

Response of potential fish community indicators to fishing

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Fish community metrics and diversity indices have often been proposed as indicators to support an Ecosystem Approach to Fisheries Management. The response of eight fish community indicators to spatio-temporal changes in fishing effort was studied by comparing the rate of change in (i) areas subject to different levels (high, medium, and low) of fishing effort and (ii) areas inside and outside a closed area (the plaice box) where management measures led to changes in fishing effort over time. Several indicators showed broadly consistent responses to fishing effort (slope of the biomass-size spectra, mean weight, and mean maximum length) while a range of biodiversity indices and biomass did not. The response of the indicators to spatio-temporal changes in fishing effort showed that, with the exception of the slope of the biomass-size spectra, none of the indicators reliably detected the effects of spatial management measures at the scales of time and space that were used for these analyses. While some of the size-based indicators we tested did provide evidence for fishing impacts on communities, the response of the indicators to fishing was not straightforward and may have depended on environmental conditions and historic fishing regimes. Therefore, the indicators would provide limited support for assessing the effects of short-term and small-scale management actions and must be applied with caution until we have an improved theoretical understanding of their response to fishing and the environment.

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Introduction

Indicators of the effects of fishing on marine communities are required to support an ecosystem approach to fisheries management (EAFM; Link, 2002; Rice, 2003). Many governments and regulatory bodies have already agreed to implement an EAFM but, with the exception of indicators for commercially exploited fish stocks (Smith *et al.*, 1993; Garcia and Staples, 2000), the development and testing of indicators is not well advanced (Rice, 2000; ICES, 2001; Rochet and Trenkel, 2003).

In 2002 the North Sea ministers agreed to implement an ecosystem approach and to use indicators to set management objectives (termed Ecological Quality Objectives, EcoQOs; Lanfers *et al.*, 1999) and to assess ecosystem health. ICES (2001) has identified the ideal properties of indicators that can be used to monitor and manage the effects of fishing. The indicators should be: (i) relatively

easy to understand by non-scientists and those who will decide on their use, (ii) based on an existing body or time-series of data to allow a realistic setting of objectives, (iii) measurable over a large proportion of the area in which the indicator is likely to be used, (iv) easily and accurately measured, with a low error rate, (v) sensitive to a manageable human activity (fishing) and responsive primarily to that activity, with low responsiveness to other causes of change, and (vi) relatively tightly linked in space and time to that activity. We have to accept that indicators may not have all these properties, but should seek to identify those that most closely meet theoretical requirements.

While other studies (e.g. Trenkel and Rochet, 2003; Nicholson and Jennings, 2004) mainly addressed the first four properties, we focus on the latter two properties and test the response of a set of metrics to spatial differences and spatio-temporal changes in fishing effort. The following metrics: the slope of the biomass-size spectra, mean weight

of an individual fish, trophic level, mean maximum length, the community diversity indices Hill's N_0 , Hill's N_1 , and Hill's N_2 (Hill, 1973), and the total biomass, are thought to describe the major structural and functional attributes of the fish community. Moreover, they can realistically be measured over large areas and on large time scales. For at least some of the indicators there is good theoretical and empirical evidence that they should be affected by fishing (Rice and Gislason, 1996; ICES, 2001; Daan *et al.*, 2003; Rochet and Trenkel, 2003).

Fishing has a direct effect on the structure and function of communities because it reduces mean body size within populations (Beverton and Holt, 1957) and leads to an increase in the relative abundance of smaller species (Jennings *et al.*, 1999b). Fishing may also have indirect effects because small species may proliferate when their larger predators are reduced (Gislason and Rice, 1998; Daan *et al.*, 2003; Dulvy *et al.*, 2004) and selection for specific life-history traits may occur in response to size-selective fishing (Law, 2000). The aggregate responses of communities to fishing have often been described using the slopes of size spectra and plots of abundance or biomass by body size class (Rice and Gislason, 1996). Differential vulnerability of species to fishing may also lead to changes in the distribution of life-history traits (Jennings *et al.*, 1999b) and diversity (Greenstreet and Hall, 1996) of the community.

While many aspects of diversity have ecological significance, studies of the effects of fishing have largely focused on species diversity. There have been many attempts to describe species diversity using a single statistic that is amenable to statistical analysis. These statistics usually encapsulate species richness and/or the distribution of individuals among species. Diversity indices range from counts of the total number of species recorded (species richness) to statistics that indicate both richness and the way in which the total number of individuals is distributed among the total number of species (evenness or equitability). The Hill (1973) numbers (N_0 , N_1 , and N_2) have frequently been used as univariate measures of diversity because they cover this range.

Changes in the size and species composition of fish communities are linked to changes in trophic structure, because larger individuals generally, though not exclusively, feed at higher trophic levels. Trophic levels have been estimated for many species using stable isotope analysis (Owens, 1987), diet analysis, and trophic models (Christensen and Pauly, 1993). Such estimates have been applied to species-abundance data to show that the trophic level of fish communities and landings often declines with increasing fishing pressure (Yang, 1982; Pauly *et al.*, 1998; Pinnegar *et al.*, 2002). However, changes in body size are likely to be a stronger and more universal indicator of fishing effects than changes in trophic level, because changes in trophic structure can become decoupled from changes in size structure (Jennings *et al.*, 2002).

Biomass is an important indicator of community abundance (Rochet and Trenkel, 2003). Trends in biomass will not be directly related to trends in production because communities dominated by smaller individuals have higher production: biomass ratios. Therefore, biomass is expected to show greater sensitivity to fishing than production (e.g. Ware, 2000).

In this paper, we compare the values and trends of potential indicators in areas that have been differentially exploited by the fishery to determine their sensitivity to fishing. We also compare trends in areas and periods that were subject to major changes in fishing effort after the closure of a protected area ("plaice box") to determine if the response of the indicators is tightly linked in space and time to the changes in fishing effort.

Methods

Data

Data from two bottom-trawl surveys, the International Bottom Trawl Survey (IBTS) and the Beam Trawl Survey (BTS), were used.

The IBTS survey covers the whole North Sea, Skagerrak, and Kattegat, within the 200 m isopleth. We have only used first quarter data from 1980 onwards, the year when the same survey gear, a GOV-trawl (Grande Ouverture Verticale), was adopted by all participating nations, excluding Skagerrak and Kattegat. For gear specifications see ICES (1999).

The BTS was initiated in 1985 to estimate the abundance of the dominant age groups of plaice and sole including pre-recruits. The survey is carried out in the south-eastern North Sea and uses a pair of 8 m beam trawls rigged with nets of 120 mm and 80 mm stretched mesh in the body and 40 mm stretched mesh codends. A total of eight tickler chains are used, four mounted between the shoes and four from the groundrope. The survey was designed to take between one and three hauls per ICES rectangle (boxes of 0.5° latitude by 1° longitude). The stations are allocated over the fishable area of the rectangle on a "pseudo-random" basis to ensure that there is a reasonable spread within each rectangle. No attempt is made to return to the same tow positions each year. Towing speed is 4 knots for a tow duration of 30 min and fishing occurs during daylight only. Only data collected with RV "Isis" were used to ensure maximum compatibility within the data set.

When using data from (internationally coordinated) research vessel surveys there is a risk of species mis-identification (Daan, 2001). Here, the data have been used at face value and this may affect the calculated values of the diversity indices. Fish were measured and identified to species on the surveys. Length was converted to weight using published length–weight relationships (Bedford *et al.*, 1986; Coull *et al.*, 1989).

Indicators

The following community indicators were calculated by survey year: (i) the slope of the biomass-size spectra, (ii) mean weight of an individual fish, (iii) trophic level, (iv) mean maximum length, (v) Hill's N_0 , (vi) Hill's N_1 , (vii) Hill's N_2 , and (viii) total biomass. The value of the indicator was calculated as the mean over all hauls in that year. All analyses were restricted to species that belonged to the demersal fish assemblage.

To determine the slope of the biomass-size spectrum, fish >4 g were assigned to \log_2 body mass classes, and cumulative biomass by \log_2 body mass was calculated per haul. Biomass-size spectra were normalized by dividing the biomass in a given body mass class interval by the width of that class interval. The relationship between body mass class and total normalized biomass in the body mass class (i.e. numbers in the class) was determined per haul using least-squares linear regression.

Mean weight was calculated per haul as:

$$\bar{W} = \frac{\sum_k W_k}{N}$$

where W is the body mass of an individual and N is the total number of individuals.

Mean maximum length, which is indicative of the composition of the fish community in terms of life-history types, was calculated per haul as:

$$\bar{L}_{\max} = \frac{\sum_j (L_{\max j} N_j)}{N}$$

where $L_{\max j}$ is the maximum length obtained by species j , N_j is the number of individuals of species j , and N is the total number of individuals.

The trophic levels of individuals were estimated from their length, using relationships between length and trophic level as determined by nitrogen stable isotope analysis (Jennings *et al.*, 2002). Trophic level vs. length relationships were only available for 31 species, but these accounted for more than 90% of the total weight of fish caught. The mean trophic level (TL) was calculated per haul as:

$$\bar{TL} = \frac{\sum_i \sum_j (TL_{ij} \times W_{ij})}{\sum_i \sum_j (W_{ij})}$$

where W_{ij} and T_{ij} are, respectively, the mass and trophic level of species i in length class j .

The three indices defined by Hill (1973) were used as diversity metrics: N_0 (species richness), N_1 (exponential of Shannon–Wiener's diversity index, effectively the number of abundant species), and N_2 (reciprocal of Simpson's

diversity index, effectively the number of very abundant species). An important aspect of these diversity indices is that they do not convey any information on the identity of species in the community. Values of the metrics were determined per haul.

Analyses

Time-series for each community metric were generated for IBTS and BTS. Rates of change were described using least-squares linear regression, while the significance of trends was determined using a non-parametric approach (Sen, 1968). Since no long-term international effort data exist for the North Sea we used two approaches to explore the relationship between each metric and (changes in) fishing effort: first, to compare trends in areas where management measures resulted in marked changes in effort over time; second, to use the information on the spatial distribution of international otter and beam trawling effort by ICES rectangle to distinguish between high, medium, and low effort areas, and to compare the indicators in these areas.

In the first analysis we used the IBTS data to examine differences in the trends in indicator values in areas subject to different intensities of fishing. Since depth explains much of the spatial variation in the composition of North Sea fish communities, we distinguished two study areas based on depth: a southern area with depth <60 m and a deeper (>60 m), mostly northern area. Within both areas three suites of rectangles or “treatments” were distinguished based on the distribution of international otter and beam trawling effort by ICES rectangle in 1998 (Jennings *et al.*, 1999a; Callaway *et al.*, 2002): low effort, intermediate effort, and high effort (Table 1 and Figure 1). The boundaries were chosen so as to distribute effort categories evenly among rectangles. The effort ranges differed in the two depth categories because notably beam trawl effort in the southern areas was markedly higher. This approach assumes that the spatial distribution of effort over time is relatively constant, in accordance with the observations by Jennings *et al.* (1999a) and Callaway *et al.* (2002).

In the second analysis, we examined the responses of indicators to the imposition of a (semi)closed area: the plaice box. The plaice box was established to reduce the discarding of plaice in the nursery grounds along the continental coast of the North Sea, an area between 53°N and 57°N (Figure 2). It was closed to trawlers with engine power >300 hp in the second and third quarter in 1989, in the second to fourth quarters in 1994, and for the whole year since 1995. These measures resulted in a marked shift of fishing effort to offshore areas (Pastoors *et al.*, 2000). We used the BTS data to compare indicators in two areas: inside and outside the plaice box and in three periods: (1985–1989) high effort inside the box, (1990–1994) medium effort, and (1995–2001) low effort. For each metric the mean value and mean rate of change over time was determined per area/period. Spearman's rank order correlation was used to test if the trend was significant.

Table 1. Characteristics of the six IBTS areas used to assess the impacts of fishing on indicators for the North Sea fish community.

	SL	SM	SH	DL	DM	DH
Area	Shallow	Shallow	Shallow	Deep	Deep	Deep
Effort	Low	Medium	High	Low	Medium	High
Depth range	10–60	10–60	10–60	> 60	> 60	> 60
Effort range (*1000)	< 10	10–25	≥ 25	< 5	5–12.5	≥ 12.5
Number of hauls	123	294	216	277	456	311
Average effort (h/rectangle)	5 178	17 275	34 356	2 787	8 831	19 531
Beam trawl (%)	47	53	63	14	4	7
Otter trawl (%)	53	47	37	86	96	93
Mean depth (m)	38	34	35	93	134	112
Number of rectangles	20	32	24	23	33	30

Results

Variation and trends

When calculated for the whole survey area and time-series, the indicators, mean weight and mean maximum length, showed the same significant trends over time for the IBTS and BTS (Table 2; Figures 3 and 4). Trophic level only decreased significantly, and biomass only increased significantly, when calculated from the IBTS data. The diversity metrics Hill's N_1 and N_2 increased significantly and the slope of the biomass-size spectrum decreased significantly only when calculated from BTS data. There was no significant trend in Hill's N_0 in either survey.

Responsiveness to different levels of fishing effort

The responsiveness of the indicators to different levels of fishing effort was studied in two areas (see Figure 1) using the IBTS data. In the shallow areas a distinct relationship between several of these indicators and fishing effort was observed which often was not apparent in the deep areas (Figure 5). Most indicators showed a gradual change in trend as effort increased in the shallow areas. This was not apparent in the deeper areas. For total biomass a gradual change in the deeper areas was observed which was not apparent in the shallow areas. Hill's N_0 showed inverse patterns in the deep and shallow areas.

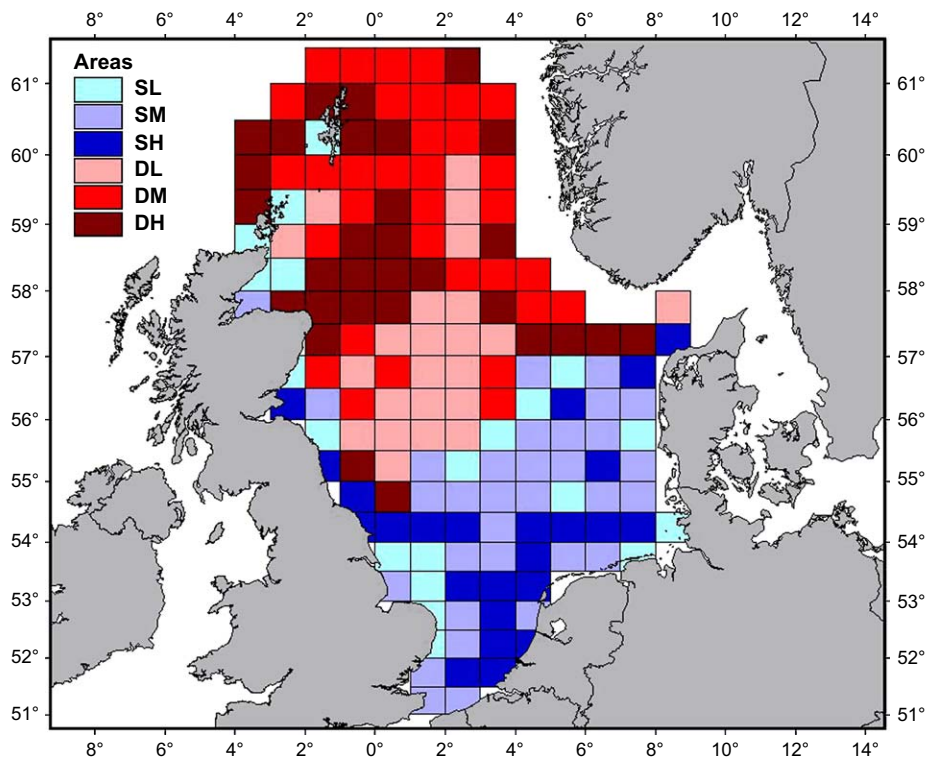


Figure 1. Location of six suites of ICES rectangles that are distinguished based on depth and fishing effort. For selection criteria see Table 1.

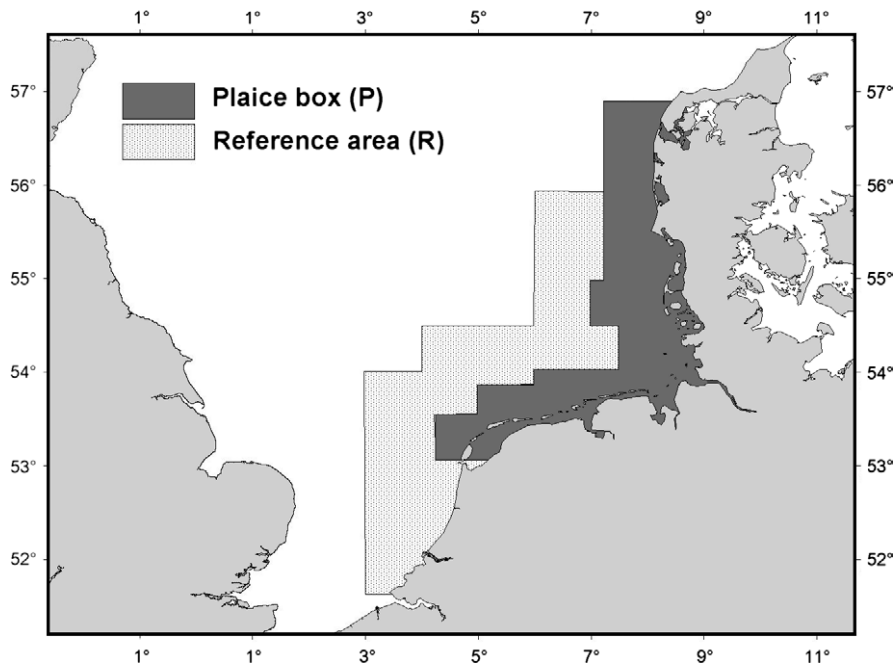


Figure 2. Location of the Plaice box (B) and Reference area (R) indicators.

If these correlation analyses reflect a causal relationship then a reduction in fishing effort will result in a slower decrease or even increase of the slope of the biomass-size spectra, mean weight, and mean maximum length while the trophic level and Hill's diversity indices N_1 and N_2 show the opposite effect.

Responsiveness to spatio-temporal changes in fishing effort

The responsiveness of the indicators to spatio-temporal changes in fishing effort could be studied using the BTS

Table 2. Slope of the time-series based on two surveys (BTS and IBTS). For the IBTS different subsets based on depth are distinguished. Significant slopes (Spearman $p \leq 0.05$) are in bold.

Indicator	IBTS			BTS
	All	Shallow	Deep	
Slope biomass-size spectrum	-0.0019	0.0015	-0.0023	-0.0030
Mean weight (g)	-6.34	-4.43	-6.45	-1.95
Trophic level ($\delta^{15}\text{N}$)	-0.039	-0.032	-0.028	-0.027
Mean maximum length (cm)	-0.35	-0.55	-0.41	-0.23
Hill's N_0	-0.26	-0.64	-0.13	0.25
Hill's N_1	-0.006	-0.005	0.058	0.141
Hill's N_2	-0.019	-0.002	0.038	0.094
Biomass (kg haul^{-1})	1.13	0.25	1.47	-0.23

catches in the south-eastern North Sea covering an area where a significant displacement of effort took place after (partial) closure of the plaice box.

For most indicators similar slopes in the trend were observed in the box and reference areas during the subsequent periods (Figure 6). Several indicators (i.e. slope of the biomass-size spectra, mean weight, trophic level, mean maximum weight) show a downward trend (negative slope) in the box area during the first period that becomes less steep in the subsequent (lower effort) periods. However, a similar pattern is observed in the reference area in spite of the assumed increase of effort in that area. The difference between the box and reference area, however, does show interpretable patterns (i.e. a gradual change in the slope in the subsequent periods) for indicators such as the slope of the biomass-size spectra, Hill's N_2 , or the biomass per haul.

Discussion

Results of two bottom-trawl surveys in the North Sea show significant downward trends in mean weight and mean maximum weight. The trends in the slope of the biomass-size spectra and trophic level (down) and the biomass and biodiversity indices Hill's N_1 and N_2 (up) are only significant in one survey. The significant trends show the same direction of change for both surveys. If these trends are caused by fishing this implies that an increase in fishing will result in a (further) decrease of the slope of the

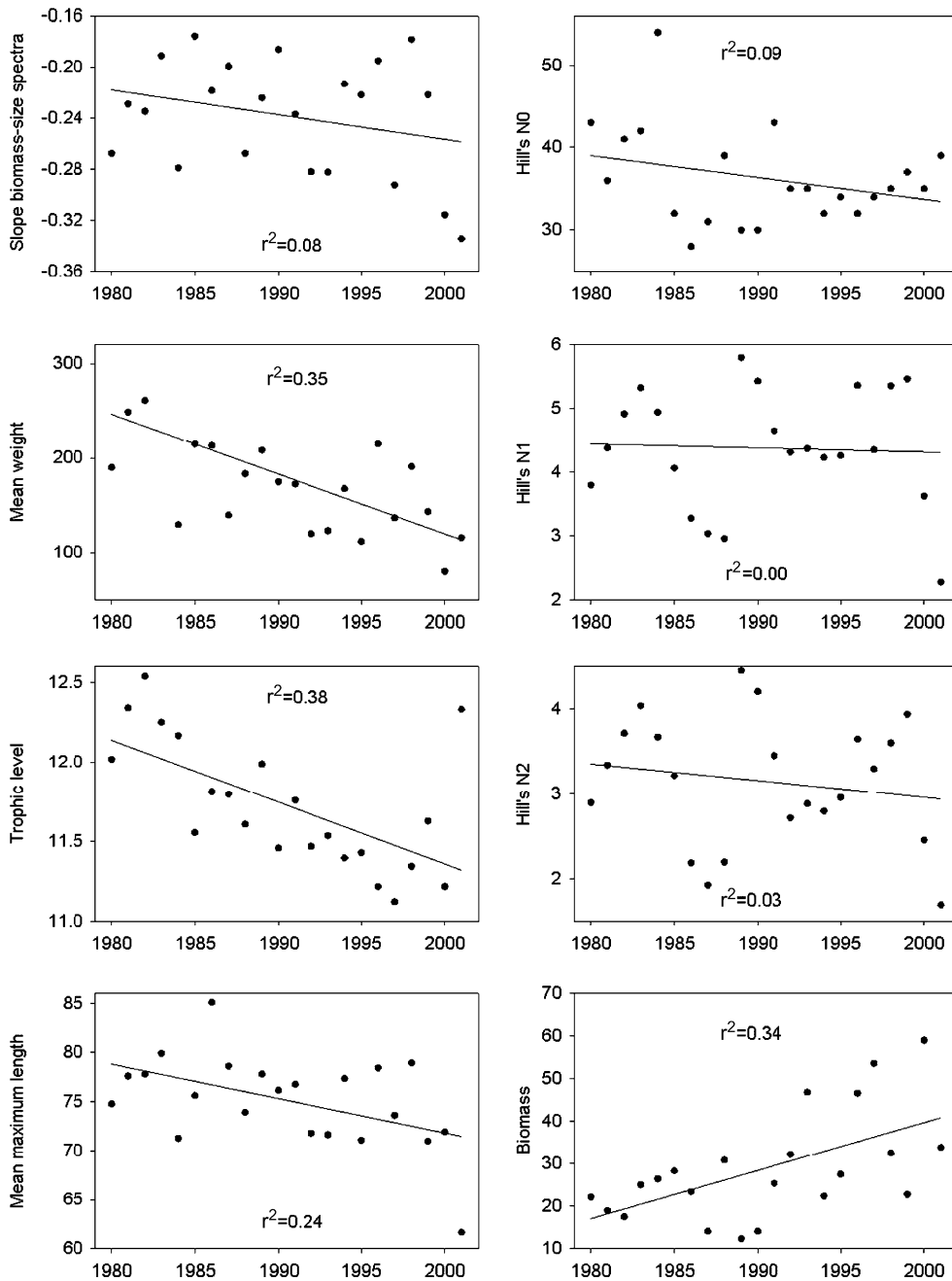


Figure 3. Time-series and trends of eight indicators calculated from the IBTS data.

biomass-size spectrum, mean weight, trophic level, and mean maximum length and an increase in the biomass and biodiversity indices Hill's N_1 and N_2 . A reduction in fishing effort should result in the opposite.

For the analyses aimed at identifying the relationship between fishing effort and fish community indicators, the assumption of an integrated response of the assemblage implies that those indicators that are expected to decrease

as a result of fishing will show the strongest decline in the slope of the trend in the high effort areas. In the shallow area this is observed for most of the indicators (slope of the biomass-size spectrum, mean weight, mean maximum length, and the biodiversity indices). However, in the deep area we did not observe a relationship with fishing effort, as there was no gradual change of slope with a change in effort. Also remarkable was that for several indicators the

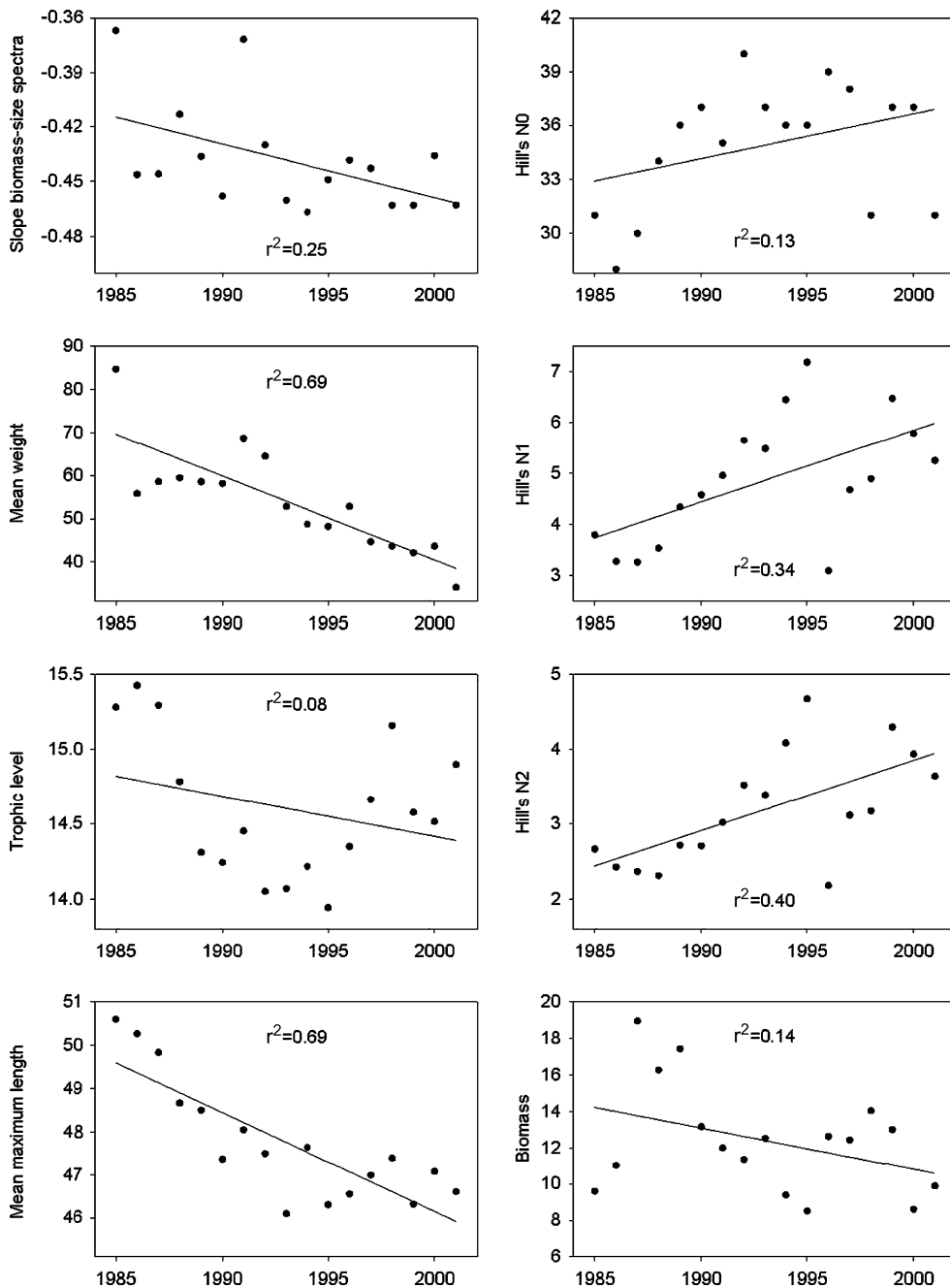


Figure 4. Time-series and trends of eight indicators calculated from the BTS data.

slope in the deep area resembled that in the high effort shallow areas, despite the fact that most of the “high” effort in the deep area is equivalent to the “low” effort in the shallow area (Table 1). This shows that, in general, there is no straightforward linear relationship between fishing effort and the indicators.

For the biodiversity indices, even the cause–effect relationship, as well as the direction of the effect of fishing

on these metrics, is unclear, consistent with the expectations of Rice (2000) and Bianchi *et al.* (2000). Fishers may select areas with relatively high abundance of few commercial species and through exploitation of these stocks increase evenness and thus biodiversity. Or, in case of a community with high evenness, the least abundant species may be depleted to the point where they do not occur in the survey catches, which results in a decrease of species richness and

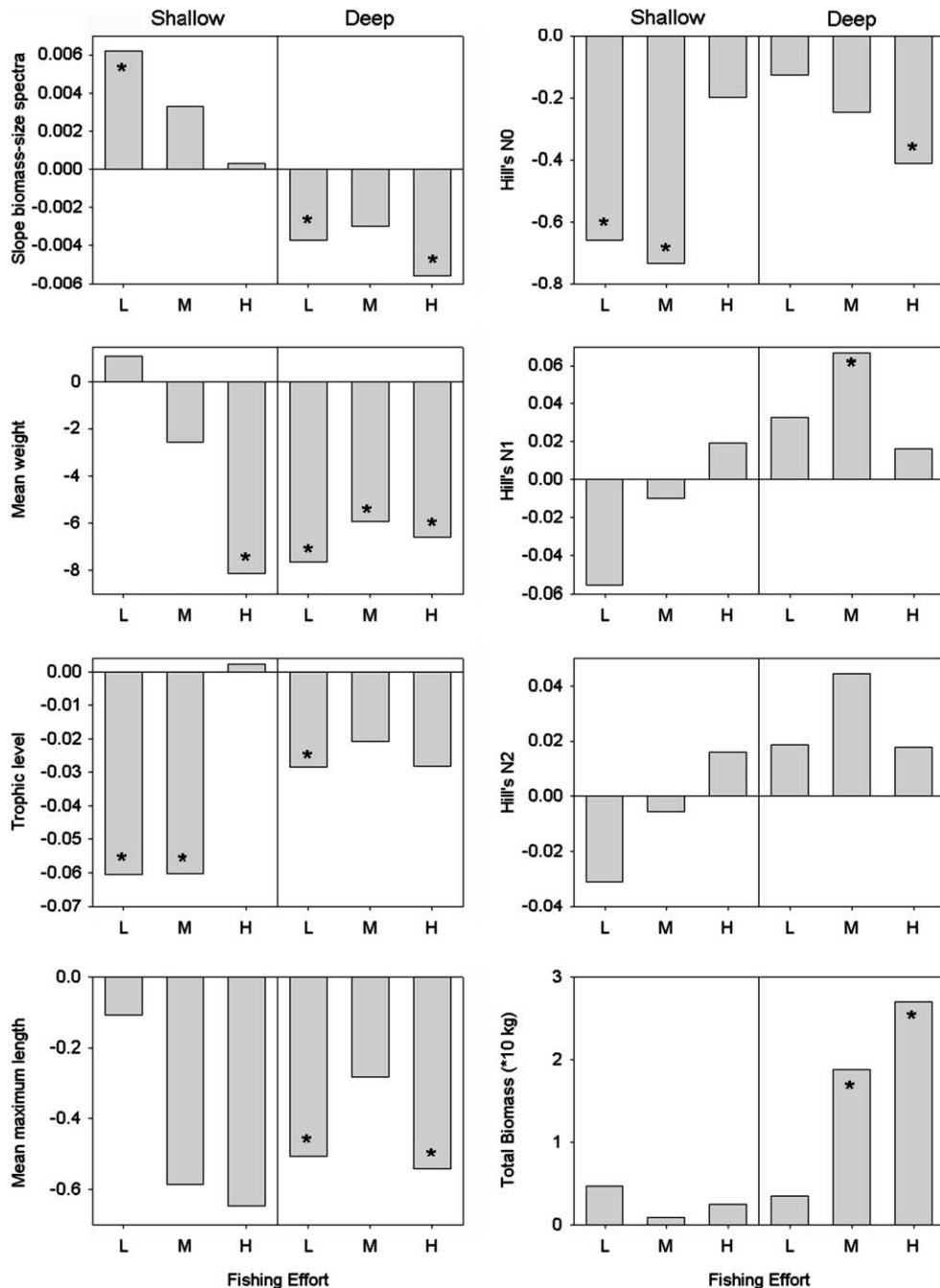


Figure 5. Slope of the trend of eight indicators in six suites of ICES rectangles based on the IBTS. Two areas are distinguished: shallow and deep, L = low effort, M = medium effort, and H = high effort (see Table 1 and Figure 1).

thus biodiversity. In previous studies, both positive and negative responses of community diversity to fishing have been found (Greenstreet and Hall, 1996; Rogers and Ellis, 2000). These contradictions are also observed in this study where Hills diversity indices for different surveys, areas, and analyses respond inconsistently to fishing (Table 2,

Figures 3–5). Community biomass did not appear to be a useful indicator of fishing. This is perhaps not surprising when biomass falls most rapidly when fishing is initiated and the greatest reductions in the overall biomass of the North Sea fish community probably took place well before the start of the surveys used in this analysis (Greenstreet and

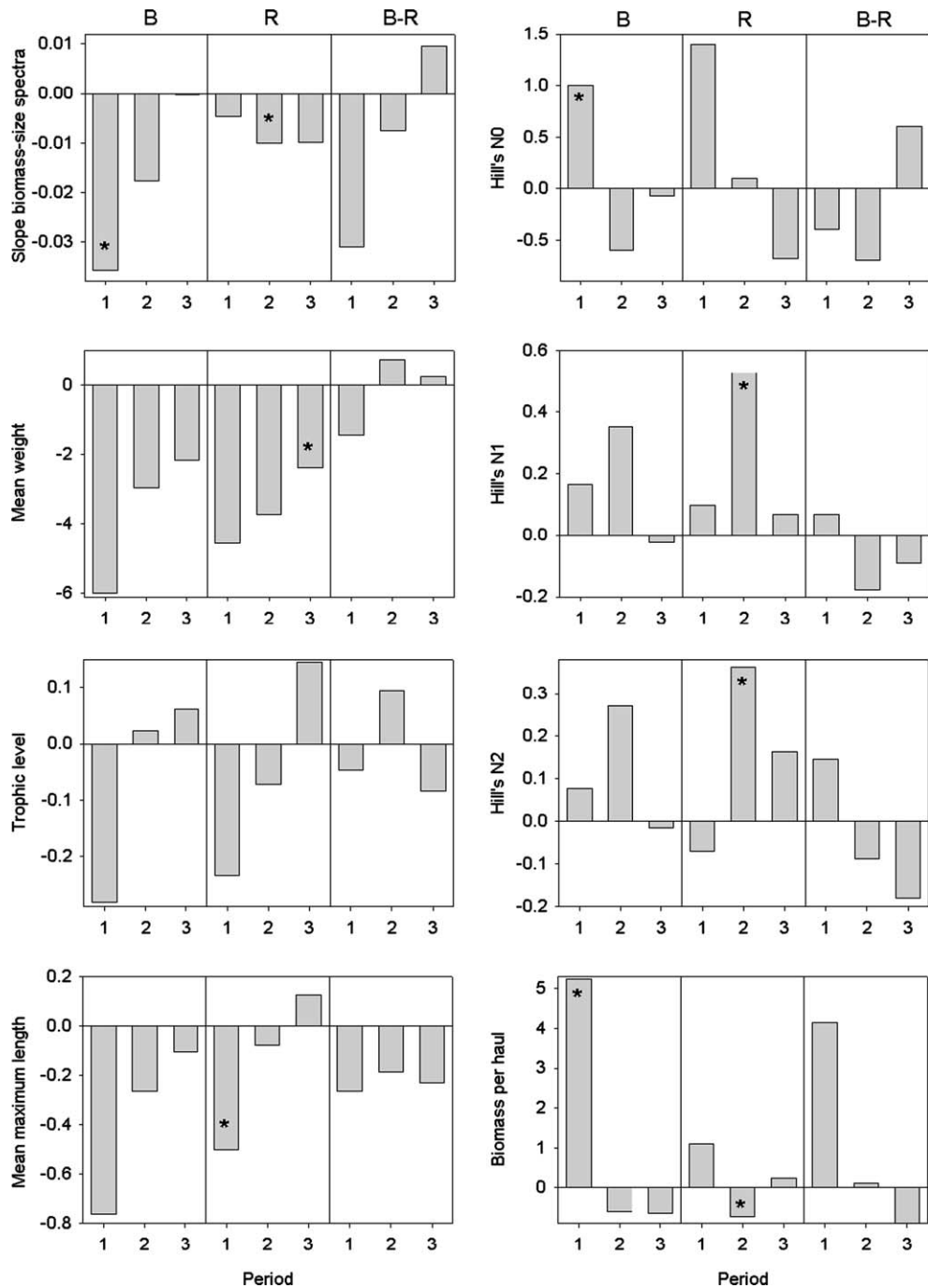


Figure 6. Slope of the trend of eight metrics in six area/period subsets based on the BTS. Codes: B = Box (closed) area, R = reference (open) area; 1 (Box open, High effort in Box) = 1985–1989, 2 (Partial closure, medium effort in Box) = 1990–1994, and 3 (Year-round closure, Low effort in Box) = 1995–2001. B–R indicates the difference in trend between the box area (B) and the reference area (R) (= $\text{Slope}_B - \text{Slope}_R$). Asterisks indicate that the calculated slopes were significantly different from zero (Spearman $p < 0.05$).

Hall, 1996). Moreover, there has been compensation in the North Sea foodweb as a consequence of exploitation and, since the late 1970s, the biomass of larger individuals and species has fallen while the biomass of smaller individuals

and species has increased (Daan *et al.*, 2003). This compensation would limit the apparent effects of fishing on total biomass, but would lead to decreases in size-based indicators such as mean size, mean maximum size, and

slope of the size spectrum, because the decrease in large fish and the increase in small fish act additively to reduce the value of the indicator.

Spatial analyses of the responses of fish communities to exploitation such as the above are easily biased if the fish communities routinely move and migrate on larger scales than the scale of analysis (Dingle, 1996). This is because the effects of fishing in one area will be manifest in another. It is certain that many North Sea species routinely move on scales greater than the ICES rectangle (Hunter *et al.*, 2004), but there is also clear spatial structure in fish communities that is persistent over time (Callaway *et al.*, 2002). The movements of fish to areas where fishing intensity is lower or higher than in the rectangle for which a metric is calculated will undoubtedly modify the relationship between the trend in the indicator and fishing intensity. Thus, at progressively smaller scales, and notwithstanding the other influences on fish communities that we describe, trends in the indicator are expected to provide less information on the impacts of fishing.

Our assessment of the effects of spatial management measures resulted in a response of one indicator (slope in the biomass-size spectrum) that was completely in accordance with theory: a decrease in effort in the box area results in the downward trend becoming less negative while the trend in the reference area becomes more negative as effort from the box area is reallocated to this area. For other indicators the difference between the two areas does reveal a pattern in accordance with theory (Hill's N_2 and to a lesser extent mean weight and Hill's N_1), but this is not the case in each of the areas separately. The remaining indicators do not show interpretable patterns (trophic level and mean maximum length) or even patterns that indicate an inverse relationship with fishing effort (Hill's N_0 and biomass). Therefore, the majority of community indicators do not appear to be suitable for assessing the effects of spatial management measures. Whether this is a result of the size of the closed area or other factors remains to be assessed.

In the case of the plaice box, Pastoors *et al.* (2000) have shown that the measure resulted in a reduction of more than 90% of the effort in the box, and part of this effort was probably reallocated into the reference area, so increasing fishing effort. Although this provides a clear example of spatio-temporal variation in effort, the observed changes in the indicators may not be determined by this effect alone. Pastoors *et al.* (2000) suggested that increased predation or changes in the distribution of juveniles, factors not necessarily directly related to fishing effort, may have impacted the stock developments in plaice and thus the wider fish community. In addition, temperature measurements during the survey indicate that (at least in the third quarter) there has been a marked increase in the temperature in the south-eastern North Sea that coincided with the establishment of the plaice box and this may have confounded the effects of the changes in fishing effort.

Because the size-based indicators reflect relative changes in the fish community they not only respond to a fisheries-induced decrease of large fish (direct effect) but also to an increase of small fish through a release from the predation pressure (indirect effect) (Daan *et al.*, 2003). This direct effect can be short term, whereas the indirect effect must occur at longer time scales. The indirect effect therefore prevents the size-based indicators from being "relatively tightly linked" to fishing, thereby making them less suitable as short-term indicators to support an EAFM.

The disaggregated analyses provide convincing evidence that fishing accounts for much of the observed change in some indicators. For these indicators, however, there remains a concern that the theoretical understanding of their response to fishing is not well developed. In particular, the rate of change may not only depend on current levels of fishing but also on environmental conditions and the historic fishing regime. A fixed level of fishing effort may cause an exponential decrease in the indicator, where the rate of change as well as the terminal value depends both on the level of effort and community structure and function, as determined by the environment and history of fishing. This is consistent with the observation that the impact of a given level of fishing effort on a previously unexploited community is greater than any subsequent effects (Figure 6). Likewise, the observation that several indicators show the same response in the deep and shallow areas, despite differences in the absolute levels of effort (Figure 5), may reflect the fact that fishing has only relatively recently started to exploit the deep area and therefore the indicators are in the early phase of their trajectory.

In general, the size-based indicators that we tested did provide evidence for fishing impacts on communities, consistent with other studies of these indicators in the North Sea (Rice and Gislason, 1996; Jennings *et al.*, 2002). Our examination of their response to changes in fishing effort and management action at smaller spatial scales and over shorter time periods than previously considered, has demonstrated they are not tightly linked to fishing effort and management action in space and time. Moreover, since previous studies have shown that the power of North Sea surveys to detect significant temporal trends in indicator values is quite low (Nicholson and Jennings, 2004), the application of size-based indicators should proceed with caution. In particular, any application in support of the EAFM at relatively small temporal (typically <5–10 years) and spatial (typically <30–70 ICES rectangles) scales should be underpinned by an improved theoretical understanding of their response to the direct and indirect effects of fishing, historical fishing activities, the environment, and relevant spatial processes.

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