of the world's biodiversity, and they provide a pragmatic way of informing society about changing patterns of biodiversity (Butchart et al. 2004, 2005).

Declines of many vulnerable species occurred before the start of fisheries survey time series, and the application of decline criteria to survey data may underestimate the level of threat (Hutchings and Baum 2005). Moreover, if decline criteria are applied at any point in time and the last 10 years or three generations worth of data are used to assess decline, then threat will be underestimated for species that underwent major declines before this period and (or) that have become so scarce that the statistical power of surveys to detect contemporary trends is poor (Maxwell and Jennings 2005). These are examples of the shifting baseline effect, where the baseline represents an increasingly exploited state over time and may mask the true magnitude of fishing effects on biodiversity (Pauly 1995; Baum and Myers 2004; Sáenz-Arroyo et al. 2005).

Here we develop a biodiversity indicator for assessing and reporting the threat status of a suite of marine fishes. We assess threat for each species over time by applying the IUCN A1 decline criteria to fisheries-independent survey abundance data and derive a composite threat indicator based on the weighted average of species threat scores. To assess the effects of shifting baselines on the indicator, we determine how changes in the time window (survey years) used to calculate the indicator affect trends in the biodiversity indicator and the assumed status of the suite of fishes.

Materials and methods

We used the North Sea English groundfish survey data to assess changes in abundance and threat status of adult fishes. Currently, a survey grid of 75 stations is fished annually. Stations were fished with a Granton demersal trawl until 1991, but from 1992, a Grand Ouverture Verticale (GOV) demersal trawl was used. Tow duration for up to 1991 was 60 min, but for 1992 onwards, the tow duration was 30 min (B. Harley, Centre for Environment, Fisheries and Aquaculture Science (CEFAS), Lowestoft, Suffolk NR33 0HT, UK, personal communication). The Granton trawl gear was fitted with a cod-end liner of 14 mm stretched mesh and the GOV trawl was fitted with a cod-end liner of 20 mm stretched mesh. Both gears were towed at a speed of approximately 4 knots. All fishes caught were identified and measured. Catch rates were raised to number of individuals caught per 60 min tow. Not all stations in the survey grid are fished every year because of poor weather, equipment damage, or ship failure, and in the earlier surveys, more stations were sometimes surveyed (for more details see Maxwell and Jennings 2005).

Species were excluded if they were known to be poorly sampled by the gear, rare, or found in peripheral North Sea habitats and had a maximum length of <40 cm (Sparholt 1990; Knijn et al. 1993; Maxwell and Jennings 2005). Specifically, species were excluded if <150 individuals had been caught in the history of the survey, or if morphology, behaviour, and habitat preference were expected to lead to very low and variable catchability. Individuals <40 cm have increased in abundance in recent years, possibly as a result of the depletion of their larger predators, so we restricted this analysis to species with a maximum size greater than 40 cm total length (Daan et al. 2005). The 23 species retained for analysis were representative of the breadth of morphology, life histories, ecology, and taxonomic diversity of the larger bottom-dwelling fishes sampled on the English groundfish survey in the North Sea. The average age of maturity of all species in this suite of fishes was 4.9 years (Table 1.)

Threat was assessed using IUCN A1 decline criteria, which are based on the reduction in population size over the greater of 10 years or three generations in which causes are reversible, understood, and have ceased (IUCN 2004). The qualifying decline thresholds are "critically endangered" (≥90% decline), "endangered" (≥70% decline), and "vulnerable" (≥50% decline). Adult abundance estimates were $\log(x + a)$ transformed, where a was defined as one-half the minimum nonzero abundance value in the time series (Venables and Ripley 2002).

We measured threat retrospectively over the time series using two methods of measuring the change in adult numbers between two points in time: rate of decline and extent of decline. The rate of decline was estimated using subsets of the annual abundance data, as determined by a "moving window". A least squares linear model was fitted to a moving window spanning x years (we chose windows of 10 and 15 years) to represent the qualifying time spans used for measuring decline in the IUCN Red List A1 decline criteria. Fifteen years is approximately equivalent to three generations for the fishes considered here (Table 1). Specifically, the linear model was fit to the first x years of data, $t_1 - t_{1+x}$, and to each successive year, i.e., $t_2 - t_{2+x}$, $t_3 - t_{3+x}, \ldots, t_{\text{maximum}-x} - t_{\text{maximum}}$. Thus model fits and the estimates of percent change in abundance are determined by the span of the moving window and its placement in the time series. The extent of decline was calculated by comparing subsequent changes in adult abundance with a fixed start date of 1982. A linear model was fit to the first 10 years of data, $t_1 - t_{10}$, and to each successive year, i.e., $t_1 - t_{11}$, $t_1 - t_{12}, \ldots, t_1 - t_{\text{maximum}}$. The percent change in adult abundance was calculated from the start (t_1) and end (t_{10} to t_{maximum}) abundances as predicted from the least squares linear model fit (IUCN 2004).

Species that had met one of the decline criteria in any year of the time series and were thus identified as threatened were not delisted (categorized as not threatened) unless their numerical abundance had increased beyond a preset threshold. As an example, we chose a preset threshold of the mean catch rate averaged over the first 3 years of the time series. We chose the average of the first 3 years to minimize the

Fig. 1. Change in combined abundance of 23 large North Sea fishes over time: (*a*) average abundance ($\log_{10}$ numbers·h^{-1}); (*b*) differences in abundance between one year and the next ($\log_{10}$ numbers·h^{-1}); and (*c*) differences in abundance rescaled as a proportion of average abundance of the corresponding years ($\log_{10}$ numbers·h^{-1}). The vertical line indicates the year that the survey gear and tow length changed.

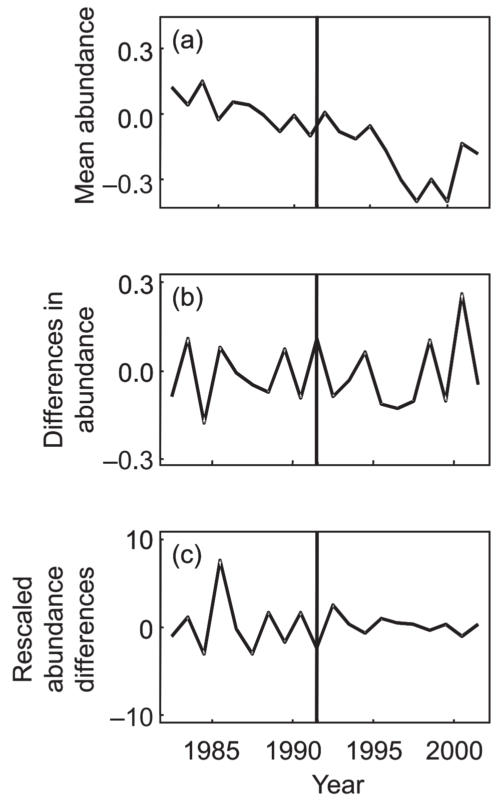
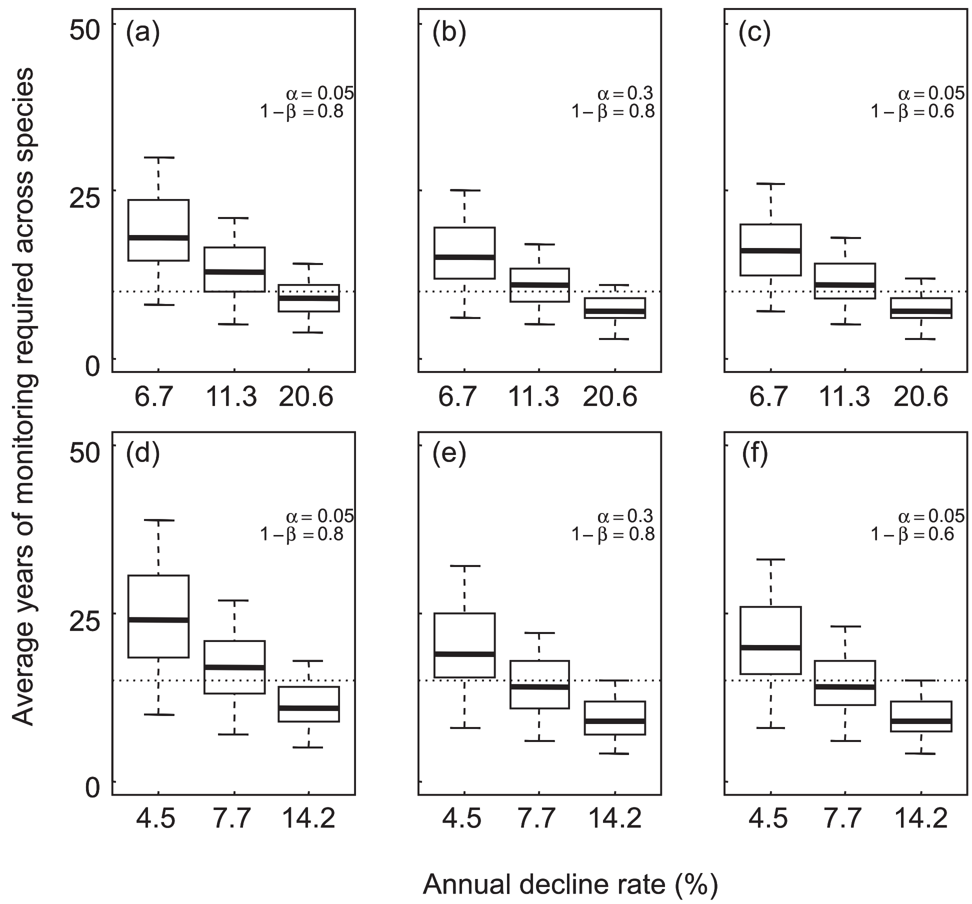


Fig. 2. Box-and-whisker plots of the distribution of the number of years of monitoring required to detect varying declines rates for two levels of power ($1 - \beta = 0.6$ and 0.8) and significance ($\alpha = 0.05$ and 0.3) across species. The dotted horizontal line denotes the qualifying time frames of (*a*, *b*, *c*) 10 years and (*c*, *d*, *e*) 15 years: (*a*) 10-year decline with $1 - \beta = 0.8$ and $\alpha = 0.05$; (*b*) 10-year decline with $1 - \beta = 0.8$ and $\alpha = 0.3$; (*c*) 10-year decline with $1 - \beta = 0.6$ and $\alpha = 0.05$; (*d*) 15-year decline with $1 - \beta = 0.8$ and $\alpha = 0.05$; (*e*) 15-year decline with $1 - \beta = 0.8$ and $\alpha = 0.3$; and (*e*) 15-year decline with $1 - \beta = 0.6$ and $\alpha = 0.05$.



influence of exceptionally high or low abundance estimates in any given year.

The composite threat indicator was calculated for each year as the weighted average of species threat scores. Individual species threat categorisations were weighted as vulnerable = 1, endangered = 2, and critically endangered = 3, following Butchart et al. (2004), and allocated to the final year of the period over which the decline was measured. Species declining at increasing rates will increase in contribution to the threat index as their status worsens from vulnerable to critically endangered. A composite threat indicator was calculated as the weighted average threat score of all species for each year. This indicator is readily interpreted because the scores can vary from 0 to 3, such that a score of 0 is equivalent to no species meeting any of the threat criteria and a score of 3 is equivalent to each species being critically endangered.

We explored the power to detect trends in adult numbers underlying the individual threat assessments. The power analysis assumes an overall nonspecified smooth trend for 1982–2002 followed by declining trend, q, beginning in 2003 and lasting for T years (Nicholson and Fryer 1992; Maxwell and Jennings 2005). The number of years (T) required to detect a specified declining trend (q) were calculated based on the observed variance (ψ^2), specified significance (α), and power ($1 - \beta$). The IUCN A1 criteria qualifying decline rates of 50%, 70%, and 90% over 10 or 15 years were converted to annual decline rates (q) using $q = 1 - (1 - d)^{(1/n)}$, where d is the proportional decrease over n years (Maxwell and Jennings 2005). Variance (ψ^2) was calculated for adult abundance using a difference-based method (Gasser et al. 1986; Nicholson and Jennings 2004). When assessing threat, it can be reasonably argued that the cost of not correctly identifying a threatened species (which is actually at high risk of extinction) may be higher than the cost of raising a false alarm (resulting in unnecessary management action), so we considered a range of type I error rates up to 0.30 (Peterman and M'Gonigle 1992; Maxwell and Jennings 2005).

Results

The average adult abundance of this suite of 23 large North Sea demersal fishes declined by 34% over the 21 years since 1983 (Fig. 1*a*). There was no evidence for a large effect of the change in survey gear or tow time on composite abundance (Figs. 1*b*, 1*c*). The greatest year-to-year differences in abundance were at the start of the time series and decreased over time. The year-to-year differences (Fig. 1*b*) and differences expressed as a proportion of average abundance

Fig. 3. Species threat scores in each year measured as (*a*) rate of decline with a 10-year window, (*b*) rate of decline with a 15-year window, and (*c*) extent of decline. Species are plotted in descending rank order of body size, with smallest species at the top. Point size is proportional to threat scores, with the largest, intermediate, and smallest symbols representing declines over the qualifying time of ≥90%, ≥70%, and ≥50%, respectively.

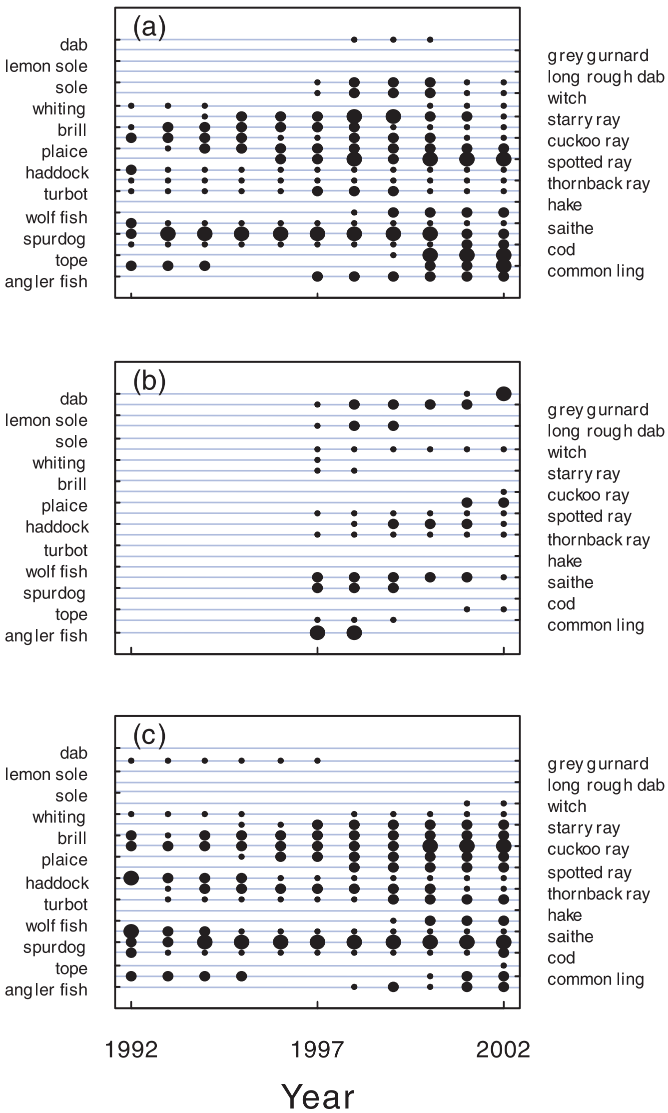
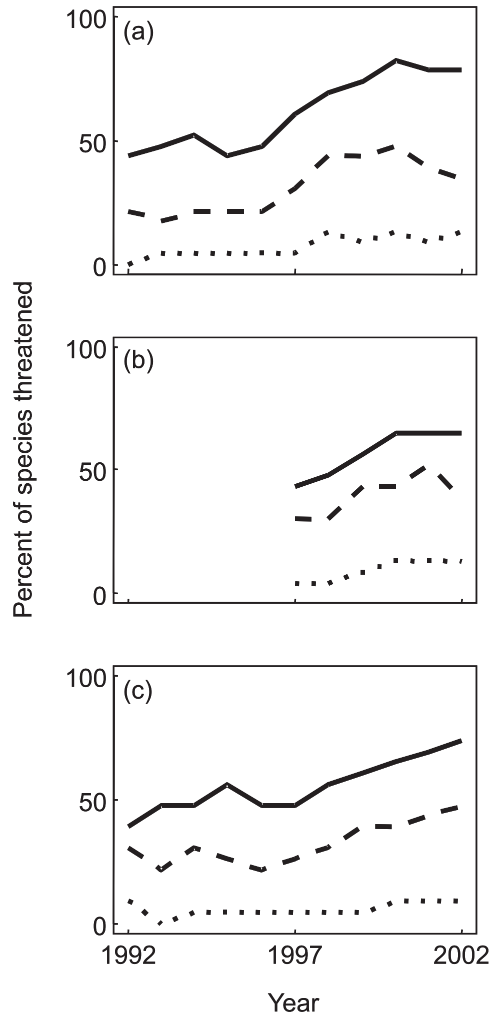


Fig. 4. Proportion of North Sea fishes meeting each of the three IUCN threatened categories (critically endangered, dotted line; endangered, broken line; and vulnerable, solid line), measured as (*a*) rate of decline with a 10-year window, (*b*) rate of decline with a 15-year window, and (*c*) extent of decline.



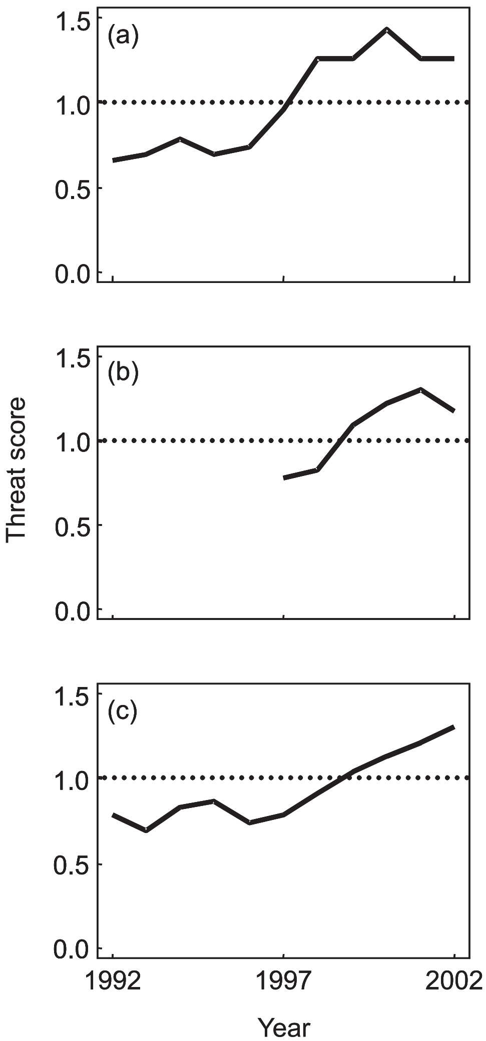
(Fig. 1*c*) spanning the survey changes in 1991–1992 are comparable to those in other years.

When averaged across species, the number of years of monitoring required to detect significant declines was usually longer than the qualifying time frame (Fig. 2). The qualifying time frame used was the longer of 10 years or 15 years, approximately three generations. Only the steepest annual declines >14.2% were detected within the qualifying time frames with high power ($1 - \beta = 0.8$) and significance ($\alpha = 0.05$) (Figs. 2*a*, 2*d*). Lower declines rates can be detected by accepting lower significance ($\alpha \leq 0.3$; Figs. 2*b*, 2*e*) or lower power ($1 - \beta \leq 0.6$; Figs. 2*c*, 2*f*).

Large-bodied species, including wolfish, cod, rays, and spurdog, consistently met one of the threat criteria (Fig. 3). The rate of decline criteria resulted in previously threatened species becoming less threatened toward the end of the time series (Figs. 3*a*, 3*b*). However, when using the extent of decline criteria, the number of threatened species and the degree of threat increased over time (Fig. 3*c*). The proportion of species declining by ≥70% and ≥50% and qualifying as endangered and vulnerable increased over time (Fig. 4). The proportion of endangered species increased to a peak and subsequently declined, and the proportion of vulnerable species increased before reaching a plateau for the rate of decline criteria (Figs. 4*a*, 4*b*). There is a continuing increase in the proportion of vulnerable and endangered species for the extent of decline criteria (Fig. 4*c*). There is a slight but highly variable increase in the number of species declining by ≥90% meeting the critically endangered criterion, with few (~5%) species qualifying (Figs. 4*a*–4*c*).

The composite threat score increased from ~0.7 in the early 1990s to a peak of >1 in the late 1990s (Fig. 5). The rate of decline criteria exhibited a peak and subsequent decline in threat score (Figs. 5*a*, 5*b*). The extent of decline criteria showed consistent year-on-year increases in threat score

Fig. 5. An indicator of threat over time for a suite of North Sea demersal fishes measured as (*a*) rate of decline with a 10-year window, (*b*) rate of decline with a 15-year window, and (*c*) extent of decline. The score can range from 0 to 3, and a score of 1 is equivalent to each species meeting the vulnerable criterion and is indicated with a dotted line.



until the end of the time series (Fig. 5*c*). Rate of decline criteria differed in the timing of peak threat, and this peak was lower and increasingly delayed as a result of longer time windows (Figs. 4*a*, 4*b*), and this lag is reflected in the composite threat score (Figs. 5*a*, 5*b*).

Discussion

We have described a composite threat indicator based on year-on-year changes in the weighted average of individual species threat status. The proposed composite marine biodiversity indicator responds to changes in the proportion of threatened species in a specified fish assemblage. The indicator provides information on trends in aspects of biodiversity regarded as important by society and has potential to inform on progress toward management objectives. Because it is calculated from species size abundance survey data, it could be adopted immediately in many regions. This is fortuitous when political commitments to the introduction of an ecosystem approach have been rapid, but the development of a supporting science base has been relatively slow (Fulton et al. 2005; Rice 2005).

Values for the composite indicator were determined from trends in trawl survey catch per unit effort (CPUE) data but could equally be calculated using abundance estimates from stock assessments, underwater visual census, and other types of surveys. All the abundance estimates will be subject to some bias. In the case of trawl survey CPUE, distribution–abundance relationships (MacCall 1990) and changes in distribution due to climate change (Beare et al. 2004; Perry et al. 2005) are two processes that reduce the correlation between CPUE and true abundance. However, CPUE data are the only available measures of abundance for most fish species and bias can be reduced by basing the composite index on CPUE trends in a suite of species with distribution centres in the survey area.

In the North Sea, the overall abundance of the suite of species declined by ~34%. This relatively modest decline belies a high degree of threat for individual species. The composite threat indicator suggests that, on average, all species were threatened from the late 1990s onwards. This reiterates previous work that suggested that combining abundance trends of a number of species masks potentially important biodiversity issues (Dulvy et al. 2000). To support effective reporting and management decision making, the composite threat indicator should respond to signal rather than noise (Rice 2003; Rice and Rochet 2005). However, when assessing threat, the cost of not correctly identifying a threatened species (high risk of extinction) may be higher that the cost of raising a false alarm (unnecessary management action). Only the steepest declines were detected within the qualifying time frames when a type I error rate of 0.05 was accepted, but when this rate was increased to 0.30, then most declines were detected. Following Peterman and M'Gonigle (1992), we argue that a higher type I error rate, in this case 0.30 (falsely reporting a decline when one has not occurred on 30% of occasions) can be accepted when detecting declines and is consistent with the precautionary approach. If high type I error rates were unacceptable to users, then the indicator could be derived from the proportion of species undergoing the steepest declines (≥90%) in the qualifying time period. However, trends in the proportion of these (critically endangered) species are weak and highly variable, and an indicator based solely on the highest decline threshold may be unreliable or uninformative. Instead we suggest that the trends in each of the three threat categories be used to interpret the composite indicator, rather than be used as indicators. These component threat trends may be more informative if applied to surveys that yield less variable abundance estimates. Ultimately, the degree of acceptance of higher type I error rates and therefore the risk of unnecessary action (Di Stephano 2003) will be based on societal perceptions of risk, and one of the few aspects of marine biodiversity that policy makers have consistently emphasised is the risk of biodiversity loss (World Summit on Sustainable Development (WSSD) 2002). Moreover, nonsignificant trends in abundance can be informative (Jennings et al. 1998), and fish species have been shown to be threatened in the absence of suitable data for statistical treatment (Musick et al. 2000; Dulvy et al. 2003; Sadovy and Cheung 2003).

We suggest that the extent of decline method be used to assess the change in threat status and the composite indicator over time for two reasons. The extent of decline has the advantage of using all available data from the start of the time series to the year in which the indicator is calculated (IUCN 2004). Any threat indicator based on this method not only would minimise the influence of natural variability, but also would be consistent with the IUCN Red List guidelines. Thus species meeting one of the criteria as applied here could be considered for listing. Indeed, some of the species receiving high threat scores are already listed as threatened by the IUCN Red List, e.g., tope, spurdog, and cod (Baillie et al. 2004). The other key benefit of using the extent of decline method is that the decline is measured relative to abundance in the early years of the survey. This avoids the bias introduced by shifting baselines during the survey period, as occurs with the rate of decline method.

A possible disadvantage of using extent of decline is that the composite indicator may be biased against detecting genuine population recovery; however, our use of a delisting criterion safeguards against this problem. We delisted species once they recovered to the abundance (average catch rate) observed at the beginning of the time series. For assessed populations, standard assessment models could be used to calculate a more realistic baseline, based on the theoretical abundance in the absence of fishing mortality (Quinn and Deriso 1999).

Neither the decline methods used nor the data presented describe the dramatic reductions in abundance or regional extinctions of some North Sea species before 1982. Species thus affected include common sturgeon (*Acipenser sturio*), common skate (*Dipturus batis*), white skate (*Rostroraja alba*), long-nose skate (*Dipturus oxyrhinchus*), angel shark (*Squatina squatina*), northern bluefin tuna (*Thunnus thunnus*), and Atlantic halibut (*Hippoglossus hippoglossus*) (Brander 1981; Walker and Hislop 1998; Baillie et al. 2004). The disappearance of most of these species was identified using informal methods, such as historic and present-day comparisons of species lists or catch rates, and traditional knowledge (Quero 1998; Rogers and Ellis 2000; Wolff 2000). Although it would be possible to include some of these demersal species in the composite threat indicator, both to highlight their status and to allow for the effects of future recovery, such recovery is unlikely in the medium term. So, unless these species do start to reappear in research surveys, we suggest that lists of threatened species could be reported alongside the composite indicator, thereby avoiding reports of biodiversity change that are biased by the shifting baseline syndrome (Pauly 1995).

The composite threat indicator is a higher-level indicator and cannot directly guide management action. Moreover, although fishing has had a pervasive long-term impact on the overall abundance of large fishes in the North Sea (Jennings and Blanchard 2004), the abundance of individual species also responds to climate change, ecological interactions, and changes in primary production (Beaugrand et al. 2002; Brander 2005). Management to meet biodiversity targets would require that the species responsible for trends in the composite indicator be identified, as these species may need to be managed individually, in small groups, and (or) by region given the large spatial extent of the North Sea and the diverse fisheries that operate there.

Reference points, trajectories, or directions are used to relate the values of indicators to management objectives (Caddy and Mahon 1995; Jennings and Dulvy 2005; Link 2005). We suggest that a suitable reference trajectory would be a significant reduction in the rate of increase of the composite indicator before 2010. This is consistent with policy commitments to achieve a significant reduction in the rate of loss of biodiversity by 2010 (WSSD 2002). Limit and target reference points are more difficult to define from a policy perspective, but we suggest that a biologically defensible limit reference point for the composite threat indicator is a score of 1. This is based on previous findings that fish species qualifying as threatened under IUCN A1 decline criteria are also outside the safe biological abundance limits defined by stock assessment scientists (Dulvy et al. 2005). This suggested limit reference point has been exceeded by the threat index since the late 1990s, and the interpretation would be that on average all 23 species are overexploited. The most stringent target reference point would be a value of zero, if threat and extinction risk were regarded as unacceptable. However, even in the absence of fisheries exploitation, it may not be ecologically feasible to have no declining species in a community that is also affected by ecological interactions and environmental variation and change, and this target might be aspirational rather than realistic. Pragmatically, a less stringent but arbitrary target reference point of 0.25 or 0.5 could be set. These values would correspond to one-quarter (8 species) or one-half (15 species) of the suite of fishes qualifying as vulnerable (i.e., population decline rate between ≥50% and <70% over 10 years). Ultimately, the selection of the target would best be based on an assessment of the number of species that might be expected to be undergoing decline at any point in time as the result of factors other than fishing (Myers and Worm 2005).

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