

How Far have the Ecological Effects of Fishing in the North Sea Ramified?

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With 5 Text-Figures and 1 Table

Keywords: long-term, ecosystem perspective, food web, direct effects, indirect effects, nutrient dynamics.

Abstract

[FRID, C. L. J. & CLARK, R. A. & PERCIVAL, P. (2001): How far have the ecological effects of fishing in the North Sea ramified? – *Senckenbergiana marit.*, 31 (2): 313-320, 5 figs., 1 tab.; Frankfurt a. M.]

Man has been taking fish from the North Sea for 1,000s of years and intensive, mechanised fishing has been occurring for over 100 years – longer than the period of marine biological studies. It is therefore impossible to use the standard approaches to the assessment of the ecological effects of fisheries – there are no baseline/pre-impact data. In this study a variety of approaches are used to attempt to assess to what extent the direct (removal of target species) and indirect (removal of non-target species, habitat change, nutrient, organic flux changes etc.) effects of fisheries have altered the North Sea ecosystem. The approaches used include classic benthic time series, comparisons of historic records and modelling. Each approach has its flaws and limitations but by generating a synthesis of these findings it is possible to make an assessment of the ecological role of fisheries even with no pre-impact data. The results suggest that in many areas changes in the benthos have occurred as a result of fishing activities. In some areas such changes may have occurred as early as 1915-20, but in other areas changes were still occurring in the 1980s as fishing gears/intensity altered.

While some 'vulnerable' taxa have been heavily impacted many of the observed changes are subtler and may represent indirect effects such as altered predation rates or organic material/nutrient fluxes.

Introduction

In marine ecosystems fishing, which represents the biggest anthropogenic impact (DAYTON et al. 1995), is both widespread and has been on going since pre-history (DESSE & DESSE-BERSET 1993). Fishing impacts the populations of target species, other species captured in the net – the by-catch – and potentially all the other species in the community, which interact with them (DAYTON et al.

1995). In addition, bottom fishing gears impact on the seafloor, causing mortality and injury to surface living and shallowly buried fauna (TUCK et al. 1998), altering physical habitat features (AUSTER et al. 1995), sedimentation (CHURCHILL 1989) and nutrient cycling (MAYER et al. 1991). In the short term such disturbed areas may attract mobile scavengers and predators (KAISER & SPENCER

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1994). It is difficult to assess the impact of these changes at the scale of the ecosystem (THRUSH et al. 1998) or the lengthy time scales over which exploitation has occurred.

It has been suggested that there have been long-term changes in benthic communities of the North Sea (e.g. REISE 1982; REES & ELEFThERIOU 1989; KRÖNCKE 1995). Several causes have been proposed such as the impacts of towed fishing gears, especially on the heavily fished areas of the eastern and southern North Sea, along with the effects of climatic and salinity fluctuations, eutrophication, and changes in zooplankton abundances. However, further evidence is required to determine whether anthropogenic influences, as opposed to natural or cyclical events affecting the benthos, are responsible for any changes, if, indeed any real changes have occurred.

Recent estimates for the biomass of the entire fish component of the North Sea ecosystem vary, without trend, between 8 and 12 million tonnes (DAAN et al. 1990; SPARHOLT 1990). Even if the total biomass of fish in the North Sea was somewhat higher in 1900, the fish landed at that time (DAAN et al. 1990) represents removal of around 10% of the total standing crop. Even by this time then fishing may have induced changes in the North Sea ecosystem.

That the amount of fishing, particularly in the North Sea, has increased considerably over the last nine decades is incontrovertible (DAAN et al. 1990; RIJNSDORP & VAN LEEUWEN 1996; SERCHUK et al. 1996) and now stands at around 25% of the standing crop biomass (SPARHOLT 1990). Ecological theory would suggest that such marked changes in some populations would have effects on other species in the community (e.g. PIMM & HYMAN 1987; BEDDINGTON 1995).

Not only has fishing a long history but fishing impacts are wide spread; fishing grounds are extensive (JENNINGS et al. 2000) and tend to be on the scale of coastal seas.

Fishing activities influence marine ecosystems in a number of ways. These include,

1. direct removal of the target species,
2. direct changes in size structure of target populations,
3. alteration in non-target populations of fish and benthos (RUMOHR & KROST 1991; CAMPHUYSEN et al. 1995; TUCK et al. 1998),
4. alterations in the physical environment (CHURCHILL 1989; MESSIEH 1991; RIEMANN & HOFFMANN 1991; AUSTER et al. 1995; SCHWINGHAMER et al. 1996),
5. alterations in the chemical environment (ICES 1998),
6. trophic cascades (CARPENTER et al. 1985) and altered predation pressure (FRID et al. 1999b).

To date it has been difficult to assess the extent of any ecological changes induced by fishing. Standard experimental testing of hypotheses is limited by; the lack of control areas (TUCK et al. 1998), the lack of pre-impact data (UNDERWOOD 1991, 1996), the lack of information on intensity of fishing (RIJNSDORP et al. 1998) and the fact that fishing is wide spread and long-term - so fishing effects are confounded in with environmental variation (DAYTON et al. 1995; DAYTON 1998; THRUSH 1998).

We attempt to use a number of approaches simultaneously to address the extent to which fishing may have

brought about effects on ecosystem functioning. We employ time series of benthos in fishing grounds, temporal comparisons of the diets of benthic feeding fish and ecological modelling of fish predation and benthic regeneration of nutrients. In making these studies we must be mindful that the data used were generally collected for other purposes and that correlations and 'changes' are weak indicators of causation (UNDERWOOD 1990, 1994, 1996). We are aware of these, and so apply 'conservative' criteria and examine the 'weight of evidence' (DAYTON 1998).

Northumberland Benthic Time Series

Long-term monitoring of two benthic stations of the Northumberland coast has been carried out since 1971 (BUCHANAN & MOORE 1986a, b; BUCHANAN et al. 1986). One station, at 80 metres (Station P) is located within a *Nephrops norvegicus* (Dublin Bay Prawn) fishing ground (where otter trawling is used), while the 55 m station (Station M1) is located on the edge of the fished area. Up to 1986, trends in the abundance of benthic infauna at both sites paralleled changes in phytoplankton index (with a two year lag to allow recruitment to the sieve (BUCHANAN 1993), suggesting that benthic productivity was controlled by organic matter input (Fig. 1a). From 1986 until 1990 there was an increase in fishing effort within the *Nephrops* ground which was maintained until 1990 (Fig. 1a). This increase in fishing effort caused a decline in benthic productivity at Station P (Fig. 1b) and large-scale year-to-year changes in community structure (Fig. 2) indicative of a stressed community (WARWICK & CLARKE 1993). Year to year changes at Station M1, outside the *Nephrops* ground, continued to closely mirror changes in phytoplankton input throughout the time series (Fig. 1a), and had little effect on year to year changes in community structure at this site (Fig. 2) indicating a relatively unstressed community.

Dab Diet 1950s-1980s

Many authors (e.g. CREUTZBERG et al. 1987; DECLERKE & TORRELE 1988; HALL et al. 1990; GREENSTREET & HALL 1996) have described the consumption of benthic macrofaunal species by the dab, *Limanda limanda* (L.). Since fish stomach contents have been closely correlated with benthic infaunal population densities (KNUST 1996), it seems reasonable that historic fish stomach contents data may be compared against contemporary data to establish whether there have been any changes in the benthos of North Sea fishing grounds.

Between July 1951 and July 1953, 73 otter trawl hauls were taken at five stations on Smiths Bank, Moray Firth, east of Scotland, by research vessels from the Aberdeen Fisheries Laboratory. A size matched sample of fish was obtained in August 1996 for comparison with dab stomach contents data obtained from trawls in August 1951 and 1952. A size matched sample was used to limit biases associated with ontogenetic changes in feeding preferences (details are given in FRID & HALL 1999).

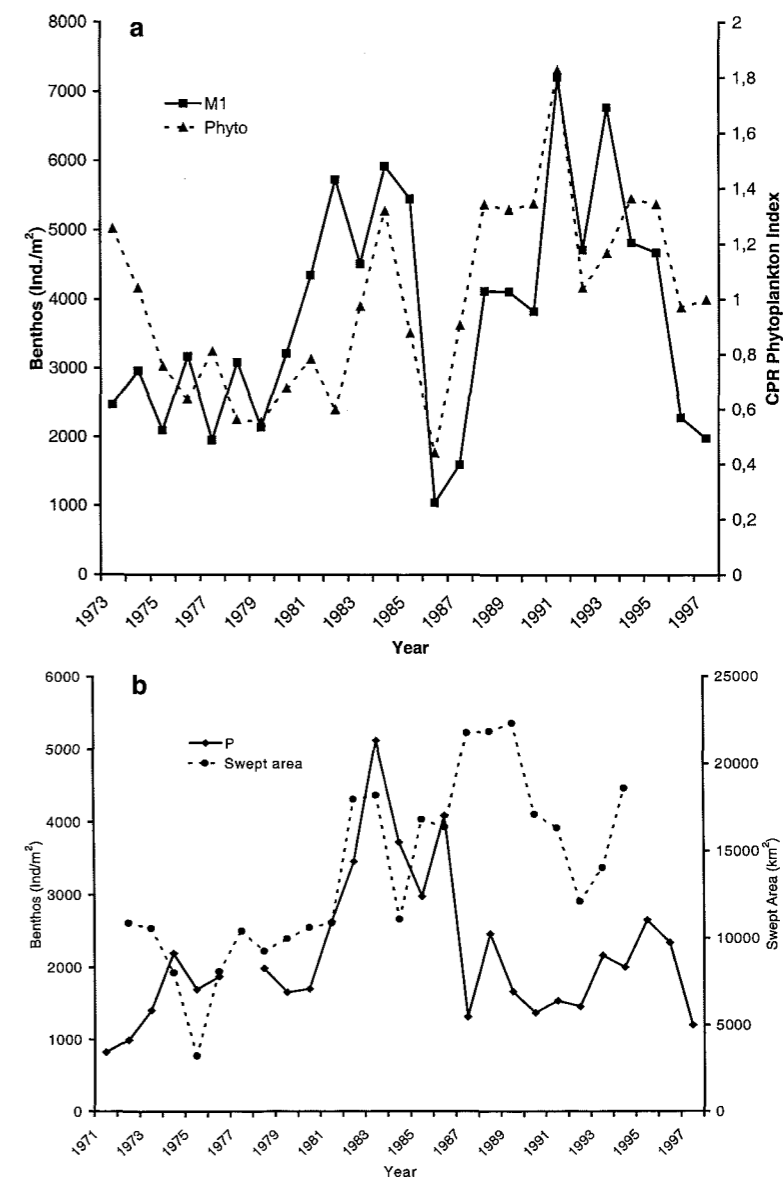
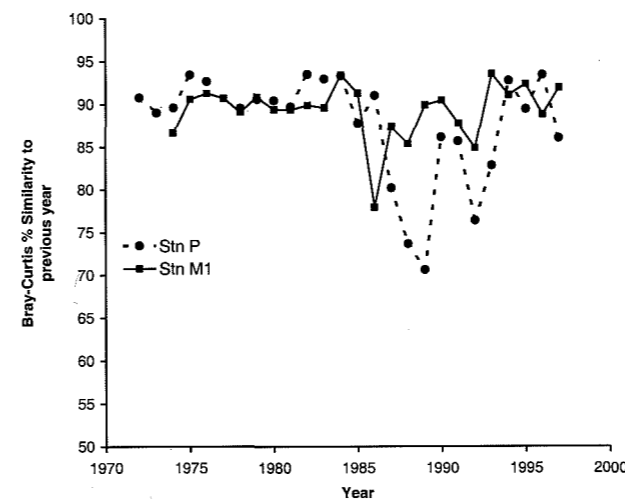


Fig. 1. The Northumberland benthic time-series. - (a) macrofaunal abundance (ind. m⁻²) at Station M1 and CPR phytoplankton index, (b) macrofaunal abundance (ind. m⁻²) at Station P and fishing effort (swept area) in ICES statistical rectangle 39E8.



Before undertaking any statistical analysis, all prey taxa found in the stomachs were placed into one of three categories: taxa expected to increase under fishing (e.g. scavenging species such as decapod crustaceans and carnivorous gastropods), taxa expected to decrease (e.g. large bivalves, sedentary tube building polychaetes) and taxa for which no *a priori* predictions could be made. The latter include multi-taxa prey categories in which some taxa would show a positive response while others would decline (The membership of these categories and

Fig. 2. The interannual Bray-Curtis similarities of macrobenthic species abundances from Northumberland Stations M1 and P. - The species composition of each assemblage in each year is compared to the composition in the previous year.

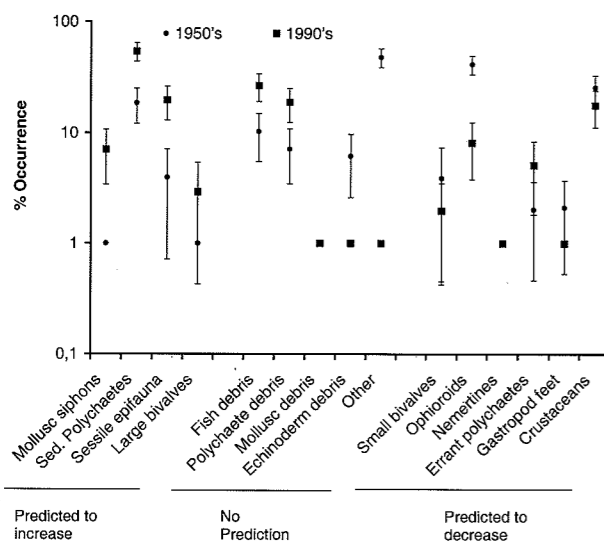


Fig. 3. Mean (●1950s; ■1990s) and bootstrapped 95% confidence intervals, derived from 1,000 randomisations, for the occurrence of various prey types in dab stomachs from August in 1951-52 and 1996-97. Prey types sorted *a priori* on the basis of their likelihood to increase or decrease as a result of fishing impacts.

the justification for their inclusion is given in Tab. 3 of FRID & HALL 1999). By placing taxa into such categories we, in effect, set up *a priori* hypotheses for the changes we expect to see in stomach contents if fishing has affected benthic communities. These hypotheses were based in large part on intuition and previous analyses of the effects of trawling (DEGROOT & LINDEBOOM 1994; LINDEBOOM & DEGROOT 1997; JENNINGS & KAISER 1998; THRUSH et al. 1998; TUCK et al. 1998).

It was found that, with one exception, the observed changes at the finest taxonomic resolution employed were consistent with a fishing effect (Fig. 3). The exception was a poly-phyletic group ('sessile epifauna') of fast growing opportunistic taxa, which can be very seasonally dominant and tend to show greater inter-annual variability in summer samples, following recruitment. At the phylum level changes in echinoderms, polychaetes and crustaceans were all in the direction predicted. Both crustacean and echinoderm prey comprised largely scavenging species and these increased, whereas the predominantly sedentary polychaete group decreased. Taken overall, we found dietary changes that were surprisingly consistent with a fisheries effect.

Ecological Modelling

Demersal Fish Predation on Benthos

By combining data sets on the abundance (Fig. 4), size frequency and size specific diet of North Sea fish it has been possible to evaluate predation pressure over the period 1970-1993 for the 8 most abundant demersal species (ICES 1998; FRID et al. 1999b). In spite of the

declines in target fish populations (gadoids and plaice) the overall level of predation on the benthos has increased from around 23 million tonnes year⁻¹ in 1970 to 29 million tonnes year⁻¹ in 1993 (Fig. 5). In addition, there are indications of a decrease in the proportion of crustaceans and molluscs in the diet and an increase in the importance of echinoderms (primarily ophiuroids).

As in all modelling studies the reliability of the findings are directly related to the validity of the underlying assumptions. Three principal assumptions underpin our models. They are that (i) the composition of the diet has not changed over time (ii) the composition of the diet derived from the 'year of the stomachs' programme can be used for the entire geographic region considered, (iii) the biomass at age estimate derived from VPA and survey data are valid and (iv) the total biomass can be used to predict predation levels outside the period covered by VPA.

Nutrient Cycling

Direct ecosystem effects of trawling have become a major focus of environmental concern, however, there have been few studies on the impacts of trawling on indirect effects. The indirect effects of trawling are less obvious than the direct effects, although may be more important in structuring benthic and pelagic communities (JONES 1992). Changes in nutrient fluxes are likely to be effected through a combination of: (i) impact on the redox status of the sediments (with concomitant consequences for microbial processes), (ii) stimulated release rates through trawl disturbance and (iii) changes brought about by the alteration of macro-benthic communities with knock on effects for the nutrient cycling process. Nutrient release from marine sediments can potentially supply a significant proportion of the nutrients required for primary production (NIELSEN & RICHARDSON 1996). Increased fisheries yield is dependent on primary production and, as a result, any activity that stimulates primary production in offshore waters, where terrestrial inputs of nutrients are limited, may help maintain fish stocks (MCCAFFEY et al. 1980).

To gain a measure of the impact of trawling on the sediment-water interface flux rate of nutrients in the North Sea a flux calculation based on current literature data was developed.

Modelling Flux Regimes

Nutrient flux processes vary depending on sediment granulometry. A broad classification and location of sediment grain sizes for the entire North Sea was obtained from the British Oceanographic Data Centre (ANON. 1965; VEENSTRA 1971). Rocky substrata were identified and excluded from subsequent calculations i.e. essentially assuming that fishing activities in these areas do not alter nutrient fluxes to the North Sea. While there is only a minimal sedimentary flux of nutrients from rocky substrata (MORIN & MORSE 1999), fishing in these areas may affect total fluxes by alterations to the biological assemblages and as a result of the addition of labile organic

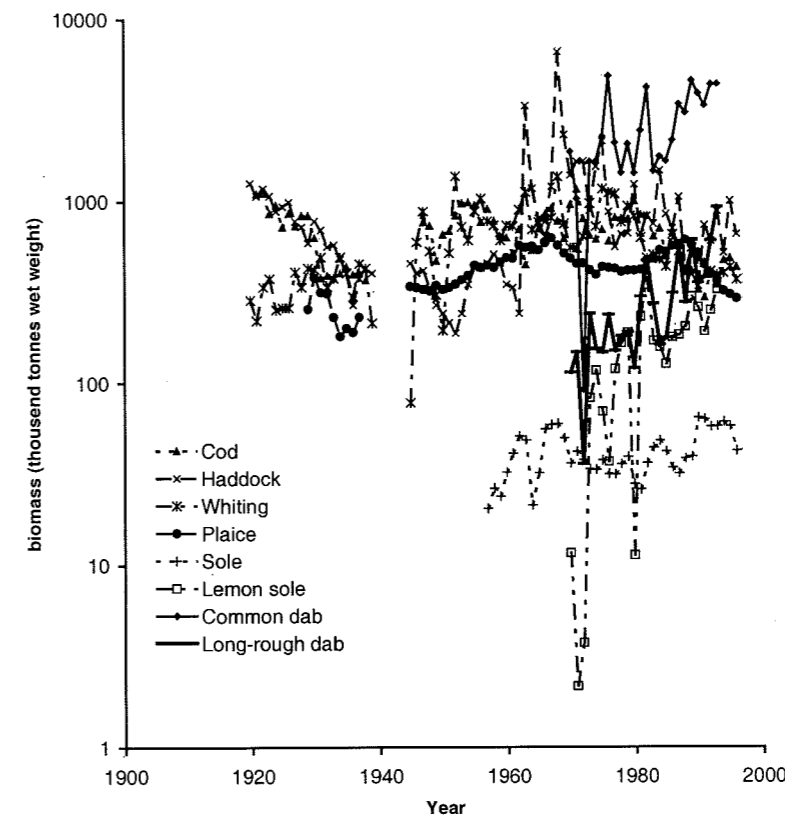


Fig. 4. Time series of the biomass (1,000 tonnes wet weight, note log scale) of 8 species of demersal fish in the North Sea.

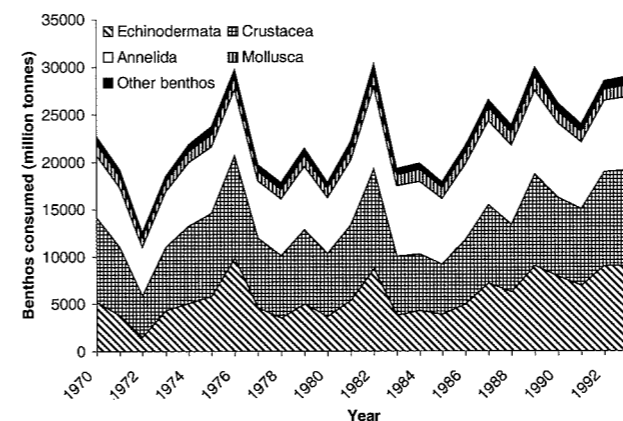


Fig. 5. Estimated total consumption of benthic food by 8 species of demersal feeding fish in the North Sea, 1970-1993.

matter, this assumption is therefore conservative. Sediments were then grouped into broader classifications (muddy or sandy) and the spatial extent of each calculated.

A background flux rate was first determined (the flux rate that would occur through diffusion, bioturbation and natural mixing without the influence of trawl disturbance)

(VAN RAAPHORST et al. 1990). Seasonally averaged flux rates were determined from the literature of studies that have made direct, in situ, measurements of benthic fluxes using chamber systems. Rates were calculated for each sediment type and for the inorganic nutrients; nitrate and nitrite, ammonia, phosphate and silicate (MCCAFFEY et al. 1980; NEDWELL et al. 1983). These were then scaled by the area of each sediment type to give a North Sea 'base flux rate' for each nutrient group.

Coastal regions generally have shallower water than offshore areas and consequently allow for easier experimental manipulation (MORTIMER et al. 1998). Trawled sediment flux rates were taken from those reported for disturbed coastal sediments (ROWE et al. 1975; EMERSON et al. 1984). Coastal marine sediments have a higher input of organic matter and these rates therefore, to some extent, mimic the additional organic input in the form of offal and discards at fishing grounds (KLUMP & MARTENS 1981, 1987; MORIN & MORSE 1999). Again rates were calculated for muddy and sandy sediments for each nutrient group. These gave us flux rates following a disturbance - a 'disturbed flux rate'.

The lack of good spatial data on fishing effort is widely acknowledged to be a major limitation in gaining an understanding of the impact of fisheries on ecosystems (JENNINGS et al. 2000). We obtained two estimates on the total trawl effort of the entire North Sea and have data for fishing effort in one ICES statistical rectangle. North Sea

wide and spatially resolved data were available for vessels >12m (JENNINGS et al. 2000), these data do not account for the large, inshore fleet of smaller vessels. The literature also includes statements about the frequency of total fishing gear impacts at the North Sea scale, with figures quoted of up to 12 passes a year over some pieces of ground but a likely maximum of 5 fishing events per year (GISLASON 1994; RIJNSDORP et al. 1998). Using average trawl speeds, average gear width and weight for otter and beam trawls it was then possible to calculate the total area covered per year for both otter and beam trawls (BERGMAN & HUP 1992). FRID et al. (1999a) provide data on the intensity of fishing in the coastal ICES statistical rectangle 39E8 (see Fig. 1b).

We use these data to model the effects of fishing on nutrient flux rates. We present here (Tab. 1) the results for 3 scenarios. The first is a highly conservative estimate for the entire North Sea with effort data only being included for >12m vessels and a short (2 hour) period of increased fluxes following a fishing event. The second scenario also covers the entire North Sea. It includes a high level of fishing activity (5 events per year at each location) and a

longer post impact recovery time (4 hours). The third scenario uses the actual fishing data for ICES statistical rectangle 39E8, the flux rates for muddy substrata (which dominate in the fished areas of this region (BUCHANAN 1963; FRID et al. 1999a) and an intermediate recovery time.

Our results show that based on even conservative criteria (scenario 1) for some nutrients fishing may be increasing benthic flux rates by around 1%. In the more realistic second scenario flux rates for some nutrients were enhanced by between 5 and 10% (total N +5.56%, P +9.6%). Off the coast of Northumberland (NE England) during the mid-late 1980s, when fishing was at its most intense, flux rates of nitrogen were increased by around 10% and those of phosphate by over 44%. Compared to the enhanced fluxes of these nutrients, due to agricultural runoff, in the southern North Sea these additions may seem small. However, in the central and northern North Sea fishing is intense and anthropogenic inputs of nutrients are relatively small (North Sea Task Force 1993). In these areas the extra nutrient supply may be ecologically important and the extent of this requires further study.

Discussion

We have shown that there is evidence of changes in benthos of a fishing ground during periods of increased fishing activity (Figs. 1b and 2). There is also evidence of change in the benthos at a number of North Sea fishing grounds over the course of the 20th century (LINDEBOOM & DEGROOT 1997; JENNINGS & KAISER 1998; FRID & CLARK 2000). Changes in many of the data sets did not include massive loss of taxa but shifts in balances, including an increase in opportunistic and scavenging taxa (see also FRID & CLARK (2000) and references therein). Such changes could result from direct impacts, habitat modifications or changed predation pressure (FRID et al. 1999b). There is also the possibility of the systems dynamics being altered by changes in primary production due to altered nutrient fluxes (Tab. 1).

Returning to our original question, it would seem that we can conclude that the ecological effects of fishing extend over multi-decadal time scales and operate at spatial scales ranging from processes within the trawl tracks (KAISER & SPENCER 1994), to changes at the scale of the coastal sea. We can also infer changes across the full spectrum of trophic levels: phytoplankton (through altered nutrient fluxes), benthos (through both direct mortality and indirectly through altered predation pressure and competition), fish (direct fishing mortality and indirect by provisioning from discards and altered competitive regimes) and top-predators (e.g. provisioning of birds - CAMPHUYSEN et al. 1995) and altered food resources for marine mammals (KAISER & DEGROOT 2000). The need to conserve biodiversity and to manage in a precautionary manner are requirements of international conventions. The extensive changes wrought by fishing on the North Sea ecosystem would therefore suggest an urgent need to explicitly incorporate ecosystem considerations into fisheries management.

Table 1. The percentage flux increase of nutrients across the sediment-water interface following trawl disturbance (% flux yr⁻¹). - Scenario 1: Trawl disturbance by vessels >12 m in length. - Scenario 2: Annual trawl disturbance estimated at approximately five times total fishable area of the North Sea. - Scenario 3: Fishing effort as recorded within the ICES statistical sub-rectangle 39E8 in the mid-1980's, fluxes rates based on local sediment conditions and a 3 hour recovery period. - See text for full details.

| Regeneration product | Scenario 1 | Scenario 2 | Scenario 3 |
|----------------------|------------|--|------------|
| | 2 hour | % flux increase for a ... 4 hour ... chemical recovery | 3 hour |
| Nitrate & Nitrite | 0.15 | 1.44 | 1.88 |
| Ammonia | 0.42 | 4.12 | 8.43 |
| Phosphate | 0.97 | 9.60 | 44.97 |
| Silicate | 0.78 | 7.66 | 8.43 |

Acknowledgements

Some of the work described here was in part supported by EC study contract 94/77 to CHRIS L. J. FRID and S. J. HALL, a Dept Marine Sciences & Coastal Management Bursary to ROBIN A. CLARK and a NERC Studentship to PHILIP PERCIVAL. The ideas and views expressed are those of the authors but we have benefited from discussions with many individuals including STEVE HALL, SIMON THRUSH, JOHN HALL, KIRSTY HARWOOD, ANDY REVILL, LEONIE ROBINSON and all the members, past and present, of the ICES Working Group on the 'ecosystem effects of fisheries'.

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Submitted: 18 August 2000

Reviewed: 25 April 2001

Accepted: 22 May 2001