Ecosystems (2006) 9: 1190–1199
DOI: 10.1007/s10021-005-0164-9

ECOSYSTEMS

Indicators of the Ecological Impact of Bottom-Trawl Disturbance on Seabed Communities

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Abstract

The Ecosystem Approach to Fisheries requires that managers take account of the environmental impacts of fishing. We develop linked state and pressure indicators that show the impact of bottom-trawling on benthic communities. The state indicator measures the proportion of an area where benthic invertebrate biomass (B) or production (P) is more than 90% of pristine benthic biomass ($B_{0.9}$) or production ($P_{0.9}$). The pressure indicator measures the proportion of the area where trawling frequency is sufficiently high to prevent reaching predicted $B_{0.9}$ or $P_{0.9}$. Time to recovery to $B_{0.9}$ and $P_{0.9}$ after trawling, depending on the habitat, was estimated using a validated size-based model of the benthic community. Based on trawling intensity in 2003, 53.5% of the southern North Sea was trawled too frequently for biomass to reach $B_{0.9}$, and 27.1% was trawled too frequently for production to reach $P_{0.9}$. As a result of bottom-trawling in 2003, in 56% of the southern North Sea benthic biomass was below $B_{0.9}$, whereas in 27% of the southern North Sea benthic production was below $P_{0.9}$. Modeled recovery times were comparable to literature estimates (2.5 to more than 6 years). The advantages of using the area with an ecological impact of trawling as a pressure indicator are that it is conceptually easy to understand, it responds quickly to changes in management action, it can be implemented at a relevant scale for fisheries management, and the necessary effort distribution data are centrally collected. One of this approach's greatest utilities, therefore, will be to communicate to policy makers and fishing enterprises the expected medium- to long-term ecological benefits that will accrue if the frequency of trawling in particular parts of fishing grounds is reduced.

Key words: production; biomass; size-based model; fisheries management; indicator; recovery; benthic invertebrates.

Introduction

Marine ecosystems cover more than 70% of the surface of the planet and provide a range of goods and services, such as nutrient cycling and fish production (Costanza and others 1997). Among the many factors that have had a negative impact on the world's oceans, fishing has undoubtedly caused the greatest ecosystem changes to date, leading to changes in energy flow through benthic ecosystems, large-scale shifts in benthic community structure, and regime shifts (for example, Jackson and others 2001; Jennings and others 2001; Myers and Worm 2003; Choi and others 2004). Fisheries research was traditionally driven by the requirement to manage single stocks of exploited species. In the last 2 decades, however, research efforts have increasingly been focused on the wider environmental global effects of fishing on nontar-

Received 11 November 2005; accepted 24 January 2006; published online 17 November 2006.
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get fauna and marine habitats (Hall 1999; Sinclair and Valdimarsson 2003); this focus is consistent with political commitments to take account of the environmental impacts of fishing in management plans (Murawski 2000; Link 2002; Pikitch and others 2004). The need to adopt and operationalize the Ecosystem Approach to Fisheries (EAF) (for example, FAO 2003; Sinclair and Valdimarsson 2003; Hall and Mainprize 2004) has prompted a wider review of the range and suitability of management indicators that might describe the state of ecosystem components or attributes and provide guidance for management decision making (Rice 2003; Rice and Rochet 2005).

It is well understood that bottom-trawling affects the biomass and production of benthic invertebrates communities (animals that live in and on the seabed) (Jennings and Kaiser 1998; Hall 1999). Trawling impacts on benthic communities have to be considered as part of any EAF, because benthic invertebrates are the major food source for many commercially exploited fish species; in addition, they provide shelter and have a significant role in supporting ecosystem processes such as bentho-pelagic coupling and nutrient cycling (Choi and others 2004; for example, Lohrer and others 2004; Widdicombe and others 2004).

Management indicators to support an EAF should provide information on the pressures affecting ecosystem components or attributes, the state of the components or attributes, and the societal response that would be required to control pressure and achieve a desirable state (for example, Garcia and Staples 2000). State indicators to support an EAF should be sensitive and responsive to fishing impacts, and it should be possible to identify a clear link between state and fishing pressure (Rice and Rochet 2005). State indicators such as the abundance of a species or group can be very difficult to measure in practice. This is because high levels of replication are usually needed to separate the signal from background noise (Nicholson and Fryer 1992; Nicholson and Jennings 2004). The slow response times of many state indicators also means that they may not provide feedback to managers on appropriately short time scales.

Short-term management guidance can be provided by pressure rather than state indicators, because these indicators respond rapidly to management action and can often be measured precisely (Nicholson and Jennings 2004; Daan 2005). A simple example of a pressure indicator for assessing trawling impacts would be the proportion of a given habitat type that is impacted by trawling in a fixed time period. This indicator could be calculated from knowledge of the spatial distribution of habitat and the spatial and temporal distribution of bottom-fishing effort. A reduction in the proportion of habitat impacted (pressure) would reduce the aggregate fishing effects on the habitat (Duplisea and others 2002), and such reductions could be adopted as a management target in data-poor circumstances (Jennings and others 2005).

However, a simple pressure indicator such as this one has two principal weaknesses. First, the indicator is not explicitly tied to any biological property of the habitat, even though habitats with a higher recovery rate will tolerate more frequent disturbance over larger areas than habitats with low recovery rates. Second, although pressure is easy to measure precisely, the state of an ecosystem component or attribute rather than the pressure is usually the focus of policy commitment—for example, to ''maintain the productivity of important and vulnerable marine and coastal areas'' (WSSD 2002). To combat these weaknesses, we need to develop indicators that account for the interaction between frequency of disturbance and capacity for recovery; we also need to link any pressure indicator to a state indicator. Although effective guidance for short-term management may be provided by the pressure indicator, the state indicator can be used to demonstrate that the expected response to a change in pressure has occurred.

We have developed related state and pressure indicators to measure the impact of bottom-trawling on benthic communities. The indicators represent the energy flow through the benthic ecosystem; thus, they are indicators of the functioning of benthic invertebrate communities and therefore the availability of food for fish. Both indicators can be used to guide fisheries management at large spatial scales. The linked indicators take account of the effects of the environment on the relationship between trawling frequency (pressure) and benthic biomass (state). The state indicator measures the proportion of an area where benthic invertebrate biomass (B) or production (P) is greater than 90% of pristine benthic biomass ($B_{0.9}$) or production ($P_{0.9}$). The pressure indicator measures the proportion of the area where trawling frequency is sufficiently high to prevent predicted B or P reaching predicted $B_{0.9}$ or $P_{0.9}$. We used a size-based model of the benthic community in the North Sea to estimate recovery time to 90% of unimpacted biomass in a range of environments, and we used widely available satellite vessel monitoring data to describe trawling frequency. The indicator is very responsive to changes in the distribution and frequency of fishing effort (pressure), as they are measured directly by the satellite vessel monitoring system, and is directly related to the state of the benthic community, as demonstrated using a previously validated size-based model (Hiddink and others 2006).

METHODS

Model of Trawling Impacts on Size-Structure

We used a size-based model of the response of soft-sediment benthic communities to trawling disturbance to predict the recovery time of these communities after trawling. Details of the development and use of the model are given in Duplisea and others (2002) and Hiddink and others (2006). In summary, the model contained 32 state variables in two faunal groups (soft- and hard-bodied macrofauna). Growth of the population biomass in each body mass – organism type compartment was modeled by modifying Lotka-Volterra competition equations to give the population biomass flux for a compartment. The interaction between habitat type and trawling effects was modeled by including relationships between growth and mortality and the environment in the model. We included the effect of sediment type on trawling mortality, the effect of bed-shear stress on population growth rate, the effects of chlorophyll a content of the sediment on carrying capacity, and the effects of sediment erosion on mortality. Recovery in the model can only take place through growth of the local populations, because the model assumes that there is no migration between adjacent areas. The model was validated by comparing the observed biomass of benthic communities at 33 stations from four habitats in the southern North Sea with modeled values. The sources of habitat data are described by Hiddink and others (2006). The model was run at the scale of 9 km^2 cells. When reporting the value of the indicators for different sediment types, we classified sediments with a gravel content of more than 5% as gravel. For sediments with less than 5% gravel, sediments with a sand–mud ratio of more than 9:1 were classified as sand. Sediments with a sand–mud ratio between 1:1 and 9:1 were classified as muddy sand, and sediments with a sand–mud ratio smaller than 1:1 were classified as mud.

Trawling frequency was calculated from European Community (EC) Satellite Vessel Monitoring System (VMS) data. From 1 January 2000 onward, all EC fishing vessels over 24 m have been required to report their location, via satellite, to monitoring centers in their flag states, at 2-h intervals. The only exception is made for vessels that undertake trips of less than 24 h or that fish exclusively within 12 nm of the shoreline. The proportion of fishing vessels less than 24 m is probably very low in the offshore areas we examined due to the cost and time associated with steaming to and from port. The VMS data do not indicate whether a vessel is fishing when it transmits positional data, but the speed of a vessel can be derived from two consecutive records. Accordingly, vessels travelling at speeds higher than 8 knots and stationary vessels were eliminated, because these vessels were assumed not to be fishing (for more details, see Dinmore and others 2003). The number of trawl passes per 9 km^2 cell per year was calculated from the number of records in a cell in the period from 1 July 2000 to 31 December 2002. For the calculation of trawling frequency (y^{-1}), it was assumed that trawlers fished at a speed of 5 knots, with a total fishing gear width of 24 m (that is, two beam trawls each of 12 m wide or one 24-m–wide otter trawl). Therefore, one record represents a trawled area of 0.449 km^2, and one record in a 9-km^2 cell over the 2½ year period therefore represents a trawling frequency of 0.0198 y^{-1}. The lower limit to the scale at which trawling effort could be evaluated was defined by the resolution of the VMS records. The 9-km^2 scale is close to the 1 × 1 nautical mile scale at which fishing effort becomes random (Rijnsdorp and others 1998). For the Dutch beam-trawling fleet, VMS records were not available for all vessels. Therefore, effort distribution as recorded by the VMS system was corrected to represent total trawling effort as recorded in logbooks by fishers (G. J. Piet unpublished).

Recovery

The benthic community in an area was assumed to be in a state of near recovery from trawling when the biomass (B) or production (P) reached 90% of predicted biomass that would occur in the absence of trawling ($B_{0.9}$ or $P_{0.9}$). This 90% value was chosen arbitrarily as a value that was close to the pristine state. This pristine value was recorded after running the model for 1500 time steps of 30 days without trawling. Then, after trawling once, the time at which the community biomass recovered to $B_{0.9}$ was predicted. We assumed that trawling frequency was sufficiently high to prevent predicted B reaching predicted $B_{0.9}$ (pressure) if the trawling frequency was lower than the inverse of the recovery time (y^{-1}) to $B_{0.9}$ or $P_{0.9}$. Thus, if recovery time to $B_{0.9}$ is 3 years, and the average frequency of trawling is once every 4 years (0.25 y^{-1}), an area is considered not to be ecologically impacted by trawling for the purposes of deriving the indicator. Biomass recovery can be modeled using the actual historical development of trawling intensity within each 9-km^2 cell of the sea.

Large-scale Effects of Trawling

The model was used to examine the effect of bottom-trawling on benthic biomass in the Dutch and UK sectors of the North Sea, in the area south of 56°N for 9-km^2 cells (Figure 1A). The areas inside the coastal 12 nautical mile zone and the plaice box (a gear restriction area) were excluded from the analysis, because many vessels fishing in these areas are not required to record their position and this would lead to underestimation of trawling intensity. Given that relatively complete VMS data for the North Sea have only been available since 2001, we assumed that trawling intensity and distribution from 1981 to 2001 was the same as in 2001 when we modeled the state of the benthic community from 2001 to 2003. For observed trawling intensity and distribution, we used the VMS data for 2002 and 2003.

Validation of Modeled Recovery Rates

To test whether the modeled recovery times were realistic, we searched the scientific literature for studies that reported trends in the abundance of benthic communities after the cessation of trawling. We used the following two approaches: (a) recovery of community biomass from large scale comparative studies and (b) recovery in numerical abundance in experimental trawling studies.

For (a), we only used the results of large-scale comparative studies, rather than specific and relatively small-scale experimental studies. Biomass abundance was reported as a proportion of unimpacted biomass, where one equals no change and 0.5 equals a 50% reduction in biomass. For (b), to assess rates of recovery in numerical abundance, we used the database of experimental trawling studies collated by Collie and others (2000) and later expanded by Kaiser and others (2006). This is the most extensive quantitative database describing the direct effects of trawling on the population size of benthic organisms. Only those studies that examined the effects of otter- and beam-trawling on gravel, mud, muddy sand, and sand habitats in the subtidal zone were included in our analysis. The magnitude of the response of abundance to fishing was calculated using the following equation for each species in each study:

$$\%\ Difference = \left(\frac{A_f - A_c}{A_c}\right) \times\ 100 \qquad (1)$$

where A_f is the mean abundance in fished plots and A_c is the mean abundance in control plots (Collie and others 2000). Data was transformed to $\ln(x + 101) - \ln(101)$, where x is the % difference. This transformation normalized the distribution of residuals and centered the data such that zero corresponds to no effect (Collie and others 2000). Time to 90% recovery of abundance ($x = -0.104$) for species in each of the four sediment types was estimated using a linear regression analysis of the transformed % difference against time.

Results

Modeled Recovery

Biomass recovery rates were highest directly after trawling, but they slowed down as biomass approached the carrying capacity (Figure 2). Production recovered faster than biomass. Within the study area, predicted recovery to $B_{0.9}$ ranged from 1 to 41 years and recovery to $P_{0.9}$ from 1–12 years (Figure 1B and C). Biomass recovery was fast in the southwestern North Sea (1–3 years), whereas recovery in the northern western North Sea was slower (typically more than 10 years for biomass). An exception was the muddy sand Oyster ground area north of the Netherlands, which recovered to $B_{0.9}$ in approximately 7 years. Recovery of production was fast in the southwestern and northeastern North Sea (less than 2 years). Recovery of production was slower in the northwestern North Sea (around 7 years) and in the oyster ground area (approximately 5 years).

Figure 3 shows the development of the production and biomass state indicators after homogenously trawling the whole study area once for each of the four sediment types. For production, both sand and gravel habitats showed fast recovery (less than 2 years) throughout most of the area, whereas a small fraction of the area recovered after about 7 years. Mud habitat recovered within 4 years. A large fraction of the muddy sand habitat recovered in about 2 years, whereas the majority of this habitat had recovered after 5 years. The areas that recovered most slowly were sand and gravel areas in the northwestern North Sea. Recovery trajectories for the biomass state indicator were similar to those for the production indicator, but recovery generally took twice as long.

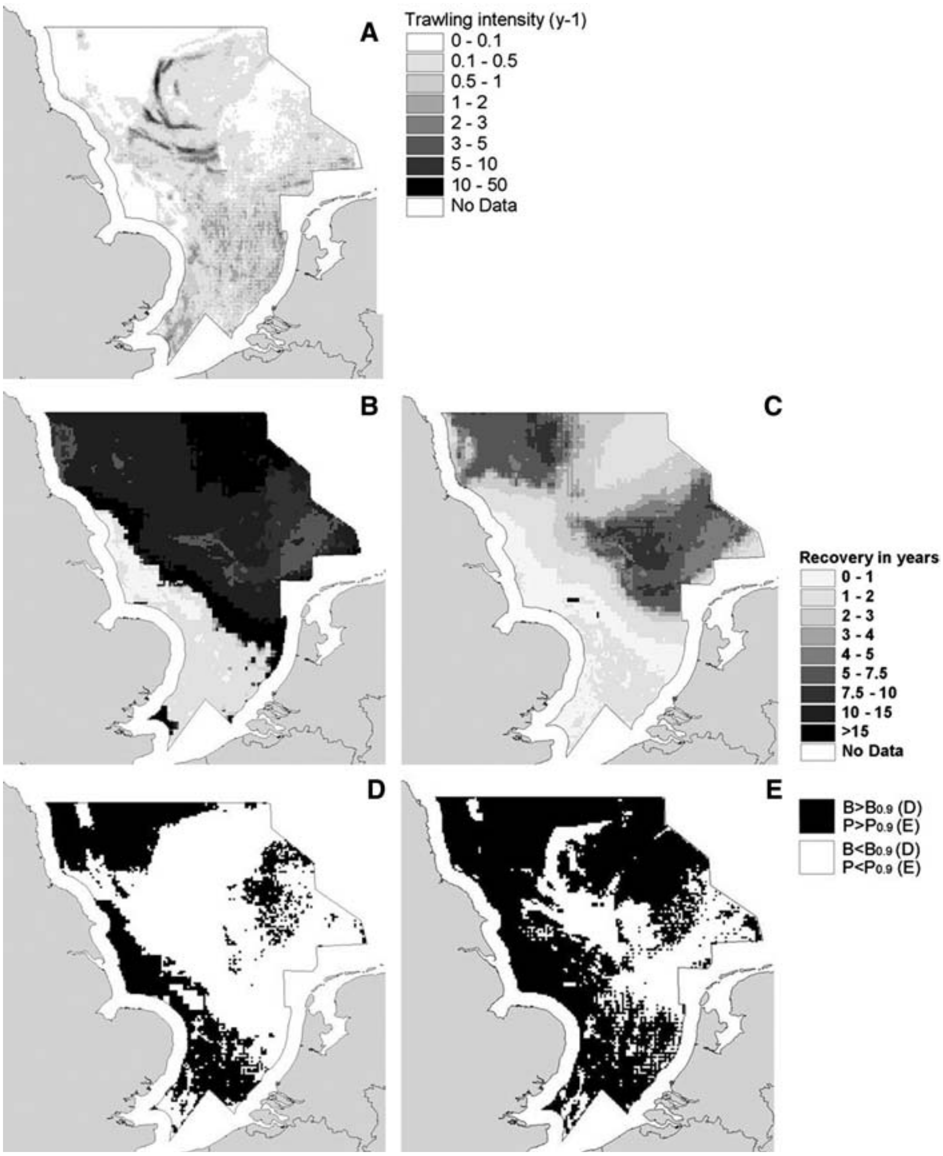

Figure 1. **A** Study area. Dutch and UK North Sea south of 56°N, as indicated by the solid line. Trawling intensity (y^{-1}), as calculated from VMS records, is indicated by gray shading. **B** Recovery time (y) of benthic community biomass (B) after a single trawl. **C** Recovery time (y) of benthic community production (P) after a single trawl. **D** Areas where trawling frequency is sufficiently low to enable predicted B to reach predicted $B_{0.9}$. **E** Areas where trawling frequency is sufficiently low to enable predicted P to reach predicted $P_{0.9}$.

Using modeled recovery times and 2003 trawling intensity data, we calculated that 53.5% of the southern North Sea had a trawling intensity that was too high for biomass to reach $B_{0.9}$ (pressure, Figure 1D). The remaining 46.5% comprised areas with low trawling intensities or high recovery rates. Within the North Sea, 27.1% of the area had trawling intensity that was too high for production to exceed $P_{0.9}$ in 2003 (Figure 1E, pressure). The area where $B > B_{0.9}$ decreased from 46.1% in 2001 to 44.2% in 2003 (state). The area where $P > P_{0.9}$ increased slightly from 72.8% in 2001 to 72.9% in 2003 (state). Thus, in 2003, in 56% of the southern North Sea benthic biomass was below $B_{0.9}$ and in 27% benthic production was below $P_{0.9}$ due to trawling.

Observed Recovery in Biomass

We consulted four published studies that describe the recovery of the biomass of benthic communities after the cessation of trawling. Blyth and others (2004) sampled seven stations on gravel or scallop-dredging grounds from 0 to 22 months after the last trawling event; they also sampled two stations that had never been trawled, which were treated as controls. Extrapolation of the recovery trajectories, as derived by linear regression, indicated that recovery of community biomass would take about 2.5 years for mobile epifauna (animals that live on the seabed and can move freely) and 3.5 years for sessile epifauna (animals that live on the seabed and do not move) (Figure 4). Hermsen and others

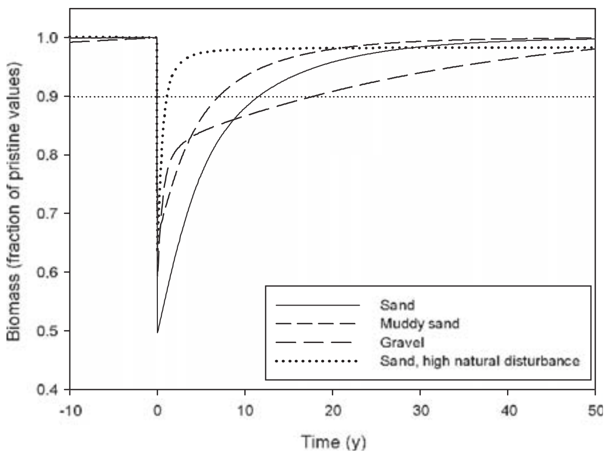

Figure 2. Trajectories of modeled recovery of benthic community biomass after a single trawling event. Different lines indicate the predicted recovery in different environments.

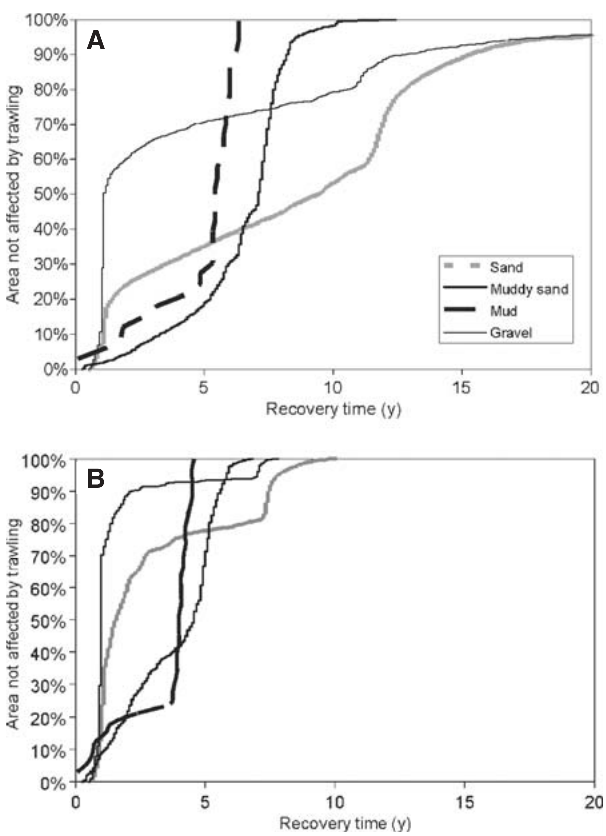

Figure 3. Development of the state indicator in the North Sea after a single trawl pass. **A** Biomass, **B** Production.

(2003) examined recovery on gravel scallop-dredging grounds. Biomass increased continuously until the last sampling event 6 years after the cessation of dredging. Unfortunately, they did not

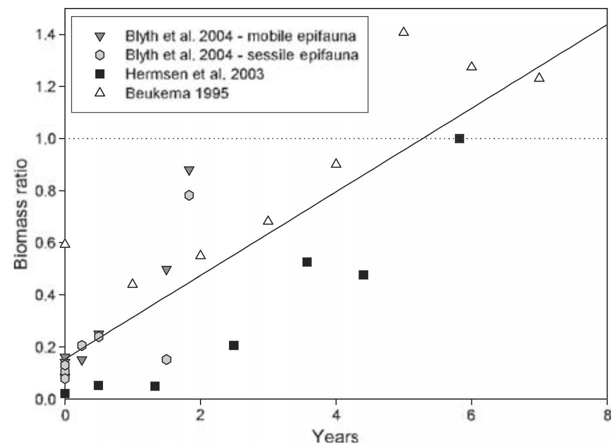

Figure 4. Recovery of the relative biomass of benthic communities after cessation of bottom-trawling. Data from Blyth and others (2004), Hermsen and others (2003), and Beukema (1995). The biomass ratio is defined as the ratio of the current biomass to the unimpacted biomass.

sample reference areas that had never been fished. We therefore had to use the 6-year sample as the reference level, and this will lead to an overestimate of recovery rates. Recovery to long-term mean biomass took 5 years after intertidal lugworm-dredging, relative to the average biomass in the same area in the 8 years before lugworm-dredging (Beukema 1995). Hoffmann and Dolmer (2000) found no effect of mussel-dredging on the biomass of the epibenthic community. Because there was a simultaneous large-scale temporal decrease in benthic biomass, no suitable reference level was available and their data were not included in Figure 4. Combining data for the first three studies, recovery to 100% of pristine biomass took approximately 5 years (regression through all data, $R^2 = 0.71$, $F_{1,28} = 72.5$, $P < 0.0001$).

Observed Recovery in Numerical Abundance

Recovery times based on the numerical abundance of benthic species ranged from 25 days for muddy sediments to no recovery on gravel (Table 1).

Discussion

The proposed state and pressure indicators of the impact of bottom-trawling on benthic communities can be used to guide management at large spatial scales and are thus applicable to an EAF where the principal management unit is the ecosystem or a related sociopolitical construct. The indicators

Table 1. Recovery Time of Benthic Species Densities

Sediment Type	Days to 90% Recovery
Muddy sand	193
Mud	25
Gravel	No recovery
Sand	111

Calculated from Collie and others (2000) and Kaiser and others (forthcoming).

measure the proportion of an area where predicted benthic biomass or production is greater than predicted biomass, or production at 90% of carrying capacity (state), and the proportion of the area where trawling frequency is sufficiently high to prevent predicted biomass or production from reaching the predicted $B_{0.9}$ or $P_{0.9}$ (pressure).

The pressure indicator is responsive to short-term changes in the distribution and frequency of fishing effort, whereas the state indicator shows the effects of changes in fishing pressure on the benthic community. In practice, predictions about state that are made on the basis of modeled links with pressure would need to be substantiated by regular sampling of benthic communities at locations subject to different levels of fishing pressure in different habitats. The methods for conducting such validation are already established (Hiddink and others 2006). The benthic biomass predicted by the model correlated positively with the observed biomass at 33 stations over a range of trawling intensities and habitats in a large area of the North Sea and explained 39% of the observed variation. The model is holistic; and it is not based on a detailed consideration of the critical but poorly quantified processes that underpin productivity and biomass accumulation, such as recruitment and immigration. As a result, recovery from trawling may have been underestimated by the model. Unfortunately, we found very few empirical studies of recovery after the cessation of trawling and had to rely on studies of recovery in areas that we would not expect to be truly representative of North Sea fishing grounds, including one intertidal study. However, our modeled recovery times are consistent with the few empirical data available, and they are in the same range as recovery times modeled by Pitcher and others (2000) for benthic invertebrates using a simple logistic growth model.

Modeling of the biomass and production of benthic communities at a large scale is inevitably ambitious, so we had recourse to many simplifications and generalizations (Hiddink and others 2006). The model does not take account of year-to-year variations in environmental attributes, such as variations in primary productivity, and possible differences in patchiness in impacted versus non-impacted areas. These processes may affect the recovery trajectories and the ratio of recovery through immigration versus recruitment in real benthic ecosystems (Auster and Langton 1999). We have to accept these shortcomings if research on the large-scale distribution patterns of communities—and in particular, on the large-scale effects of trawling—in marine ecosystems is to progress at the speed required to support management decision making.

Recovery times that are estimated from small-scale experimental trawling studies are very short, in particular for mud bottoms. These estimates seem rather optimistic; a comparison with recovery from trawling in closed areas indicates that such values are unlikely to be representative for recovery of biomass on fishing grounds. In densely trawled fishing grounds, a source of immigrants cannot be relied on; moreover, it is simply local redistribution of biomass, not an increase of biomass. Thus, it appears that the reports of recovery in small-scale experimental studies have been overestimated because this type of experimental have immigration rates that are not realistic for real fishing grounds.

No data from sediment-mining experiments have been used to parameterize our recovery model. Because the recovery of benthic communities after disturbance by marine sediment extraction has been studied more extensively than recovery after bottom-fishing disturbance, sediment-mining data could be useful for comparison with modeled recovery rates. Sediment extraction is different from trawling in that the animals and their habitat are both removed together; thus, recovery must begin with recruitment and immigration. Studies of recovery after sediment extraction show that recovery of numerical abundance was much more rapid than recovery of biomass (Newell and others 2004). Available evidence suggests that substantial recovery of biomass would be expected within 2–8 years of the cessation of dredging (Newell and others 1998; van Dalfsen and others 2000; Boyd and others 2003). This time frame is broadly consistent with the recovery times predicted by our model of recovery after trawling in many areas of the North Sea.

The recovery of biomass and/or production is a simple measure of the consequences of the cessation of human impacts on benthic communities, and the measure of recovery does not account for variation in the life histories of component species. Thus, the recovery of species with low intrinsic rates of increase and slow growth will not have occurred in the times predicted, even if community biomass and production are close to the levels expected in the absence of impact. For example, larger bivalves such as scallops may continue to increase in biomass for a decade or more after the cessation of scallop dredging (Murawski and others 2004; Beukers-Stewart and others 2005). Recovery rates of the most vulnerable species could be measured in terms of the return to the expected age composition in the absence of fishing mortality. Because many benthic species reach ages of over 10 years, and some vulnerable species, such as *Arctica islandica*, can reach ages of over 100 years (MARLIN 2003), recovery times thus measured would be substantial. We would justify the use of indicators of recovery that are based on biomass and production by pointing out these attributes determine how much energy is exchanged with higher and lower trophic levels and that benthic communities have a function in the food web that sustains fish production. The generality of recovery measures based on biomass and production also means that they can be applied across very broad spatial scales, whereas indicators for the recovery of all individual species would require detailed and costly local understanding of interactions between the species and their biological and physical environment. Moreover, it is much easier to predict the recovery dynamics of individual species and to validate the recovery trajectories than it is to predict the recovery of higher-level community attributes that are principally governed by energetic constraints, scaling laws, and the environment (for example, Brown and West 2000).

Modeling the effects of pressure to predict the state over large spatial scales reduces the requirements for the direct monitoring of state. Thus, the indicator can be used to judge the short-term progress of management actions (year on year) toward expected targets for state. Simultaneously, empirical measurements of state could be made at selected locations over longer time scales to confirm that changes in trawling effort, as predicted by the pressure indicator, had in fact led to the predicted response in state.

One property of the proposed indicator is that it does not reflect spatial differences in the potential production or biomass of benthic communities. Thus, recovery of an area with a high unimpacted biomass is weighed in the same way as recovery in an area with a very low unimpacted biomass. We chose to relate state to the unimpacted state because it would enable comparison of the magnitude of human impacts over a range of different environments. However, if necessary, state and pressure indicators for the total biomass or production in an area could be developed with the same model, or the indicators could be reported for areas of habitat grouped by unimpacted biomass and/or recovery time.

State and pressure indicators are usually used in conjunction with reference points or directions to guide management decision making. At the simplest level, the proposed indicators can be used (a) to describe the predicted effects of changes in fishery management actions on benthic communities and (b) to assess whether observed changes in fishery management actions have had positive or negative effects. In the first case, the effects of different fishery management actions on the spatial distribution of trawling effort would be predicted using a suitable model, such as the random utility model of Hutton and others (2004), and effort predictions would be used to calculate indicator values associated with each management action. This information would then be used to identify the actions that had the most acceptable impact on benthic communities. In the second case, values of the indicator would be calculated from data on the observed effort, as collected using VMS, and trends in the indicator would show the extent to which changes in patterns of fishing effort had an ecological impact on benthic communities. The selection of reference points or directions is principally a matter of societal choice, but scientists can act as advisors on the consequences of selecting different values. Limit reference points for the indicators we propose could be based on the minimum amount of production required to sustain specific ecosystem functions, such as the provision of food for fish communities, with precautionary limits set to take account of measurement, process, model, and estimation error. Alternatively, they might be set to take account of political commitments to sustain some proportion of biomass in a relatively undisturbed state. Target reference points might be set to maximize the amount of benthic production available to support fish production. The development of any of these reference points will require significant additional research and a dialogue with policy makers and interest groups.

ACKNOWLEDGEMENTS

This study was funded by the UK Department of Environment, Food and Rural Affairs (DEFRA) (contract MF0731). The VMS data were provided by DEFRA in a raw, uninterpreted form, and we thank Paul Eastwood and Craig Mills (CEFAS) for their assistance. We are grateful to Gerjan Piet (RIVO) and Rob Blyth-Skyrme for supplying VMS and recovery data, respectively. We thank the two anonymous referees and Terry Done for comments that improved an earlier version of this paper.

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