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REPORT OF THE STUDY GROUP ON ECOSYSTEM EFFECTS OF FISHING ACTIVITIES

Copenhagen, 7 - 14 April 1992

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TABLE OF CONTENTS

1	INTRODUCTION	1
1.1	Participants	1
1.2	Terms of Reference and Scope of Work	1
2	SUMMARY FOR NSTF	1
3	OVERVIEW OF LONG-TERM CHANGES	5
3.1	Changes in Biotic Parameters	5
3.2	Changes in Hydrography	6
3.3	Spatial and Temporal Distributions of Fishing Effort	7
3.3.1	Introduction	7
3.3.2	Timetable of significant events	7
3.3.3	Time series of effort and mortality data	7
3.3.4	Distribution of fishing effort by NSTF areas	9
3.4	Changes in Other Anthropogenic Parameters	9
4	QUANTIFICATION OF THE DIRECT IMPACTS OF FISHING ACTIVITIES	9
4.1	Introduction	9
4.2	Estimates of Areas Affected by Towed Gears	10
4.3	Impacts on Physical Structure of the Sediment and Abiotic Processes	11
4.4	Mortality	12
4.4.1	Benthos	12
4.4.2	Fish	13
4.4.3	Seabirds	13
4.4.4	Marine mammals	14
4.5	Non-catch Mortality and Dumping of Discards and Offal	15
4.6	Lost Gear and Other Fisheries Debris	17
5	INFLUENCE OF ANTHROPOGENIC ACTIVITIES OTHER THAN FISHING	18
5.1	Eutrophication and Oxygen Depletion	18
5.2	"Unusual"/"Noxious" Algal Blooms	19
5.3	Effects of Eutrophication on Benthos	19
5.4	Hypoxia/Anoxia and Implications for Benthos and Fisheries	20
5.5	Organic Discharges, and Associated Contaminants	21
5.6	Shipping, Offshore Structures, Hazards and Port Activities	23
5.7	Mineral/Substrate Extraction and Disposal	24
5.8	Offshore Exploration, Exploitation, and Impacts	25
6	COMPARING THE DIRECT EFFECTS OF FISHING WITH THE EFFECTS OF OTHER ANTHROPOGENIC INFLUENCES AND NATURAL PROCESSES	27
6.1	Introduction	27
6.2	Impacts on the Physical Structure of the Sediment and Abiotic Processes	27
6.3	Mortality	27
6.3.1	Benthos	27
6.3.2	Fish	28
6.3.3	Seabirds	29
6.3.4	Marine mammals	30
6.4	Discards and Offal as Food	30
6.4.1	Seabirds	30
6.4.2	Other animals	31
6.5	Lost Gear and Other Fisheries Debris	31

TABLE OF CONTENTS (cont'd)

7	LONG-TERM EFFECTS OF FISHING ACTIVITIES	32
7.1	What Can be Known About the Long-Term Effects?	32
7.1.1	Effects of fishing on feeding interactions	32
7.1.2	Effects of fishing on the genetic make-up of populations	33
7.1.3	Other effects of fishing	34
7.2	Long-term Changes in the North Sea	35
7.2.1	Benthos	35
7.2.2	Fish	36
7.2.3	Seabirds	40
7.2.4	Marine Mammals	42
8	POSSIBLE MANAGEMENT OBJECTIVES, CONFLICTS AND SOLUTIONS	43
8.1	Fisheries Management Objectives	44
8.2	Ecosystem Management Objectives	44
8.3	Management Conflicts	45
8.4	Possible Solutions	45
8.4.1	Effort reductions	46
8.4.2	Gear restriction.	46
8.4.3	Gear modifications	46
8.4.4	Closed areas	46
8.4.5	Closed seasons	47
8.4.6	Other solutions	47
8.5	Conclusions	47
9	FOOD FOR THOUGHT	47
9.1	Changes in the Fish Assemblage Within Three Regions of the North Sea	47
9.2	Productivity (P/B) Ratios (Life History Characteristics) and Benthos Mortality	48
9.3	Managing Fish Stocks while Considering Wildlife - Problems of Scale	49
10	RECOMMENDATIONS	50
11	REFERENCES	50
Tables	3.1.1 - 7.2.3.1	67
Figures	1 - 9.1.4	91
ANNEX 1:	List of Working Documents	123
ANNEX 2:	Distribution of Fishing Effort by NSTF Areas in 1989	124
ANNEX 3:	ACFM Advice	138
ANNEX 4:	The Application of ACMP Advice in a Management Context	143

1 INTRODUCTION

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1.2 Terms of Reference and Scope of Work

The Study Group on Ecosystem Effects of Fishing Activities met at ICES Headquarters from 7-14 April with the following terms of reference:

- a) describe, as far as possible, the present direct impacts of fishing practices in terms of physical disturbance of the sea bed, mortality of target and non-target species including non-catch mortality, discarding and production of offal, and lost fishing gear and other fisheries-generated litter;
- b) compare, where practicable, the impacts of fishing activities on the marine ecosystem with the impacts of other anthropogenic activities in terms of type and scale;
- c) describe the historic trends in fishing effort, type of fishing, and abundance of fish and other organism;
- d) identify and describe long-term effects of fishing on the North Sea ecosystem and describe, as far as practicable, how historic trends in abundance of fish and other organisms relate to fishing and other factors;

- e) identify and describe the various fisheries and environmental management objectives and examine options for common measures;
- f) prepare a three-page summary of the findings which may be used as a component of the 1993 Quality Status Report on the North Sea environment;
- g) submit a written report to the Consultative Committee, ACFM, and ACMP which may serve as a background document to the summary.

Ecosystem changes may be described and quantified in a variety of ways. In this report, we have focused on describing changes in the abundance of individual species. However, we do recognize that for benthos, in particular, the practical problems involved in describing changes at the population level are large, due to the numerous species. In some cases, community properties have, therefore, been considered.

The effects of fishing activities may occur at several levels of spatial scale. At its previous meeting, the Study Group decided to consider three levels as targets: local, North Sea Task Force Areas (Figure 1) and North Sea wide. Because this report is going to serve as the background for a holistic assessment of the effects of fishing activities in the 1993 Quality Status Report, most effort has been put into describing the effects on a North Sea wide scale.

In a few cases, however, much of the North Sea population of a species may be restricted to a smaller area. In these instances, the effects of fishing may have a significant effect on the total population. These cases are addressed as far as possible in the report.

In other cases, particularly in coastal zones, fishing events may have large, but very local effects. These cases can be important for local subpopulations, but cannot be shown to have significant population impacts at the scale of the North Sea, often due to a lack of research. When such events are thought to be representative of a wider area, they are highlighted in the report, even though the amount of data does not allow a quantification of the impact for the entire North Sea population.

2 SUMMARY FOR NSTF *North Sea Task Force*

On the basis of the concern expressed by the North Sea Task Force regarding the impact of the fishing industry on the North Sea ecosystem and the subsequent request by the Third International Conference on the Protection of the North Sea that the NSTF address this issue in the 1993 Quality Status Report, ICES established the Study Group on Ecosystem Effects of Fishing Activities to review and report on this issue. This extended summary

describes the main findings of the Study Group. The full report of the Study Group will be available in September 1992.

Introduction

Ecosystem effects of fishing activities may occur at all scales of space and time. Although a clear distinction between local, regional and North Sea-wide effects cannot always be made, the global approach taken in this report is inappropriate for consideration of some more local ecosystem effects, both of fishing and other human activities, which will be considered in regional reports.

Fisheries exploit species against the background of a variable environment which is a major source of perturbation to the system. The effects fisheries cause should thus be viewed as one of several anthropogenic interactions in a non-equilibrium system.

Fishing has a number of direct effects:

- a) It causes mortality on the target fish and incidentally on other biota;
- b) It makes food available to other species in the ecosystem by i) discarding unwanted catch of fish and benthos, ii) discarding wastes, and/or iii) by killing or damaging animals in the path of the gear during its deployment;
- c) It disturbs the seabed by the action of some fishing gears; and
- d) It generates litter composed of lost or dumped gear as well as other, non-specific, debris.

These direct effects in turn can lead to indirect effects such as the modification of predator-prey relationships, thereby changing the flow of energy through parts of the system. They can also lead to the modification of habitats.

Temporal trends in fishing and marine biota in the North Sea

Fishing in the North Sea has a long history (see Table 3.3.2); some significant technical developments took place by the end of the Nineteenth Century, and the pace of innovation has accelerated since then.

Time-series data on total catches by species are available from the beginning of the century onwards. Total catch increased gradually from 1 million tonnes around 1900 to 2 million tonnes around 1960 (see Figure 3.1.2.a). During the 1960s, the catch increased steeply to 4 million tonnes, followed by a gradual decline to around 2.5 million tonnes in recent years. The catch of major fish

categories (e.g., pelagic, roundfish, flatfish, industrial) has been rather more variable (see Figure 3.1.2.b). Roundfish catches increased significantly in the 1960s, an increase that is generally referred to as the "gadoid outburst", followed by a gradual decline since 1970. Pelagic fish catches decreased sharply after about 1970, while industrial fish catches increased at about the same time. More recently, some pelagic fish catches have increased, some roundfish catches have decreased to their lowest levels in the past 30 years, while industrial catches have been maintained, albeit with major changes in species composition.

Fish catches depend both on fish abundance and on the intensity of fishing. Estimates of fish abundance for a number of fish species have become available during the past 40 years, and time series of these are shown in the reports of the ICES Advisory Committee on Fishery Management.

The development of the intensity of fishing, expressed as fishing mortality rate, for a number of important North Sea stocks is shown in Figures 3.3.3.1 to 3.3.3.4. Fishing pressure has generally increased over the last century. For some stocks, notably haddock, exploitation has been high since the early part of this century.

Time-series data for number of seabirds go back to the beginning of the century and show large increases for many species (see Figure 3.1.5). Some species have, however, shown declines over the last 15 years (see Table 3.1.1). Time-series data exist for common and grey seals since 1965 and show that seal numbers increased until 1988, but then common seal numbers declined sharply owing to the phocine distemper epidemic (see Figure 3.1.6). For cetaceans, the trends in abundance are far less certain. However, the available evidence suggests that declines have occurred in porpoises in the southern North Sea since World War II and in bottle-nosed dolphins over the last century. No appropriate large-scale time-series data exist for benthos.

Direct effects of fishing

Some of the direct effects of fishing can be quantified.

a) Mortalities

The current levels of fishing mortality (expressed as percent of the population present at the start of the year which is caught during the year) for the most important commercial fish species in the North Sea are summarized in Table 4.4.2. For some species, these levels imply that more than half of the fish of exploited ages will be captured during a year.

Fish also escape through the meshes, and some of these are damaged or killed. The quantities which escape cannot readily be assessed, however, the proportion of these which is likely to survive is given in Table 4.5.1.

The fishing effort (intensity of fishing) deployed in 1989 in almost all of the NSTF areas has been calculated (see Tables 3.3.4.a-i) for various gear types. For towed gears the effort data have been converted into estimates of the total swept areas¹ (see Tables 4.2.3 and 4.2.4). It should be noted, however, that the application of fishing effort is very uneven and, thus, certain areas will be fished many times while others are missed.

In the case of benthos, towed fishing gears cause mortalities on infauna and epifauna.

Infauna is most affected by gears that penetrate the seabed, such as beam trawls. Mortalities on animals in the path of beam trawls have been estimated for a limited number of species and ranged from 15% to 55%. The conversion of percent mortality in the trawl path into mortality for individual Task Force areas or the entire North Sea is problematic. This would require estimates both of the spatial distribution of the gear deployment and of the benthic species. Among the benthic animals caught in the beam trawl, the mortalities range from virtually zero to close to 100% depending on the species.

Epifauna is affected by all towed fishing gear, but insufficient information is available on the relative catchabilities of different species in different types of gear.

In the case of seabirds, there are many observations that attest to mortality from entanglement in fishing gear. Evidence points to the largest impact being from gill nets and other fixed nets on diving seabirds. This impact cannot be quantified at present. It is thought to be sporadic and localised. The local mortality rates can clearly be high, but at the North Sea population level and even at the colony level, they do not preclude population increase.

Seals can become entangled and killed in fishing gear. Evidence suggests that the largest impact

comes from fixed salmon nets. In some countries, seals may also be shot legally if they interfere with fishing gears. The resulting mortalities have not been quantified, but the North Sea seal population levels have not declined in this context.

Small cetaceans are caught in gill nets, but the data are not adequate to quantify the resulting mortality.

There is a shortage of relevant demographic information on biota affected by fishing gear.

b) Food inputs

After capture, fish and benthos may be discarded for either regulatory reasons (undersized or over-quota fish) or because no market for them exists. For haddock and whiting, caught in demersal fisheries in the northern North Sea, extensive discard data are collected routinely (see Table 4.5.2). For other fisheries and areas, some data have become available and approximate discard rates can be derived (see Table 4.5.3). However, no global estimate can be provided at present. Some of the discarded animals survive, but many are dead or moribund. In addition, some fishing operations dump fish offal (fish processing waste). Both discards and offal provide an important food resource for scavenging organisms, notably seabirds. Fish and other organisms that have passed through the meshes, either already dead or dying as a result thereof, add to these inputs, but will be more available to benthic scavengers.

c) Physical disturbance of the seabed

The action of some fishing gears physically changes the seabed, but the effect will depend on the gear and the nature of the substrate. There are rough estimates of how far individual gears penetrate into the seabed (Table 4.2.1). There are also estimates of the total area swept² (Tables 4.2.4 and 4.2.5). Heavy towed gears can change the sediment characteristics of the seabed, displace boulders which form a primary substratum for benthic organisms, mobilize sediment particles, leading to transport of fine particulate matter and modification of sediment geochemistry including sediment-water exchange (e.g., of nutrients).

¹ The swept area is the seabed area that would be covered by the deployment of the gear if there were no replication. N.B. This should not be used to infer the area of the North Sea impacted by trawling because replication may frequently occur.

² The total area swept is the seabed area that would be affected by the deployment of the gear if there were no replication. N.B. This should not be used to infer the area of the North Sea affected because replication may frequently occur.

d) Litter

Fishing operations generate litter through the accidental loss of gear and by the dumping of damaged gear. In addition, fisheries produce debris comparable to that produced by shipping in general, such as the plastic litter which ends on the sea floor and on beaches.

Comparison with other anthropogenic activities

Anthropogenic activities other than fishing also affect the North Sea. The anthropogenic activities covered in the context of this report preclude fishing and/or mimic fishing effects on biota. Thus, it may be very difficult to separate these effects from the consequences of fishing.

Offshore structures (e.g., wrecks and platforms) locally preclude fishing operations. They provide refuges and sites of increased biomass and diversity for fish and benthos. Point sources of wastes from oil production, and organic/sewage enrichment, alter benthic communities in the vicinity, frequently resulting in a dominance of short-lived, opportunistic species; this is essentially comparable with the effects of intense bottom trawling. Hypoxia can be caused by the combined action of several factors, e.g., poor water exchange, elevated water temperature and eutrophication. Areas of hypoxia have resulted in emigration of fish and other motile biota and in mortality of more sessile biota (e.g., benthos and shellfish); these effects may initially result in elevated catches of benthos and shellfish in trawls before mortality occurs, while hypoxia-caused emigration and mortality may mimic some of the results of overfishing.

Long-term effects

Generally, the long-term effects of fishing on marine biota are impossible to predict at the species level. While short-term effects may be predicted, the long-term consequences to species may be quite different. Predictions may, however, be possible of broader community attributes, such as species diversity or size distributions. A further important exception to the rule of limited predictability is the possible elimination of a vulnerable species by sustained over-exploitation. Slow-growing species with low fecundity are the most likely candidates for such local extinctions.

Fisheries have altered the size distribution of exploited fish species by reducing the abundance of larger fish.

Populations of fish species taken for human consumption have been heavily exploited for most of this century, but have been able to withstand this. Experience suggests, however, that intensive exploitation requires careful control during periods of naturally induced decreases in recruitment. Some gadoids are at their lowest level of

spawning stock biomass in the past 30 years, and this gives reason for concern.

Far fewer data are available for the short-lived fish species exploited by the industrial fisheries. These species constitute an important source of food for a number of other species and changes in their abundance may thus have important direct and indirect consequences.

Consideration of the biology of many benthic species suggests that they are unlikely to suffer from recruitment failure at the current levels of fishing. However, undisturbed reference sites are not available and areas closed to fishing for scientific investigation would facilitate understanding of the processes.

Despite low fecundities, birds and seals have sustained their populations while subject to fisheries-induced mortality. The situation for cetaceans is uncertain due to an ignorance of their distributions, abundances and mortality rates. Clearly, a better understanding of their population dynamics is needed, but the generally low rate of reproduction in cetaceans suggests that they might be particularly vulnerable to additional mortality caused by fishing.

Apart from the long-term population effects of fishing, it is possible that the selective pressure of fishing might lead to evolutionary changes in the biology of affected species. It might also reduce genetic diversity in exploited stocks.

The fishing industry currently introduces considerable amounts of food into the marine environment in the form of discarded fish, offal and animals killed by non-catch fishing mortality. Changes in discarding practices due to changing fisheries regulations (e.g., mesh changes to eliminate the capture of discards or the prohibition of discarding) could, therefore, produce large effects on scavenger species and the species with which they interact.

The physical effects of fishing can lead to structural changes in habitat and thus changes in species assemblages. Some bottom communities may be more vulnerable than others, but the extent to which changes have occurred cannot be assessed at present.

Given the complexity of the interactions among the various components of the system and the variability of the environment, it is difficult to separate the long-term effects of fishing from changes due to other factors. Very few of the observed long-term changes in North Sea biota have thus been conclusively linked to fishing.

3 OVERVIEW OF LONG-TERM CHANGES IN THE NORTH SEA

During the past three decades, a considerable amount of research effort has been devoted to improving the understanding of the ecology of the North Sea. In particular, several symposia have been aimed at the integration of the knowledge available within different scientific disciplines dealing with the marine environment in order to address the issue of the causes of observed changes (Goldberg, ed., 1973; Hempel, ed., 1978; De Wolf *et al.* eds., 1990a,b, 1991; Dickson ed., in press). Although a wealth of time series information has been made available on a large variety of biological and hydrographical characteristics of the North Sea as well as of parameters describing anthropogenic activities, the outcome in terms of cause and effect has not been particularly conclusive so far. It is not possible to condense all the available information effectively here and the reader is referred to the original symposium volumes. However, in order to allow for a discussion on the long-term effects of fishing on the system (Section 7), the outstanding events are summarized in this section. In addition, new data are provided describing the global development of the fisheries during this century as well as a description of the spatial distribution of the present effort, categorized by gear type, over the different NSTF areas in 1989.

3.1 Changes in Biotic Parameters

Phytoplankton

Data collected since 1932 during the Continuous Plankton Recorder Survey show consistent patterns of geographical, seasonal and annual variation in the distribution of phytoplankton and its major taxonomic components (Reid *et al.*, 1990). Coloration of recorder silks (assumed to be indicative of algal biomass larger than 280 μm) shows little evidence for long-term trends except possibly in the southern North Sea where there was a general increase in levels until the mid 1970s, since when colour has declined. Long-term observations made at Helgoland since the 1960s also show trends of increasing phytoplankton biomass up to 1984.

The authors conclude that there is no evidence from existing reports for an increase in the frequency of occurrence of algal blooms, although some recent years stand out with larger numbers.

Zooplankton

The zooplankton community in the North Sea varies geographically in relation to the origin of the different water masses (Fransz *et al.*, 1991) and variations have been observed in the area of penetration of oceanic species in the northern North Sea. The Continuous

Plankton Recorder Survey data indicate a marked and consistent decline in zooplankton abundance (predominantly copepods) from 1950 to 1980, followed by a recovery in the late 1980s to the level of the 1960s (Figure 3.1.1).

Benthos

Although recent efforts have greatly enhanced our information on the quantitative distribution of both macrobenthos (e.g., Eleftheriou and Basford, 1989; Duineveld *et al.*, 1991) and meiobenthos (Heip *et al.*, 1990), there is a paucity of time-series information that would allow an evaluation of long-term trends in the bottom fauna, except on a very local scale (e.g., German Bight, Rachor, 1990; off northeast England and in the Skagerrak, Austen *et al.*, 1991). An EC initiative to coordinate time-series studies across large geographical areas is notable in this respect (Keegan ed., 1991).

Fish

Thanks to the routine stock assessment of commercial fish species, detailed information is available on changes in the structure of these populations at least for the last 30 years (Daan *et al.*, 1990a; Anon., 1991a, 1992c). These publications provide detailed time-series information on biomasses, fishing mortalities and annual recruitment. Figure 3.1.2 summarises only the trends in landings in Sub-area IV. Figure 3.1.2.a shows the total landings and Figure 3.1.2.b the landings of the 11 major species split according to 4 groups. The important features include a significant increase in the landings of the four gadoid species in the 1960s, which is generally referred to as the 'gadoid outburst', followed by a gradual decline since 1970. Both the herring and mackerel yielded extremely high catches in the 1960s, followed by a marked decline. The catch of short-lived industrial species also increased in the 1960s and early 1970s. Although maintained at a high level, the composition of the industrial catch has undergone some marked changes in the 1980s. Sandeels make up the vast majority of the catches and sprat has markedly decreased. The catch of sole has remained remarkably stable since the late 1940s whereas the plaice exhibits a steadily increasing trend over the entire period.

At the Århus Symposium, time-series data have also been presented on a variety of by-catch species of lesser economic importance (Hempel, ed., 1978). A regular bluefin tuna fishery developed after World War II, although specimens had been taken as a by-catch in the herring fishery from the second half of the 19th century. The North Sea fishery collapsed in the 1960s, apparently due to a change in migration (Tiews, 1978). Changes have also been recorded for a number of 'southern' fish species (sardine, sea bream, horse mackerel) penetrating irregularly into the North Sea (Postuma, 1978). How-

ever, a problem is that many of these investigations have not been brought up to date, although a considerable amount of new information must be available from the various trawl surveys that have been carried out since 1975. Data from other sources are in some cases available (e.g., De Vooy *et al.*, 1991), but have to be integrated before they can be properly evaluated.

One indication of changes in abundance for by-catch species of commercial interest is presented by the total landings as reported in ICES Bulletins Statistiques. Figure 3.1.3 provides trends in the reported landings of less common large gadoids, large flatfish species and elasmobranchs. Although considerable care is required in interpreting such data in terms of absolute abundance, these data suggest that catch levels of the roundfish and the flatfish have remained remarkably constant with the exception of the halibut. Among the elasmobranchs, the rays exhibit a steadily declining trend in reported landings over most of the century. In contrast, the catches of sharks increased remarkably after World War II, and have only fairly recently shown a decline.

Birds

Due to the relative ease with which some seabirds can be counted at their colonies, there are good time series of counts available. Table 3.1.1 indicates changes between the two most complete censuses on British North Sea coasts. Many of these figures represent changes for the whole North Sea population. Figure 3.1.4 shows trends in numbers of gulls and terns in the German Wadden Sea where censuses are conducted on an annual basis. Such information exists also for a few other parts of the North Sea.

Some time series may be extended further into the past. Fulmars did not nest in the North Sea until 1878, but are now the second most numerous species (Figure 3.1.5). Gannets have increased throughout this century with new colonies still being established to extend their breeding range (Leopold, 1991). Kittiwakes increased rapidly for much of the present century, but the rate of increase has slowed recently. The most rapid increase was in the southern North Sea, and most of the recent declines in numbers have occurred around Shetland (Coulson, 1963, 1983, Lloyd *et al.*, 1991).

Mammals

There are reliable estimates of the seal populations in the North Sea (Figure 3.1.6; Reijnders and Lankester, 1990), indicating a steady increase in grey seals since the 1960s. Common seals in the coastal areas showed a steady decline until the early 1970s, after which time the populations increased, with a major interruption in 1988.

For the cetaceans, the trends in population size are far more uncertain. Harbour porpoises are regarded as being primarily coastal in their distribution, occurring mostly in shallow continental water. The North Sea is, therefore, regarded as the most important habitat for porpoises in the northeast Atlantic (IWC, 1991). Information on the population size of porpoises in the North Sea is limited. Norwegian surveys have provided an estimate of 82 600 for the northern North Sea (Björge and Öien, 1990), but no surveys have been published for the southern North Sea. However, the available evidence suggests that the abundance of porpoises in this area has decreased considerably since World War II (Reijnders and Lankester, 1990).

Only one resident population of bottlenose dolphins is known to live in the North Sea (Moray Firth, northeast Scotland, Anon. 1991b). Also for this species, the available evidence from sightings and strandings suggests a general decrease during the last century (Reijnders and Lankester, 1990).

Other small cetaceans observed in the North Sea include white-sided dolphins, white-beaked dolphins, pilot whales, killer whales and others. No quantitative data are available on the present or past absolute abundances.

3.2 Changes in Hydrography

As with many other features relevant to this study group, the variation in physical oceanography of the North Sea is patchy in space and time. One must heed admonitions about the unrepresentativeness of short-term means in physical measurements, and of local measurements. Some generalizations are possible, such as the importance of climate and weather to the state of the physical environment, and the fact that there appear to be many weather-related events at time scales of around 50 days. There are major circulation patterns, as well, with important inflows through the Strait of Dover and via the Fair Isle Current, a flow first out to and then returning from the Skagerrak, and a major outflow through the Norwegian Current. Although the turnover time for water in the North Sea is of the order of one year, in the places of major inflow or outflow the turnovers are in the order of 4 months (Otto *et al.*, 1990). There is a great deal of variation in the timing of these water transports, particularly the influxes from the North Atlantic, with the occurrences of major pulse events varying by several weeks over a few years. The variability of these oscillations has large effects on production at lower trophic levels of the North Sea.

Through this background of substantial variability in oceanographic conditions in space and time, some significant anomalies have been visible in recent decades. These events have been large enough and persistent enough to be trackable. The most marked event has been

the large anomaly in Atlantic water inflow in the late 1970s, known as the "Big Slug", that moved through the Northeast Atlantic. This produced significant cooling of sub-surface waters. Salinity showed a corresponding decrease. The major salinity anomaly was more pronounced in the north, and had moved out of the North Sea by the beginning of the 1980s. Temperatures remained relatively cool until very recently, when they have increased abruptly to levels matching the warmest on record (Svendsen and Magnusson, 1992, Figure 3.2.1).

On a longer time scale, there appears to have been a prolonged and gradual warming trend for most of the first half of this century, with the trend most marked in the summer, and in the northern parts of the North Sea (Figure 3.2.2). No overall trend is apparent in the salinity record, but positive anomalies persisted for the late 1940s and late 1950s, with the intervening period tending to be low, but variable. The pattern of salinity changes shows a slightly later timing in the south than in the north. There is substantial variation in salinity on a year-to-year basis, more variation than is captured in the coarse multi-year patterns (Hill and Dickson, 1978; Otto *et al.*, 1990; Figure 3.2.3).

3.3 Spatial and Temporal Distributions of Fishing Effort

3.3.1 Introduction

Fishing activity in the North Sea has a long history, as do complaints about its effect on the environment. Some fishing gears, such as gill nets and hooks and lines, have their origin in prehistory whereas other more modern gears have surprisingly long histories. Complaints about possible environmental effects of the use of trawls in the North Sea are known as early as the 13th century (de Groot, 1984).

3.3.2 Timetable of significant events

The technological developments of the past two centuries have had a profound effect on the types of fishing gear deployed in the North Sea and on the intensity with which they can be deployed. Many of these innovations have led to a more efficient exploitation of the various fish resources of the North Sea. The increased efficiency aggravated difficulties in limiting fishing effort. These factors have produced increases in the mortality rates on fish stocks. Table 3.3.2 shows some of the more important developments in the technology of fishing during the last 120 years. The effects of these innovations on other parts of the North Sea ecosystem are, however, less clear. Many may have decreased the effect of fishing on non-target species.

Some reductions in impact are consequences of directing fishing more precisely onto the target species, e.g., the echo-sounder or fishing charts. Other reductions in impact follow from increasing the fishing efficiency of the gear while not increasing its effect on the bottom community, e.g., Vigneron-Dahl gear, high headline gear etc. Some innovations have been designed specifically to reduce impacts on non-target species, e.g., grills on shrimp trawls (although their effects are not always as expected). Other innovations may have produced greater effects on the ecosystem by increasing by-catches of other species in the quest for higher fish catches, e.g., the twin beam trawl and some gill nets.

For these reasons, the effect of fisheries on the marine environment cannot be related in any simple way to the level of fishing effort generated on the various species of fish. Historically, fishing effort data have been collected with the objective of measuring the mortality rate on fish. Even in those direct applications, their use is not always straightforward. Considerable caution should be used when interpreting effort data as measures of effects on other parts of the ecosystem.

3.3.3 Time series of effort and mortality data

Fishing effort data attempts to measure the amount of fishing that is done. Fishing effort is a rather vague term which means different things to different groups of people. To an economist, for example, it might mean the monetary cost of fishing, to a fishery manager it might be some measurable quantity of fishing time by particular classes of vessel, and to a fisheries biologist it would usually be seen as a measure of fishing intensity which correlated with the level of fishing mortality generated on particular fish stocks. Each of these various viewpoints might well generate different data time series from the same fishing fleet and we need to be rather careful in how we combine and interpret such time series. For the purposes of this report, we wish to obtain an overview of the development and deployment of fishing fleets. In doing this, we may well be aggregating data too far for other more detailed purposes.

Effort data are available in the national statistics of a number of ICES member countries and in some international collections of fisheries statistics. The motivation for collecting and publishing national statistics is often to inform governments about the domestic industry. Therefore, the data collected and reported at a national level can change from time to time as can the diligence with which the data are collected. International time series of effort data were more often compiled with the intention of measuring the impact of fishing on fish stocks. Nevertheless, these series suffer from the incompatibilities and incompleteness of the national collection systems upon which they are based. Moreover, such series seldom record the small progressive changes in

practice and efficiency that have occurred. Rather, incremental changes are not addressed for some period, and then an abrupt compensation is made in the series. This makes the compilation of long time series of effort and fishing mortality data difficult. It also begs the question as to what the time series should seek to describe. One approach to take might be to try to record the raw inputs to the fishery, such as energy usage or the number of men employed in the industry. Calibration of such a series would be difficult, so a more feasible approach, given the existing data sets, is to try to measure the fishing intensity on various key fish species.

An alternative to effort time series data for describing the intensity of fishing on different fish stocks has become available in more recent years. This is to use the time series of fishing mortalities from virtual population analysis (VPA), as reported in ICES working group reports. This technique uses the data sets of total international catch-at-age data by species, compiled by ICES, to estimate fishing mortality on various species in the North Sea. Table 3.3.3 gives the references to these time series and their extent.

A number of distinct fisheries exist in the North Sea and there is no single measure of fishing effort and associated mortality that can be applied to describe the evolution of the entire fishery. Rather, fishing effort must be seen as a multidimensional attribute. Careful study of relationships among fisheries may allow calibration of effort and mortalities for some combinations of species, but is likely to indicate that fishing mortality on one species simply cannot be used to estimate mortality on other species. If fishing mortalities differ among stocks targeted by fisheries, it is even less certain to use the trends of fishing mortalities on fish as direct indicators of impacts on other components of the North Sea.

Despite these reservations it would be worthwhile to use available fishing effort series to extend the VPA-based fishing mortality series as far back as possible. A number of attempts to do this have been made in the past. These attempts were made by scientists whose experience was nearer in time to the data collection than our own, so it seems appropriate to use their results rather than to go back to the historical statistics which they interpreted.

Anon. (1969) provides a number of effort measures for various fisheries in the North Sea which extend back in some cases to 1914. Calibrating these series with the VPA results enables the effort series to be extended forward in time and allows the VPA results to be extended back in time.

Figure 3.3.3.1 shows such extended time series of fishing mortality (F) for cod, haddock and whiting which are caught in the North Sea roundfish fishery. These are based on catch rates obtained by Scottish trawlers applied

to the international catch of each species for the period 1914-1966. The resulting total international effort was converted into fishing mortality and extended to 1989 using VPA results. The area of overlap between the effort and the VPA series is relatively small and there may be problems with calibration. However, there is reasonable similarity between the reconstructed series of fishing mortality for haddock and the Fs from an earlier, independent VPA (Pope, pers. comm.) (see Figure 3.3.3.5). Earlier than about 1930, the two series of Fs for haddock diverge, suggesting that the effort units used in earlier than the 1930s are inconsistent with present units. In order to remove this inconsistency, estimates of levels of fishing mortality for haddock prior to the Second World War were derived from Scottish research vessel catch-at-age data given by Sahrhage and Wagner (1978). These estimates were then used to downscale the fishing mortalities obtained from the effort times series prior to 1930. Assuming that the downscaling represents a change in the fishing power in the roundfish fishery, a similar downscaling was applied to cod and whiting.

The reconstructed series of Fs shows that mortality on haddock and whiting has been high and fluctuating, but without a trend, for most of this century. If the values for the early part of this century are correct, fishing mortality on haddock was as high as, or higher than, has been attained since. Fishing mortality was lower on cod for most of this century, but has increased quite markedly since 1960 and now matches that on haddock and whiting. Fishing mortality on haddock, and to a lesser extent on whiting, is generated in the northern and north central North Sea. Fishing mortality on cod in the North Sea is generated in a less localised fashion.

Fishing effort for the two major species of flatfish, sole and plaice, are also available in Anon. (1969). The estimates of effort directed at plaice are based upon the catch rates of English trawlers per hour's fishing applied to the international catch from 1909 to 1966. These data were converted into fishing mortality using results from the VPA which extends from 1951 to the present and further calibrated using data from Bannister (1978) on average fishing mortality in various time periods since 1929. Fishing effort for sole was derived from catch rates of Dutch otter trawlers from 1946-1966. This series was converted into fishing mortality using the VPA series which covers the year 1957 to the present.

The trends in fishing mortality for sole and plaice are shown in Figure 3.3.3.2. Fishing mortalities in both flatfish fisheries have increased steeply in recent years. This has largely resulted from the introduction of twin beam trawlers in the 1960s, which enabled the use of heavier tickler chains used to stir up flatfish into the net. Such fishing practice, and the resultant levels of fishing mortalities, are mainly associated with the North Sea south of 56°N.

The fishing mortality series for herring (Figure 3.3.3.3) comes directly from the VPA made by the Herring Assessment Working Group for the Area South of 62°N, extending back only to 1947. F increased slowly through the 1950s, and then trebled abruptly to values well over 1.0 from 1964 to 1968, due to redirection of the herring purse seine fleet from Norway (Saville and Bailey, 1980). Fishing mortality stayed at those elevated levels until the fishery collapsed between 1976 and 1978 and fishing was reduced greatly. With the rebuilding of the North Sea herring stock due to strong recruitment, fishing mortality values have recently returned to levels that persisted in the earlier part of the series.

The industrial fisheries started in the early 1950s. In the beginning, herring constituted the main part of the landings, but then sandeel, Norway pout and sprat increased in importance. Norway pout and sprat were mainly important in the 1970s, after which the catches of these two species declined. From 1985 onwards, sandeel has constituted approximately two-thirds of the total catches. Figure 3.3.3.4 shows the trend in fishing mortality on sandeel. The values from 1974 onwards have been extracted from the report of the Multispecies Assessment Working Group (Anon., 1991e) and extended by effort data from Anon. (1977) and Anon. (1978) using the same procedure as for roundfish. The time series show that the fishing mortality increased in the late 1970s to a level of around 0.6. This level has since been maintained, albeit with large year-to-year fluctuations in fishing mortality in the most recent years.

3.3.4 Distribution of fishing effort by NSTF areas in 1989

Data on catch and effort from 49 fleets composed of types defined by nation/gear combinations were made available from the archives of the EC's STCF working group on the improvement of North Sea exploitation patterns. Data were made available by all countries except Belgium. Also, effort data were not available from the Kattegat and Skagerrak (NSTF area 8) and from several gears in the English Channel (NSTF area 9).

The available effort data are in a variety of different units. Most reflect the time spent fishing rather than the time and power used. Therefore, it is not very sensible to combine the raw data as they stand. On the other hand, presenting the data disaggregated by fleet would present an uninterpretable data set. The data clearly need to be intercalibrated and presented by gear. To achieve this, the catch per unit effort (CPUE) for each fleet for each of the NSTF areas and for each quarter of 1989 were used in an analysis of variance (ANOVA). The 49 fleets were classified into nine main gear types: beam trawl, otter trawl, pair trawl, industrial otter trawl, industrial pair trawl, seines and ringnets, pelagic trawl,

purse seine, and others (see Annex 2). For each of the gear types, the CPUE for fleets, areas and quarters was fitted using the model

$$CPUE = F(f) + AQ(a,q) + \text{error}$$

where $F(f)$ is a factor for fleet(f) and $AQ(a,q)$ is an area(a), quarter(q) interaction term. The linear model provides combined estimates of area quarter catch rates, standardized to the units of one fleet. These estimates were then used to divide each fleet's area, quarter catch data, to produce standardized effort values for each gear, area and quarter (see Tables 3.3.4.a-i). The quarterly catch data and further details of the analysis are presented in Annex 2. The choice of which fleet to use as a standard is arbitrary, and naturally cannot be the same among gear types. Therefore, comparisons of values among tables is not valid (different calibration standards), but comparisons within tables are legitimate.

3.4 Changes in Other Anthropogenic Parameters

Nutrients

In some coastal regions of the North Sea, sufficient data exist to identify increasing trends in the concentrations of nitrogen and phosphorus nutrients. However, the timeseries of data do not allow unequivocal analyses to be carried out for all regions. There is no evidence for increasing nutrient concentrations in the offshore waters of the North Sea (Anon., 1990c, 1991f).

Others

No comprehensive timeseries information on a North Sea wide scale was available to the group for other human activities.

4 QUANTIFICATION OF THE DIRECT IMPACTS OF FISHING ACTIVITIES

4.1 Introduction

In this section, the direct effects of fishing are described for a variety of biological sub-systems within the North Sea. The direct effects are as follows:

- a) Fisheries cause mortality on the target fish and incidentally on other biota;
- b) Fisheries make food available to other species in the ecosystem by i) discarding unwanted catch of fish and benthos, ii) discarding wastes, and/or iii) by killing or damaging animals in the path of the gear during its deployment;

- c) Fisheries disturb the seabed by the action of some towed fishing gears;
- d) Fisheries generate litter composed of lost or dumped gear.

It is recognized that estimates of these effects on a North Sea-wide scale, or at least for defined sub-areas within it, are in principle desirable. In practice, however, most of the information available is based on small-scale studies and it is very difficult, with present knowledge, to extrapolate from these to larger scales. One exception concerns the effects of fishing on the fish stocks themselves. For the main stocks of exploited species, estimates are available of the proportion of the stock that is removed by fishing each year either for the whole North Sea or smaller parts of it.

Some of the information used in this Section is derived from studies in areas other than the North Sea. However, such studies are only used where they are helpful in providing indications of the likely effects in the North Sea itself.

4.2 Estimates of Areas Affected by Towed Gears

It has long been recognized that the absolute level of disturbance of the seabed is very dependent on the type of gear employed and the nature of the seabed over which the gear is towed. Thus, light gears such as shrimp beam trawls penetrate the seabed to a lesser degree than heavy beam trawls designed to catch flatfish. Although the precise relationships between gear design, towing speed and bottom type have not been determined for all types of gear, available data allow a qualitative classification of the relative levels of disturbance from each, and identification of the parts of the gear which are most responsible. This is given in Table 4.2.1, which shows that heavy beam trawls for flatfish species can penetrate the sediment 6 cm or more, whereas the different dredges and the doors of the otter trawl can also have a significant penetration depth.

Information such as that provided in Table 4.2.1 gives only a qualitative indication of the relative impact of the various types of gear per unit of fishing effort. This is mirrored by the majority of impact studies which focus on the scale of the individual trawl or dredge track. As a basis for estimating the importance for benthic communities, this is in many ways inadequate.

Estimates of the total area disturbed by fishing have traditionally come from two sources, namely direct observation of visible signs of trawls on the sea bed and analysis of the distribution of fishing effort from fishing records. Using side-scan sonar records from the Kiel Bight (southwestern Baltic), Krost *et al.* (1990) estimated that the most disturbed regions had up to 35% of the area

as visible tracks (mean value 25%). During a survey in the Dutch part of the North Sea, 70% of the research area (9 km²) was covered with trawl tracks (BEON, 1992). As with all estimates of this kind, however, the interpretation of these values is difficult because the persistence of visible tracks is uncertain and depends on the sediment type and current regime.

From the distribution of fishing effort, estimates of the spatial or temporal distribution of fishing disturbance can be made. Churchill (1989), using records from the US Fisheries Service, estimated the distribution of fishing effort for the northeast coast of the United States. These data allowed an estimate of the cumulative area fished annually in separate 30' Latitude x 30' Longitude boxes. At this scale of resolution, the total area fished in some boxes, notably off Long Island and Narragansett Bay, was more than 3 times the actual area. Rauck (1985) estimated that several areas of the North Sea are trawled 3 - 5 times per year. For different ICES statistical rectangles of 30 x 30 nm in the Dutch sector, Welleman (1989) calculated values of 0.5 to 7 times per year.

Using the estimates of area covered in 100 hours fishing in Table 4.2.1 and estimates of fishing effort by gear in Tables 3.3.4.a-i, estimates have been made of the area of seabed impacted by each fishing gear per year within each NSTF area. Using scaling factors to express fishing effort by different fleets in the same units, the values in Table 4.2.1 were applied to the effort scaled to the fleet which corresponded most closely to the characteristics given in the Table. The list of fleets chosen for this purpose is given below:

Beam trawl	Netherlands beam trawl >300 HP assumed to fish 16 hours per day from port
Otter trawl	Scottish trawl (effort in hours fishing)
Ind. Otter tr.	Danish single boat industrial trawl >100GRT assumed to fish 16 hours per day absent
Ind. Pair tr.	Danish industrial pair trawl >100GRT assumed to fish 16 hours per day absent
Pair trawl	Scottish pair trawl demersal (effort in hours fishing)
Danish seine	Scottish seine (effort in hours fishing)

The scaling factors used are given in Table 4.2.2. Separate effort data were not available for scallop trawls and dredges.

In Table 4.2.3, estimates of the area swept per year by those parts of the gears that have a major effect on the seabed are given. These consist of the entire length of the beam trawls and the area covered by the otterboards of otter trawls and industrial trawls. It was assumed that pair trawls and Danish seines do not have a major effect on the seabed.

Although the ground rope of trawls and seines is not expected to have a major effect on the seabed and its fauna, estimates are given in Table 4.2.4 of the total area swept by each gear. It was assumed that pelagic trawls and purse-seines do not touch the seabed.

In the case of area coverage by the ground rope, it should be noted that this is not equivalent to the effective area swept by the gear for demersal fish because the herding effect of the bridles of the otter trawls and ropes of the seines increases the effective swept area for fish considerably.

Table 4.2.5 compares the swept area of beamtrawls and otterboards with the size of each NSTF area. It shows that the percentage covered ($100 \times \text{total area swept per year} / \text{surface of NSTF area}$) ranges from 0.3 to 321% in areas where beamtrawling occurs. NSTF areas 4 and 5 are the most frequently trawled (280% and 150%, respectively). For NSTF areas 1 to 7 as a whole, grounds fished by beamtrawl occupy at most 34% of the total sea area. The areas affected most by otter trawl boards are NSTF areas 5 and 6.

The estimates of total area swept presented in Tables 4.2.3 to 4.2.5 are rough estimates based on the qualitative information presented in Table 4.2.1. They are presented in order to illustrate the scale of physical disturbance caused by fishing and do not take the distribution of effort on various fishing grounds within each NSTF area into account. On a "micro-scale" the surface area affected may therefore be completely different. This was demonstrated by recent attempts to measure the micro-distribution of fishing effort (Rijnsdorp *et al.*, 1991). In this study, the behaviour of individual fishing boats in localized (1 mile x 1 mile) blocks was followed and the results showed that effort is very patchily distributed in space. Extrapolation of the observed distribution of fishing effort using Monte-Carlo simulations which assumed that effort is distributed at random between fishing trips, but patchily within a trip, suggested that of the five ICES statistical rectangles studied, two showed a random distribution of fishing effort, and for the remainder less than 60% of the available area was trawled. This pattern may be explained by the concentration of effort on good fishing grounds and the avoidance of areas where gears may be lost; however, the distribution of unfishable grounds was not presented.

In the North Sea, scallop dredging represents very localized and minor fisheries, whereas mussel and cockle dredging are the major activities in localized coastal areas. Scallop dredging, however, dominates Channel bivalve fisheries. Between 1974 and 1989, the estimated total area dredged by large English scallop boats increased from 132 to 1600 km² on beds in the Channel (ICES Divisions VIIId,e) and from 0 to 90 km² on localized beds in the North Sea (ICES Division IVb) (MAFF, unpublished data). Similar data for areas heavily fished by French scallop dredgers and otter trawlers in the Bay of St. Brieuc (800 km² in area) indicate that 160 km² and up to 5600 km² of the Bay are swept annually by the two types of gear (Hamon *et al.*, 1991). These data suggest that the Bay is swept 0.2 to 7 times per year by the two types of gear, respectively.

4.3 Impacts on Physical Structure of the Sediment and Abiotic Processes

All towed fishing gears which exploit bottom-living species disturb the sediment and hence may have an impact on the structure and processes at the seabed. Properties which may be affected include grain size distributions (including the removal or displacement of boulders), sediment porosity and chemical exchange processes.

Fishing may change sediment characteristics. In the Dutch sector of the North Sea, it was observed that the passing of heavy beam trawl gear changed the median grain size of the top 5 cm of sediment, whereas some minor changes in porosity and reciprocal formation factor (i.e., sediment conductivity) were observed (BEON, 1991). Another direct consequence of fishing is the displacement of boulders which would otherwise offer a primary substratum for many epifaunal taxa. This effect has been noted, particularly with reference to dredge fishing for scallops (Caddy, 1973; Chapman *et al.*, 1977; Fowler, 1989; MAFF, unpublished data; Dupouy, 1982). Also, beam trawl fishing is known to remove or displace boulders.

A direct consequence of disturbance of the sediments is an increase in the suspended sediment load and the possibility that fishing will facilitate the net transport of finer sediments. This effect was studied on the northeast coast of the United States by Churchill (1989), who concluded that trawling could be the primary source of sediment transport over the outer shelf in areas where storm-related bottom stresses were weak. Despite these conclusions, however, trawling did not appear to result in significant short-term erosion of the outer shelf sediments. The resuspension of sediments may influence the uptake or release of contaminants and, depending on the frequency of disturbance, the redox status of the disturbed sediment, and the nature of the contaminant(s). Clearly, such effects may be more significant where

contaminant burdens are relatively high, e.g., near to areas affected by major industrialization.

Another potential consequence of sediment resuspension is a shift in sediment-water exchange, e.g., of nutrients. Depending on sediment type, season, and chemical compound concerned, the net flux may be influenced positively or negatively. For sediments normally releasing nutrients, the disturbance will result in an immediate increase in nutrient release, followed by a period of lower flux until the original nutrient profile of the sediment is restored. The net results on the nutrient exchange may be rather small, although an increase in ammonium or nitrate flux (Riemann and Hoffmann, 1991) may influence the denitrification rate. It should be pointed out that repetitive trawling on the same ground may enhance nutrient release from sediments and that estimates of average trawling effort for large areas may be unsuitable for estimating these effects.

Nutrient release by otter trawl disturbance has been studied by Krost (1990) in the Kiel Bight (southwestern Baltic), who calculated the additional phosphate and ammonium input to the water-column resulting from this type of activity at 33-167 and 96-435 tonnes per year, respectively (see also Section 6.1). On the other hand, the oxygen penetration into the sediment might be enhanced by the fishing activity, resulting in shifts in mineralization patterns and redox-dependent chemical processes. Among other consequences, a change from anaerobic to aerobic conditions facilitates the degradation of hydrocarbons. Finally, reworking of the sediments may result in burial of organic matter.

4.4 Mortality

4.4.1 Benthos

In Table 4.2.1, the bottom area in contact with various types of fishing gears was calculated by estimating the area affected per 100 hours fishing. This was achieved using the expected length of a tow and the width of the relevant parts of each gear.

Gears can be subdivided into those which penetrate the sediment to a marked degree, thereby affecting infaunal taxa, and gears for which effects are probably restricted to epifauna. Mortalities of infauna might occur, for example, from damage by tickler chains, the teeth of scallop dredges, or the doors of otter trawls. Ground ropes of ottertrawls rigged without chains will probably only affect epifauna. For shrimp trawls, pair trawls, Danish and anchor seines, there is little penetration of the gear into the seabed and the main effect is on epibenthos, either as the gear passes or by capture with consequent damage in the cod-end or on deck. The quantity of epibenthos that is brought on board can be minimized when the ground rope is rigged with rollers or

other devices to keep it clear of the bottom. Fixed gill and tangle nets have minimal effects on benthic taxa, with the exception of crabs which become entangled.

The gears which disrupt the sediment most are beam trawls and shellfish dredges. Again, the method of rigging the gear can have a profound effect on the level of disturbance and, in the case of the beam trawl, there is a clear positive relationship between the number of tickler chains used and the biomass of benthos caught (de Groot and Apeldoorn, 1971; de Groot, 1984; Creutzberg *et al.*, 1987). From Table 4.2.5 it appears that benthic mortalities caused by the otter trawl boards are likely to be at least one order of magnitude lower than those caused by beamtrawls, but overall, Table 4.2.5 shows that the benthos of large sectors of the North Sea is potentially vulnerable to trawl-induced changes. The results of quantitative studies which have examined the effects of beam trawls and scallop dredges are summarized below.

For 12-metre beam trawls, the most comprehensive data available on effects on the benthos are provided in two reports by the Dutch BEON group (BEON, 1990, 1991). Survival rates for infauna and epifauna caught in the net were between 70 and 90% for starfish, up to 50-90% for many molluscs, and 40-60% for crab species. However, survival rates of only 10% were recorded for the large bivalve *Arctica islandica*. Trawl-caught whelks and hermit crabs were largely unaffected. The presence of benthic taxa in beam trawl catches in other studies indicates that other species, such as *Tubularia*, *Lagis*, *Ensis*, and *Solen* may also be affected (Graham, 1955; Bridger, 1970; Houghton *et al.*, 1971; de Groot and Apeldoorn, 1971; Margetts and Bridger, 1971; de Groot, 1973). Comparison of box core samples taken before and after trawling indicated that there were extensive changes to the infauna which remained after the passage of a 12-metre trawl. A significant reduction in the numbers of burrowing sea urchin *Echinocardium* (a 15% reduction for large and 55% reduction for small individuals) was observed and the density of tube-building polychaete species was also reduced by 50% (BEON, 1990). Taken together, these limited results suggest that in the path of a beam trawl a relatively high proportion of some benthic species can be killed.

Otter trawling may have an important impact on non-target commercial species. For example, in the Western Channel newly moulted spider-crabs and scallops may be killed or damaged (Hamon *et al.*, 1991).

In common with beam trawl studies, there have been relatively few quantitative studies of the effects of scallop dredges on benthic communities. Early studies on scallop dredging concentrated on documenting the nature of the physical disturbance, and on qualitative analysis of the effects on the epibenthos in the by-catch and at the sea-

bed. Epibenthic mortalities can be marked. Substantial mortalities may include specimens of the target (commercial) species, which are left injured on the seabed or discarded (Caddy, 1973; Chapman *et al.*, 1977; Dupouy, 1982). Scallop grounds tend to occur in relatively deep water areas (30 - 70 m) with high species diversity, much of which may be associated with rocks and boulders in the area. Dredging in such areas has been implicated in reducing species abundances in these areas (Holme, 1983), especially of the sessile species which occupy the hard rock substrates (Bullimore, 1985).

At least nine species of bivalve molluscs are harvested from diverse habitats in the North Sea and Channel, by a range of specialized traditional and modern dredges (Table 4.4.1). Their effects on the seabed and benthos depend on substrate type, hydrographic features, and community structure, as well as upon the particular design and operation characteristics of the gears. Table 4.4.1 provides a summary of the dredge fisheries and an indication of the levels of their likely impact. Two main gear classes can be recognized: (a) non-hydraulic or traditional dredges, including several new designs; (b) hydraulic dredges which use water-jets to extract burrowing species from the sediments. However, with the exception of cockle hydraulic dredging, information on dredging effects is fragmentary, superficial, descriptive, and even anecdotal.

Apart from affecting mortality, dredging and trawling may contribute to the dispersion of some introduced "pests" (e.g., *Crepidula fornicata* in the English Channel), either directly through transfer by gears along the seabed or indirectly as by-catch discards.

4.4.2 Fish

In recent years, the landings of fish in the North Sea have been in the order of 2.5 - 3 million t. Very approximately, this can be broken down into 600,000 t of demersal fish (gadoids and flatfish), 1,000,000 t of pelagic fish (herring and mackerel) and 1,000,000 t of species landed for reduction to meal and oil (Norway pout, sprats and sandeels). To a first approximation, these landings are taken from a biomass of the same species in the order of 7-8 million t (Anon., 1991e).

Estimates of mortality rates of the main exploited species in the North Sea are made regularly by ICES working groups and published in their reports. They are also summarised in the reports of the Advisory Committee on Fishery Management (ACFM), the most recent of which is that for 1991 (Anon., 1992c).

A summary of recent fishing mortality rates for the main species exploited in the North Sea is given in Table 4.4.2. These are given for the age groups most heavily exploited and for juvenile fish. They are given in terms

of instantaneous rates of fishing mortality (F), which is related to the fishing effort on the stock. The mortality rates are also expressed as the percentage of the stock in number alive at the beginning of the year that is caught by fishing during the year. These two values do not bear a simple relationship to each other because the percentage caught also depends on the natural mortality rate which differs among species and age groups.

In general terms, fishing mortality rates on the main North Sea stocks are high, although it should be noted that they are in some stocks very variable from year to year (see Anon., 1992c). In the case of the two roundfish species, haddock and whiting, for which discarding is thought to account for a significant part of the catch, the estimates given include the mortality of fish discarded. For some other stocks in which discarding is known to occur, however, discards are not routinely included in the estimates of fishing mortality. The extent of discarding is dealt with in Section 4.5.

Apart from the mortality generated on juveniles and adults, various towed gears in contact with the bottom may damage fish eggs attached to the seabed (e.g., herring). Some spawning beds are closed to fishing during the spawning season.

4.4.3 Seabirds

Seabirds are capable of becoming entangled in most types of fishing nets, but more birds are killed in some types than others. Birds have been reported killed particularly in gill and other static nets. There have been no comprehensive studies of entanglement in the North Sea, but the available evidence indicates that in those areas that support both fixed net fisheries and diving seabirds, entanglement is likely to occur.

There have been quantitative studies on the effects of gill net fisheries at a local scale. In the English Channel, inshore gill nets have proved to have particularly high by-catch at some times and places. Robins (1991) showed that these nets, set for bass, have caught virtually all of the diving birds (mostly razorbills *Alca torda* and divers *Gavia* sp.) using St. Ives Bay. For instance, in eight fishing days in January 1988, about 900 auks were removed dead from these nets. The numbers caught at this single site are not significant at the North Sea population level, but this situation is probably typical of sites where diving seabirds are common and netting is undertaken. An investigation by the Royal Society for the Protection of Birds in the UK in the winter of 1991/1992 indicated that most entanglement events are sporadic and localised (ref to come). The extent of usage of gill nets on other English North Sea coasts has not been quantified as many of these nets are deployed from vessels less than 10 m in length (which are not required to complete logbooks); however, several thousand kilo-

metres of monofilament gill net are imported into England and Wales each year. There is some qualitative information on the distribution of gill net fisheries around England and Wales (Figure 4.4.3; Potter and Pawson, 1991). Monofilament gill nets are not in legal use off Scotland, though there is some illegal use. The impact of these illegal nets on seabirds is not easily quantified. There is widespread belief that monofilament gill nets are a particular problem (Mead 1989), but seabirds are taken in all types of gill nets.

Oldén *et al.* (1988) estimated that 25,000 diving seabirds were killed by drowning in herring and cod gill nets in the southeast Kattegat between 1982 and 1988. The majority (90-95%) of these birds were guillemots, and had migrated from populations breeding on the Scottish North Sea coast. Most of the fishing activities here are for herring, but most birds were found in bottom-set cod nets with a mesh size of 150 mm. Herring nets were of 55 mm mesh and were set either on the bottom or near the surface. These tended to catch cormorants.

In the northwestern North Sea, nets set for salmon near seabird colonies have trapped and drowned auks (Melville, 1973). Guillemots *Uria aalge* have been recorded being caught in nets being trawled for sandeels on the northwestern Dogger Bank (M.L. Tasker, pers. obs.). Galbraith *et al.* (1981) found that shags (*Phalacrocorax aristotelis*) were caught in lobster traps, but most of these birds were recently independent juveniles that had failed to learn the skills of foraging. The overall extent of these mortality sources is unknown.

There is only limited gill netting off the Netherlands, Germany and the North Sea coast of Denmark. There is little information available on gill netting off Belgium. Off France (English Channel), there is a fishery for bass, pollack, cod and sole undertaken by approximately 600 fishing boats over 4300 boat/months (1990) (IFREMER, unpubl. data). There have been no studies of entanglement in this fishery. In Norway, salmon drift netting was associated with high levels of by-catches in the 1980s, but this form of fishing was terminated in 1989. Returns of leg-rings from auks found dead indicate that birds off the coast of Scandinavia have been killed in large numbers by net entanglement. The proportion of birds caught this way in relation to other methods of recovery (e.g., shooting, oiling) increased substantially in the 1980s (Mead, 1989) (Table 4.4.3).

Outside the North Sea, severe episodes of entanglement have been documented (e.g., in the southeastern Kattegat, northern Norway, southern Pacific, off California, west coast of Ireland, southwest Greenland) and have been implicated in substantial declines in bird populations in some areas (Evans and Waterston, 1978; King, 1984; Piatt *et al.*, 1984; Whilde, 1979). However, in only one case among eight listed worldwide by Robins (1991),

could breeding seabird population decline be attributed with certainty to entanglement. Inadequate investigative resources contribute to the lack of scientific proof.

4.4.4 Marine mammals

An ICES Study Group on Seals and Small Cetaceans in Northern European Seas was established in 1991 and has now met on two occasions. Included among the terms of reference for the most recent meeting, in March 1992, were requests that the Study Group "assess trends in, and the current status of, seal and coastal dolphin populations in the North Sea". The Study Group was also asked to "evaluate available information on the by-catch of those species, the extent as well as type of fisheries involved".

It was generally agreed at this Study Group that incidental catches of marine mammals in fishing gear in the North Sea were poorly documented. Existing data had mostly been collected on an opportunistic basis, and were therefore not representative, and could not be used to estimate the total number killed by fishing operations. Even if the total number killed could be estimated, the level of mortality could not be accurately assessed because of the uncertainty regarding true population sizes.

4.4.4.1 Seals

Both seal species may become trapped in towed gears, but most seal by-catch mortality is attributed to fixed type gears (Anon., 1992d). Tags from grey seals (*Halichoerus grypus*) tagged in Norway have been recovered from pups entangled in gill nets (Wug and Oien 1987; 1988) and common seals (*Phoca vitulina*) have been recorded trapped in fyke nets in Dutch coastal waters (Boer, 1989). In the U.K. seals of both species have been drowned in fixed nets set for salmon or have been shot by netsmen protecting their nets and catch (Rae, 1968; Pierce *et al.*, 1991).

4.4.4.2 Cetaceans

Table 4.4.4 gives a summary of the results from some studies of cetacean by-catch mortality in the North Sea. These studies indicated that gill nets killed the most cetaceans, and that among gillnet-caught porpoises, most were taken in large-mesh nets. Catch rates varied seasonally and, in German waters, animals were mainly taken at night. In the Danish study, incidental catches were mostly of young animals.

Around the British Isles, several species of small cetaceans have been reported as incidental catches, but in the North Sea, reported by-catches of species other than the harbour porpoise (*Phocoena phocoena*) are rare. Porpoises are taken in various types of gear, but most by-catch records came from coastal gill nets, except

around Shetland where all reports for which the gear was known came from whitefish trawls. A voluntary scheme for fishermen to report catches to local fisheries officers has been in place in England and Wales for a number of years, but reports have been rare. Over the last 10 years, Scottish scientists have made more than 600 voyages on commercial fishing vessels and not a single instance of a cetacean becoming entangled in the net was recorded (Anon., 1992d).

By-catch has been recorded in the Netherlands for the last 6 to 7 years (IWC, 1991). Small numbers were recorded in most gear types used by Dutch vessels in the North Sea (Anon., 1992d).

There is no systematic procedure for recording by-caught cetaceans in Norway. Small cetaceans were known to be caught in some fisheries, especially the salmon drift-net fishery which has now been closed. In 1988, 96 porpoise carcasses were bought from salmon-netters; about 35 by-caught porpoises were bought from the traditional fisheries in 1989 and 1990 (Björge *et al.*, 1991; Anon., 1992d).

Harbour porpoises were trapped in Pound nets in Denmark in the 1960s, but the use of these nets has declined in recent years. Surveys conducted between 1979 and 1991 indicated that considerable numbers of porpoises were still caught; few were taken in trawls, the majority were entangled in bottom-set gill nets. Exact figures for the total by-catch are not available, but estimates suggest that it is more than 1000 per year and could run to several thousand (Anon., 1992d). About two-thirds are caught in the west and north of Denmark and the rest in Danish inshore waters.

4.5 Non-catch Mortality and Dumping of Discards and Offal

Non-catch mortality

As well as the catch, fishing operations cause incidental mortality of fish which escape from the gear. In the case of most trawling gears and Danish seines, fish are herded in front of the net; some escape before entering the cod-end and some, either voluntarily or involuntarily, pass through the cod-end meshes. Of those retained, some are lost in the act of hauling the net and, in the pelagic fisheries, whole catches may be released before being taken on board. Of the fish taken on board, a portion may be discarded. Of those retained, either the entire fish may be landed, or the fish may be gutted and/or processed (e.g., filleted) and the resulting offal discarded. Corresponding considerations also apply to encircling gears and to trapping, netting and hooking gears.

Reliable data series exist for landings and in some cases discards, but there is little quantitative information on the

incidental mortality rates caused to the stocks as a result of mortality of fish that escape from fishing gear. Nevertheless, a number of studies have been carried out on the survival rates of fish that escape from fishing gear during the catching process.

Fish that escape from towed gears incur scale loss which can be caused both by abrasion against the meshes and by abrasion against objects caught in the trawl itself (Main and Sangster, 1990). The extent of scale loss differs between species and varies according to the mesh size, the type of meshes (square or diamond) and the rig of the gear. However, some scale loss appears to occur in all fish that escape from cod-ends, at least those above a certain size in relation to the mesh size in use (e.g., all fish over 18 cm in length from 70-90 mm mesh trawls).

The possible causes of death include loss of osmoregulatory ability (from scale loss), internal organ damage (from pressure within the cod-end) and secondary viral or bacterial infections (from skin damage). There are reports of diseased fish in which ulcerations were thought to have been the result of damage by nets (Mellergaard and Nielsen, 1990). Fish with net injuries appear to be more susceptible to lymphocystis than fish without injuries (Dethlefsen *et al.*, 1987). There is also some evidence that stress due to the capture process may increase mortality (Black, 1958).

Based on experiments in which escaping fish were caught and retained in cages *in situ*, mortality rates of gadoids ranged from 6-33%, depending on the type of mesh and mesh size in use (Main and Sangster, 1991). Differences in scale loss and mortality also exist between species, haddock being more susceptible than cod (Isaksen, 1991; Engas *et al.*, 1990). The mortality rate of fish in a net also depends on haul duration, catch quantity, and catch composition. In the German shrimp fishery, Von Kelle (1976) found a direct relationship between towing time, total catch weight and the survival of small sole, dab and plaice. The survival rate of undersized flatfish was 51% for plaice, 57% for sole, and 26% for dab. Survival rates decreased when large quantities of jellyfish (*Cyanea* and *Pleurobrachia*) were present in the net.

In the case of beam trawling, the reports available indicate rather variable survival rates of small fish that escape. In one set of experiments, most small fish apparently escaped through the meshes of a commercial trawl fairly undamaged. At least 56% of dab (*Limanda limanda* L.), 85% of plaice, 100% of sole (*Solea solea*) and 68% of dragonet (*Callionymus* spp.) and solenette (*Microchirus boscanion*) which escaped from the cod-end into a cover survived the first 24 hours after capture (BEON, 1990). In another set of experiments, the survival of soles that escaped through the meshes was estimated at 60% (Van Beek *et al.*, 1989). Deaths were

attributed to the action of the tickler chains and the injuries inflicted while in the net.

There appears to be no information on the survival of fish that escape from gill nets.

Survival experiments on pelagic fish are less easy to interpret because of the difficulty of maintaining them in cages. Reported survival rates were around 60% for herring escaping through square mesh panels (Suuronen, 1991), but mortality was also recorded among controls, so the true survival rates may have been higher than this. In experiments on Baltic herring escaping from diamond mesh cod-ends, reported mortality rates were 3% (Treschev *et al.*, 1975) and 10-15% (Borisov and Efanov, 1981). Lockwood *et al.* (1977) investigated the survival of mackerel after escape or release from a purse seine. It was found that 50% of the fish died after 48 hours at a stocking density of 30 fish m⁻³. Trials with fish densities comparable to those experienced in a "dried up" purse seine prior to "slipping", showed that up to 90% of "slipped" fish died within 48 hours of release.

The only direct observations of non-catch mortality caused by fishing are those of Zaferman and Serebrov (1989) who used an underwater submersible to make observations on escapes of cod and haddock from a 100 mm diamond mesh cod-end in the Barents Sea. After hauling the fishing gear, dead haddock and cod were seen lying on the sea bed in the path of the trawl.

From the data available, it is not possible to reach firm conclusions about the percentage of escaping fish that survive. For each main method, however, the range of survival rates reported and the period over which the fish were kept, where known, are shown in Table 4.5.1.

In all the studies summarised above in which the experimental period after capture was short, there is also a measure of uncertainty about the ultimate level of mortality.

As a very rough approximation, it might be assumed that in the order of 75% of fish that escape from towed gears survive. In itself this provides no estimate of the additional unrecorded mortality caused by capture and escape, because this also depends on the quantities of fish that escape from the cod-end. In general, the fish that are damaged in a net will be in a fairly narrow size range because very small fish will tend to escape through the large meshes in the front parts of the net often without damage. In a series of covered cod-end experiments reported by Robertson and Ferro (1988), the ratio of fish in the cod-end cover to fish in the cod-end itself was 1.2:1 for haddock and 0.4:1 for whiting. Assuming a ratio of 0.8:1 and a survival rate of 75%, this implies that, in this experiment, the additional non-catch mortality was about 20% of that attributable to the catch. The

additional mortality will vary considerably in different situations depending on the precise specification of the gear in use in relation to the size and species composition of the fish that are accessible to the gear.

The fate of dead and moribund fish in the short-term has not been extensively studied. In the more usual case of fish escaping gradually from a trawl, it seems unlikely that the density of dead fish in any one area will have anything more than a transient effect. It is also far from clear whether fish that ultimately die as a result of net damage add to known causes of natural mortality (e.g., predation) or whether they replace other fish in the diet of predators. On average, therefore, it seems likely that non-catch mortality adds a small but significant contribution to the overall mortality of fish. There are certain gears and areas, however, in which non-catch mortality is likely to be somewhat higher.

Discards

Discarding of fish during commercial fishing activities in the North Sea has been recorded since at least the 1920s and data exist on the proportion of fish discarded in a number of fisheries. In earlier periods, percentages of roundfish discarded in the trawl and seine fisheries were reported to be 30-40% in the 1920s and early 1960s. In the late 1970s, discarding rates of haddock and whiting were estimated to be about 14% in trawl and about 20-30% in seine catches (Jermyn and Robb, 1981). Regular sampling of the Scottish fisheries, mainly in the northern North Sea, has provided data on discards. The ICES Roundfish Working Group has used these data to estimate total discards of these two species by all countries in their assessment of the stocks of these two species (Table 4.5.2).

Representative discard data are difficult and expensive to collect and insufficient information exists to allow a quantification of the total amount of fish of all species discarded annually. However, some data were provided by Denmark, the UK and the Netherlands for seine-net, otter trawl and beam trawl fisheries. These data are summarised in terms of percentage discarded by category in Table 4.5.3. If such data are extrapolated to the total catch in the demersal fisheries, they suggest that the total weight of fish discarded in the North Sea annually is in the same order of magnitude as the landings from these fisheries. However, it must be stressed that extrapolation to the entire catch is problematic because of the small number of samples for any time period and area and the consequent danger that samples for one area may have to be used to estimate discards in other areas where the size and species composition of the catch may be quite different. Discard rates in the plaice fishery, for example, can vary considerably between inshore and offshore areas because of the different size range caught.

Using Scottish discard data and other data obtained in the area of the Shetland Islands, Furness *et al.* (1988) used observed discard rates to estimate the total quantity of fish discarded around the British Isles. For the North Sea and Channel, they estimated a total quantity of about 90,000 t per year in the whitefish (i.e., demersal) fisheries. The percentages given in Table 4.5.3 suggest that the total quantities are higher than this.

Studies on discarded fish indicate rather variable survival rates. However, these studies are probably of rather little relevance because a high proportion of discards may be eaten by scavenging organisms, e.g., seabirds (see Section 6.4).

The quantities of fish offal discarded into the North Sea from vessels fishing for roundfish and *Nephrops* have been estimated by Furness *et al.* (1988). Estimates of offal as a proportion of total fish weight range from 6.5-14.3%, with a mode at around 12.5%. Applying this percentage to demersal fish landings of around 450,000 t in the North Sea gives an estimate of 56,000 t of offal discarded annually. This makes no allowance for the relatively small amount of fish landed whole. This estimate compares with an estimate of 84,000 t based on much larger landings of roundfish made by Furness *et al.* (1988).

Some data are also available on the discarding of benthos (Table 4.5.3). In general, by-catches of benthos and inorganic debris are made mainly in gears that fish very close to, or dig into, the seabed. A further study based on limited sampling in the southern North Sea indicated that catches by beam trawl contained 35% benthos of which, depending on the species, 7 to 100% was alive on return to the sea (BEON, 1991).

4.6 Lost Gear and Other Fisheries Debris

Ghost fishing

It is known that gill nets, tangle nets, and traps may continue to fish for some time after being lost or discarded. The term "ghost fishing" is used to describe this phenomenon.

The length of time that such gill and tangle nets continue to fish depends on a number of factors, such as the current speed, the amount of fouling weed in the water, the rate of other marine fouling, the amount of fish caught, and the presence of crabs; these are all things which cause the nets to collapse to the bottom and cease fishing (Millner, 1985). In areas relatively free of fouling, the nets may continue to fish at some reduced level of efficiency until the build-up of fish and crabs forces this collapse. Once on the bottom, multifilament nets remain tangled, while monofilament nets may, once clear of fish remains and crabs, disentangle, return to an

upright position, and resume fishing. Over the longer-term, such nets gradually build up an encrusting layer of marine organisms and become more visible to fish.

The Study Group had only limited information on the abundance of lost or "ghost" fishing gear and none on mortalities resulting directly from such gear in the North Sea. Some information on both abundance and related mortalities was, however, available for areas off Norway and eastern Canada. This information came from three unpublished reports of the Norwegian Directorate of Fisheries (Misund, 1990, 1991; Kolle, 1991) and a summary report of as yet unpublished Canadian Department of Fisheries (DFO) data. While the degree to which such information may relate to conditions in the North Sea is uncertain, the findings do provide some insight into the abundance and fishing behaviour of "ghost" fishing gill nets.

The Norwegian reports indicate the capture, using towed grapnels, of large numbers of "ghost" gill nets in two separate areas off the Norwegian coast. It was observed that old nets were still fishing and that in some there was "a relatively large amount of fish". Nets lost in 1983 continued to fish, as evidenced by both boney remains and recently caught fish. More fish were observed in nets found on soft bottoms than in those over harder substrates. Nets found in deeper waters also contained more fish.

In the Canadian study, an area along the 50 fm isobar on the northern edge of Georges Bank was fished using grapnels. Long-liners and trawlermen had complained of ghost-fishing gill nets in this area. Eight percent of the 236 tows resulted in the recovery of 19 gill nets. The remains of 94 fish (cod, hake, dogfish, and unidentified skeletons) were found in the nets.

The Canadian study also provided preliminary information on the length of time various types of fish remain in gill nets once caught. Two experiments were carried out. The first indicated that the time required for scavengers to consume all the flesh of entangled fish (residence time) ranged from 1-5 days (mean = 2). In the second, residence time ranged from 2-12 days (mean = 6). No correlations were evident between residence time and water temperature or with location in the nets. A further experiment will explore the apparent relationship between amphipod densities and the residence time of the captured fish.

Other impacts

Fragments of lost and discarded nets of all types may also entrap both seabirds (Hartwig *et al.*, 1985; Schrey and Vauk, 1987) and marine mammals (Shomura and Yoshida, 1985). About 3% of all live gannets observed at Helgoland were entangled in fragments of fishing gear

and 29% of dead gannets found there had died of entanglement in plastic, including fishing net (Schrey and Vauk, 1987). Most nests at Flamborough Head and in the Channel Islands (N.M. Harrison, M.G. Hill, pers. comm.) and 90% of the nests at a colony off Newfoundland (Montevecchi, 1991) contained fishing gear. Birds have been observed to die by entanglement in gear at their nests, but there has not been a comprehensive survey of this mortality.

Unpublished data from a MAFF research survey (Cirolana 7/91) indicated a catch, using a Granton trawl, of 0.528 kg/hr of fishing nets or netting (type unspecified) (Pope, pers. comm.). This represented 24% of the total "rubbish" catch. A report on macro-pollutants on the Dutch continental shelf (Ruiten, 1991) lists fishing gear and nets, but provides no quantitative information as to their abundance. Another paper, based on visual observations, describes the overall and relative distributions of floating marine litter in some areas of the North Sea (Dixon and Dixon, 1983). Nylon netting and rope represented the least frequently encountered of the five categories of debris listed, the others being metal and glass, man-made wood items, paper and cardboard, and plastics. As most lost nets would be submerged, their frequency of occurrence must be much greater.

Further evidence as to the extent of lost fishing nets with the potential to cause mortalities comes from a Japanese survey on the distribution and density of marine debris in the Pacific. In this survey, approximately 217 fishing nets were visually observed over a track of 220,000 nm (Anon., 1991g). Again, their frequency of occurrence must be much greater than can be observed on the surface. An Australian study, on the rate of accumulation of fishing and other debris on the shores of Heard and Macquarie Islands in the Southern Ocean, found that over a 12-month period 3 nets, or parts of nets, washed ashore on Heard Island (Slip and Burton, 1991). Based on the length of shore surveyed, this amounts to one net for every 0.12 km of shoreline. Heard Island is relatively close to a fishing area. On Macquarie Island, which is far removed from any fishing area, no nets were found.

5 INFLUENCE OF ANTHROPOGENIC ACTIVITIES OTHER THAN FISHING

Anthropogenic activities other than fishing also affect the North Sea. In this section we have concentrated on describing those which mimic the effects of fishing on the biota and those which exclude fishing operations.

Areas of hypoxia have, for example, resulted in the emigration of fish and other motile biota and the mortality of more sessile biota (e.g., benthos and shellfish); these effects may initially result in elevated catches of benthos and shellfish in trawls before mortality occurs,

while hypoxia-caused emigration and mortality may mimic some of the results of overfishing. Offshore structures (e.g., wrecks and platforms) locally exclude fishing operations and thus provide refuges and sites of increased biomass and diversity for fish and benthos. Point sources from oil production and organic/sewage enrichment alter benthic communities in the vicinity, frequently resulting in a dominance of short-lived, opportunistic species; this is basically comparable with the effects of certain types of particularly intense bottom trawling.

5.1 Eutrophication and Oxygen Depletion

Eutrophication is *sensu stricto* nutrient enrichment (Anon., 1990b), e.g., an increase in the amounts of nitrate and/or phosphate in an area. Sources of such nutrients are commonly sewage (treated or untreated), atmospheric deposition of nitrogen and phosphorus compounds, leaching from farmlands and forests, and waste products from aquaculture/mariculture. Under undisturbed conditions, phytoplankton production is linked to nutrient input in such a manner that, although there are seasonal and annual variations, there is no net increase in total primary production. In addition, the species composition will, within certain bounds, remain constant.

Under eutrophic conditions, an increase in primary production occurs, with the result that excessive sedimentation of dead/decaying phytoplankton cells to the bottom is noted. Under conditions of poor water renewal/exchange (e.g., low current speeds), organic material rapidly accumulates on the bottom, providing a high oxygen demand associated with its metabolism. These situations may result in oxygen depletion (*hypoxia*) or even total lack of oxygen (*anoxia*) in near-bottom waters or within the sediment itself. This may lead to the absence of species of zoobenthos/shellfish and bottom-living fish through active migration from the area. Sessile species (e.g., *Echinocardium cordatum*, and various bivalves) may well die through lack of oxygen and/or the build up of toxic H₂S. A shortage of benthic animals may also make these areas poorly suited for benthic-feeding fish.

Certain types of disturbance of the bottom sediments, especially where these are relatively poorly oxygenated, may lead to the liberation of H₂S and organic material such that an even greater reduction in oxygen levels occurs. Such types of disturbance can be caused by dredging, deeper forms of trawling, etc. (see Table 4.2.1). Bacteria living in the absence of oxygen thrive on sulphates and other compounds. Special bacteria can also convert metals in sediments, releasing them into the water column. Inorganic mercury can be converted into methyl mercury, which is readily taken up by various marine animals, e.g., fish.

Regarding evidence for eutrophication in the North Sea, trends showing increases in nutrient levels are generally restricted to shallower water areas, for example, the German Bight. Such areas are characterized by both relatively restricted water exchange/renewal and enhanced nutrient inputs from land-based sources. Studies conducted in the German Bight demonstrated a doubling of phytoplankton biomass, as well as changes in species composition from diatom-dominated assemblages towards flagellates; increased winter levels of nutrients were also apparent (Radach and Bohle-Carbonell, 1990).

5.2 "Unusual"/"Noxious" Algal Blooms

In the scientific sense, a "bloom" consists of rapid growth resulting in a high biomass of one or more algal species. Often, however, the "blooms" that attract public attention (through toxicity, foam production, etc.) are not blooms at all in the true scientific sense, but merely the presence of a "harmful" algal species within the phytoplankton community. It is, for example, not certain that the widely publicized occurrence of *Chrysochromulina polylepis* in the Kattegat/Skagerrak in May-June 1988 (Nielsen and Richardson, 1990, Bokn *et al.*, 1990) was a bloom in the scientific sense of the word. Although this species dominated the phytoplankton community at the time, the distribution of algal biomass was quite typical for this area and time of year. Thus, the only "exceptional" aspect to this "bloom" may have been the presence of a toxic species.

It is, however, well known that some toxic/noxious species exist within the phytoplankton community as a whole (Reid, 1990). Many historical examples of the presence of toxic phytoplankton exist that cannot be related in any way to eutrophication or pollution. Thus, there is no evidence that the existence of toxic algal species, in itself, can be directly related to eutrophication or pollution.

The toxins associated with "blooms" can be produced by algae. Table 5.2 provides a list of the diseases, the implicated marine foods, and the toxin-producing micro-organisms. It is clear that in some areas there is an increase in the frequency of occurrence of toxic/noxious algal blooms. In some cases, this increase may partly be "apparent" in the sense that there is more awareness and monitoring (due to mariculture interests) directed towards phytoplankton "blooms". However, in other cases, the increases seem likely to be real.

Nutrient enrichment is one of the prominent factors, directly or indirectly, associated with this increase (Wallentinus, 1990). Acknowledging the presence of toxic/noxious species and non-toxic species within the total phytoplankton community, it must be assumed that the growth of both toxic/noxious species and non-toxic species will be stimulated by eutrophication and the

occurrence of "unusual blooms" will thus increase (Anon., 1990b). In some coastal zones of the North Sea, the duration and maximum biomass reached by blooms of *Phaeocystis pouchetii* has increased over the last 20 years, with the associated nuisance of the formation of foam accumulating near shores. These blooms are related, amongst other factors, to high concentrations of nutrients (Lancelot, 1990).

Regarding the factors which lead to the occurrence of a particular algal species at a particular time and place, the list is long (Reid, 1990) and almost certainly incomplete at the present time. A list of these factors includes salinity, temperature, current speed and direction, light regime, turbulence, nutrient (including trace metals and micro-nutrients) availability, the ratio between the concentrations of different nutrients, and the presence of specific grazers (selecting/retaining particular size fractions of phytoplankton cells and species). The composition of the algal flora and the probability of the occurrence of toxic species in the sea at any given time will be a function of complex ecological interactions. The situation in the sea, compared to a terrestrial analogue, is complicated and exacerbated by the short generation time of phytoplankton relative to land plants, and the fact that phytoplankton are dispersed and transported by water movements. Thus, at the present time, it is not possible, as a rule, to identify the overriding factors determining the presence or absence of toxic/non-toxic algal species at any given time/place in the sea. However, some coastal areas appear to be characterized by a relatively frequent occurrence of some "noxious" algal blooms. In some cases, this may be related to the hydrographic characteristics of these regions.

Exceptional algal blooms have previously caused problems for fisheries, aquaculture, tourism, and recreational interests (Anon., 1992e).

5.3 Effects of Eutrophication on Benthos

Changes in the benthic macrofauna under increasing inputs of organic substances, such as may arise from the sedimentation of enhanced quantities of phytoplankton cells as an end product of water column eutrophication, can be illustrated by reference to the 'enrichment' model of Pearson and Rosenberg (1978). As the model has a bearing on other topics in this Section, a brief description is appropriate. Its relevance to generic hypotheses concerning biological responses to 'disturbance' are mentioned in Section 5.5, below; see also Rees *et al.* (1991).

The sequence of events can be summarised in terms of species/abundance/biomass curves (Figure 5.3). At low input levels, there is a general enhancement in the values of these variables. However, as inputs increase and organic matter builds up within sediments, significant

structural change in the community is induced. This phase is characterised by a proliferation of small-sized and short-lived 'opportunistic' species (typically polychaete worms) at the expense of larger, longer-lived and deeper-burrowing species. Accompanying, and partly explaining, this change is a shallowing of the Redox Potential Discontinuity layer, with the result that the fauna becomes increasingly concentrated into the surface layers of sediment. Further accumulation, leading to anoxia even at the sediment surface, results in the elimination of all macrofauna species. It will be noted that this model is equally applicable to trends in space and time.

Regarding evidence for eutrophication in the North Sea, trends showing increases in nutrient levels are generally restricted to shallower water areas, for example, the German Bight. Such areas are characterised by both relatively restricted water exchange/renewal and enhanced nutrient inputs from land-based sources. Studies conducted in the German Bight demonstrated a doubling of phytoplankton biomass, as well as changes in species composition from diatom-dominated assemblages towards flagellates; increased winter levels of nutrients were also apparent (Radach and Bohle-Carbonell, 1990).

Apparent increases in benthic biomass, reduction in species diversity and elevated abundances of certain 'opportunistic' species, compared with earlier surveys, have been cited as evidence in support of eutrophication of parts of the southern North Sea (e.g., Duineveld *et al.*, 1987; Hickel *et al.*, 1989; Kröncke, 1990; Rachor, 1990), and perhaps more noticeably in the relatively quiescent conditions of parts of the Skagerrak/Kattegat (e.g., Pearson *et al.*, 1985; Rosenberg *et al.*, 1987). The latter two surveys involved comparisons with surveys at the turn of the century. Josefson (1990) provides support for the view that observed changes may be a recent phenomenon, in parallel with increases in nutrient inputs to the system. It is notable, however, that a number of the above papers cite intensive trawling at the seabed - the disturbing effects of which may be analogous to those of organic enrichment - as a possible confounding factor.

Buchanan and co-workers have reported on bi-annual sampling of an *Amphiura* community at two muddy sites off northeastern England, the most recent of which (Buchanan and Moore, 1986a,b; Buchanan *et al.*, 1986) identified a biennial cycle of high and low densities during a stable period in the 1970s. An upward trend in numbers and biomass was discernible since 1980, with some evidence for a reversal in 1984/1985. This was tentatively ascribed to changes in the supply of organic matter to the seabed. However, in a complementary evaluation of temporal trends in benthic and pelagic (zooplankton) populations, Roff *et al.* (1988) concluded that the former were more likely to be limited by predation and competition than by food supply; further-

more, there was no evidence to link inter-annual cycles in zooplankton and benthos.

Austen *et al.* (1991) compared annual trends in the benthos off northeastern England with those at a comparable deep water site in the Skagerrak, over the period 1971-1988. The results were notable for a shift in community structure at both sites between 1980 and 1981. They further attempted to establish links with changes in plankton communities derived from Continuous Plankton Recorder (CPR) data. For the eastern North Sea, there was a less marked change in structure at about this time, but no such distinction could be drawn from data for the western North Sea. A comparison of annual changes in total abundance and, for the Skagerrak, total biomass with changes in phytoplankton colour and zooplankton abundance provided tentative evidence of a degree of co-variability. The authors speculated that a general increase in abundance and biomass of the benthos at both sites in the 1980s might be due to effects of eutrophication.

5.4 Hypoxia/Anoxia and Implications for Benthos and Fisheries

Hypoxia may result in the endofauna moving towards the sediment surface in order to get more oxygen. However, further development towards anoxic conditions may result in the death of both benthic organisms as well as bottom-dwelling fish species. Low fish numbers registered during such situations are often due to emigration from the area, and not only mortality. Unusually high catches of benthos combined with very low catches of fish, of which many caught were already dead, were observed in September 1981 about 40 miles off the Danish west coast covering an area of approximately 2 000 km (Dyer *et al.*, 1983). In August-September 1982, the bottom water in an area of more than 10,000 km² in the southeastern North Sea had oxygen concentrations of less than 4 mg/l, with the lowest values near 1 mg/l (Figure 5.4.1). Fish catches in the affected areas were low, and dead fish and macrobenthos were observed on the bottom (Dethlefsen and Westernhagen, 1983). In August 1983, oxygen deficiency with concentrations close to zero at certain stations were observed in the same area as in 1982 (Westernhagen *et al.*, 1986). These authors observed a change in the benthic communities from a fauna dominated by species tolerant to low oxygen in 1982 - 1983 to a dominance of more oxygen demanding species (e.g., short-lived polychaetes) in 1984 when more "normal" oxygen conditions were observed. In 1983, the total biomass was only 30% of that in the following years (Niermann *et al.*, 1990; Hickel *et al.*, 1989). However, the benthic community structure and biomass recovered within two years (Niermann *et al.*, 1990). After a series of "normal" years, oxygen deficiency problems reoccurred but on a more restricted

spatial scale, approximately 240 km², in 1989 (Niermann *et al.*, 1990).

Reduced levels of oxygen near the bottom have been observed particularly in the southern Kattegat over the course of many years, but the phenomenon has become acute during the last decade. The situation becomes especially marked during the autumn, typically in September - October. The southern Kattegat has a complicated hydrography with a marked halocline, with low salinity water originating from the Baltic Sea and Swedish rivers at the surface and North Sea water with high salinity at the bottom. A particularly well-documented change in oxygen concentrations was apparent in 1988-1991 (Figure 5.4.2 a-d); 1988 was characterized by very low oxygen concentrations distributed over a large area; 1989 showed a much reduced areal coverage due to wind/storm-induced mixing of the water column; the 1990 situation once again showed large areas covered by reduced oxygen concentrations extending also into the Øresund area; the 1991 situation resembled that in 1990.

The main fisheries in this region are those for cod, flatfish, and *Nephrops norvegicus*. After several years with oxygen levels reduced to < 1 ml/l, a pronounced change in these fisheries has occurred, together with that of the benthos, as well as aberrant behavioural patterns of various species. Swedish and Danish studies indicate that below 2 ml O₂/l, several fish species leave the area, *Nephrops* leave their burrows and remain on the surface of the seabed, and even infaunal species move onto the sediment surface. At about 1 ml O₂/l, many species of zoobenthos as well as *Nephrops* die (Baden *et al.*, 1990a, 1990b).

Although the distribution of various fish stocks are known to vary on a seasonal basis, the lack of food associated with hypoxia/anoxia in some areas obviously makes them unattractive to fish. As *Nephrops* left their burrows, they became more catchable to the trawls. Increased catchability and mortality/emigration has resulted in reduced stocks. *Nephrops* catches are essentially absent from a line south of the town of Falkenberg in Sweden and the island of Anholt off Denmark. There was a general trend of decreasing landings in the entire Kattegat from the early 1980s to 1991. In the southern Kattegat (Figures 5.4.3 b and c), both landings and effort exhibited decreasing trends during this period. Besides the general decline in the *Nephrops* stocks, oxygen concentrations especially during September 1988 (Figure 5.4.2 a) further caused a marked decrease in effort, landings and stocks (Figures 5.4.3 b-c). This has also been corroborated by scientific surveys and video-films obtained by Remotely Operated Vehicles (ROV). The commercial fishery has now moved to the northern Kattegat, where landings have also subsequently decreased but where effort has remained high (Figure

5.4.3 a). The lesson of this is that variability in oxygen levels may reduce stocks and landings in a given area in a manner that partly mimics that observed as a result of overfishing.

Hypoxia appears to be associated with increasing susceptibility to infectious diseases, such as lymphocystis and epidermal papillomas, in dab. This is evident in parts of the North Sea as well as the Kattegat (Dethlefsen, 1990; Møllergaard and Nielsen, 1990).

5.5 Organic Discharges, and Associated Contaminants

The main categories of point-source inputs are sewage, sewage sludge, and wastes from food processing/paper manufacture. These are discharged via pipelines or, in the case of some sewage sludge from the UK, from ships. Most major pipeline discharges of sewage are to rivers or estuaries, and hence the route to thesea is via river outflow, by which time the organic component may be substantially reduced. However, a number of large coastal settlements discharge directly to sea via short or long outfalls; much of this material is untreated, beyond initial screening of large debris. A summary of direct and river inputs to the North Sea of selected substances from all sources is given in NSTF (1990). This Section deals mainly with impacts arising from the particulate component of sewage and sludge. Possible consequences of nutrients and dissolved organic matter are considered under 'eutrophication'. The text is based primarily on experience with sewage sludge, but the impact of the organic component of other organic carbon-rich wastes would be very similar.

Sewage sludge disposal to the North Sea and English Channel is confined to UK waters, amounting to some 5 million wet tonnes per annum distributed among ten sites. A target date of 1998 for the cessation of this sea disposal option was announced prior to the Third Ministerial Conference on the North Sea. The sludge may be the end product of primary or secondary treatment of sewage, and its contaminant load varies regionally according to the balance of urban and industrial inputs to the sewerage system prior to treatment. Traces of a very wide array of contaminants will be present (Table 5.5) and, because of this, most studies of the biological consequences of discharges have treated the medium as an entity, rather than in terms of its numerous constituent parts.

Franklin (1983) showed that the short-term toxicity of dilutions of sewage sludge to a variety of organisms was relatively low. Impacts in the water column following discharge of sewage or sludge have been assessed by bioassays of field samples, e.g., using oyster embryos or caged mussels (Thain, 1990; Lack and Johnson, 1985; Roddie and Johnson, 1990). Such studies provide little

evidence of reduced water quality attributable to dispersing sludge, beyond the immediate area of discharge. There is some evidence to suggest that contaminants may enter the planktonic food chain via feeding (e.g., by copepods and mysids) on flocculent organic material held in suspension, although the quantitative significance of such a transfer route is uncertain (Chapman, 1986).

Most field studies of biological effects have concentrated on the impact of the particulate component following settling to the seabed. Benthic organisms may respond to the organic component ("enrichment") or to the purely physical consequences of accumulating particulates. "Toxic" effects, arising notably from physio-chemical changes in the sediment as a by-product of the enrichment process, have typically been observed in quiescent areas which promote a build-up of material at the sea floor.

The intensity and spatial extent of biological effects will clearly depend on the size of the discharge and the characteristics of the recipient area. Examples of well-defined gradients are provided by studies in fjords/sea lochs (e.g., Pearson, 1975; Rosenberg, 1976), where changes in the quantities of pulp mill discharges over time have provided a useful means of monitoring processes of degradation and recovery. Pearson (1987) has also reported on comparable gradients associated with sewage sludge disposal in the sheltered Clyde Sea area (Scotland).

The empirical model of Pearson and Rosenberg (1978), which describes structural changes in the benthos along an organic enrichment gradient, has been found to be widely applicable. It conforms with generic hypotheses concerning the role of "disturbance" in the maintenance of community diversity; these imply that the highest diversity occurs at intermediate levels of disturbance (Connell, 1978; Huston, 1979). "Pollution" in this context is just one manifestation of disturbance, as described by Warwick (1986) for marine benthos. Certain parallels may, therefore, be drawn between the responses of benthic organisms to many waste discharges, and to those occurring in the aftermath of deployment of certain types of fishing gear on the seabed. Thus, the enrichment process induces succession changes in the benthos, with short-lived opportunists (typically deposit-feeding worms) being favoured over the longer-lived, deeper-burrowing and more "sensitive" ("sensitivity" being defined by the ability of a species to adapt to, or tolerate, stressful events (see Gray *et al.*, 1990)) species.

Such marked "effects" gradients may be contrasted with studies at sewage sludge disposal sites in the more dispersive conditions typical of the UK east coast, where impacts on the benthos are generally harder to discern (e.g., Norton *et al.*, 1981; Talbot *et al.*, 1982; Pearson, 1986; Rees *et al.*, 1990).

Studies of the possible bioaccumulation of persistent contaminants along the food chain to commercial fish are not normally feasible on local scales because of the relatively small size of the waste field relative to the distribution and mobility of fish populations. Such work is more appropriate to regional scales; e.g., Franklin (1990) indicated that contaminant levels encountered in commercial species from UK waters are below the relevant Environmental Quality Standards, and do not pose a threat to human health.

More significant in some estuarine and coastal localities is the risk of faecal/microbial contamination of edible shellfish stocks; various conditions are imposed by local authorities concerning the treatment of such stocks prior to commercial sale.

While there is much anecdotal evidence that enrichment arising from sewage discharges leads to locally enhanced fish populations, there is little supporting scientific data for such effects in the North Sea. Again, this is partly a sampling problem arising from the relatively small spatial scales of any discharge effects. Off the Californian coast, Spies (1984) provided some evidence to suggest that structural changes in the benthic community in response to offshore discharges of sewage were accompanied by structural changes in the fish community, reflecting the ability of different species to exploit the modified food resource.

Player (1971) and Pounder (1974, 1976) have provided some evidence that sewage and certain trade waste effluents may provide an important food source for birds, either directly in the form of discharged materials, or indirectly through localized increases in benthic biomass brought about by organic enrichment.

Conclusions:

- i) Well-defined effects are invariably localized in extent and are, in principle, reversible;
- ii) Empirical models adequately predict benthic responses to organic enrichment, at least in quiescent areas; such responses may in many respects be analogous to those occurring as a consequence of physical disturbance at the seabed, e.g., arising from commercial fishing gear;
- iii) An important inference to be drawn from current hypotheses concerning influences on community structure is that the highest diversity occurs at intermediate levels of disturbance. Maintenance of high diversity as a management goal is therefore not incompatible with controlled waste disposal or other anthropogenic activity (including fishing). However, there may be other goals, such as the

preservation of sensitive species and habitats, which may not be fulfilled;

- iv) A model which adequately describes the responses of benthos (and fish) to discharges in dispersive areas is currently lacking. Scientific studies in such areas are often hampered by the occurrence of high natural variability. However, the benthic communities of such "high energy" systems may be more robust to physical disturbances or to the discharge of contaminants;
- v) It seems plausible that some organic discharges may result in locally enhanced fish populations, but the evidence is largely anecdotal;
- vi) The spatial extent of identifiable effects on the benthos arising from organic discharges to the North Sea, relative to those which may be attributable to the use of certain types of heavy fishing gear, is probably relatively small. However, outfall responses may still have local significance depending on the sensitivity of the recipient environment.

5.6 Shipping, Offshore Structures, Hazards and Port Activities

Shipping

Shipping activities are particularly prevalent in shelf seas and coastal areas. In shallow regions, vessels with deep drafts (e.g., oil tankers) may transfigure the bottom. Ships provide noise, both from engines and from propeller cavitation. It is well known that certain fish species, e.g., herring, take evasive action well before a vessel is overhead. In heavily trafficked areas, propellers "churn" significant volumes of water, sometimes injuring plankton, fish eggs and larvae. Everyday seepage of oil and fuel also occurs, and gases and chemicals are emitted from funnels.

The North Sea is one of the most actively trafficked shipping areas in the world; on average 300 or more ships pass through the Channel per day. Shipping cannot be considered to be a notable source of mortality for biota, with the possible exception of planktonic organisms. The extent to which shipping has an effect on the distribution and behaviour of biota is essentially not known.

Offshore structures

The number of offshore platforms for oil and gas exploration is increasing relatively slowly in the North Sea. The oldest at present date from about the early 1970s and are still in operation. It is expected that the

number of fixed surface installations will peak in the year 2000 but that the subsea developments will continue to increase as will the extent of the undersea pipelines. In most countries, national legislation insists that, after removal of the platform, the bottom be cleared of debris. As many of the offshore fields have passed their peak and now exhibit declining production, smaller fields that previously have been untouched will be brought into operation. As these are frequently too small for a whole platform, they tend to be exploited via "subsea completions". Thus, an increase in subsea platforms and associated well-heads may be expected in the North Sea in the future. Pile constructions are often removed by cutting the legs of the platform 6 m below bottom level. Abandoned wells are cemented below the bottom level and cut, while temporarily abandoned wells are cemented as above, but up to 3 m of pipe protrudes above the seabed. Currently there are about 160 platforms, 250 well-heads and over 5,000 km of pipeline in the North Sea.

Exploitation platforms have a safety zone of about 500 m, while exploration platforms have no safety zone, as their presence at a given locality is usually of short duration. Subsea platforms also have safety zones.

Initially, pipelines either were or had to be buried about 2 m below the seabed. At present, only pipelines with a diameter of < 17.5 cm have to be buried 0.2 m into the bottom. Larger diameter pipelines are left on the bottom, eventually sinking into the bottom by "self-burial".

Pipelines have no safety zones, and often act as focal points for attracting fish. Trawling parallel to the pipelines frequently occurs. The pipeline system of the North Sea is likely to be expanded by 30-50% in the near future. Large lines under construction/completion are Ekofisk-Emden, North Gas-transport (NGT), NOGAT (Dutch Shelf), and Zeepipe (Norway - Zeebrugge/Belgium).

Explosives

Large quantities of mines and munitions/bombs were either intentionally positioned or jettisoned in the North Sea during and immediately after the two World Wars. The dumping grounds for munitions were originally limited in area. Fishing activity has also contributed to the dispersal by re-dumping caught explosives.

Wrecks and artificial reefs

There are about 20,000 known wrecks in the North Sea (Leewis and Waardenburg, 1991). Although the wrecks may be a hindrance to fishing (e.g., snagging bottom trawls), they provide beneficial shelter for many marine species, especially fish, thus enriching the marine biota. Trend studies on the benthic community on wrecks show

an increase in abundance and biomass of "hard-substrate" species (Leewis and Waardenburg, 1991). Both non-commercial as well as commercial fish species may find "safe-havens" in wrecks. Recently, several artificial (i.e., purposely made) reefs have been constructed to provide conservation and enhancement areas (Collins *et al.*, 1990).

In conclusion, all these offshore structures function as protected areas, especially with regard to fishing activities. They demonstrate elevations in biomass and species diversity.

5.7 Mineral/Substrate Extraction and Disposal

Effects of extraction of marine sediments

The effects of extraction activities (extraction of sediments/substrates, but also of the calcareous seaweed, "maerl") on living resources include:

1) Physical impacts on the seabed and water column

- changes in substrate character;
- suspension and dispersion of fines in the water column;
- balance between deposition and erosion both in the area in question and in surrounding areas.

2) Chemical impacts on the seabed and water column

Dredging in sheltered areas may cause concern over the potential release of toxic materials and heavy metals "locked up" in deep, anoxic sediments prior to being exposed to oxygenated conditions by the action of the mechanical dredge. Resuspension of bottom material rich in organic content can place an elevated oxygen demand on the environment. In some cases, increased recirculation and liberation of nutrients from sediments occurs, adding to eutrophication risks.

3) Biological impacts

- the loss of organisms in the extracted sediments;
- the effect of the deposition of fines lost from the dredger both with regard to temporary water column effects of suspended fines on primary production (e.g., elevated turbidity), and long-term change in the character of the benthos.

Changes in substrate character may affect benthic community structure and succession, and thus the type of food available for finfish and shellfish. Particular substrate types are frequently type-specific for the

attachment of demersally attached eggs, as well as the settlement and recruitment of larval stages. Sessile organisms may become partly covered or buried by fines. High ratios of particulate inorganic matter to particulate organic matter in suspended material may result in clogging in filter feeding animals, delayed gut passage time, reduced assimilation rates and, accordingly, result in reduced growth (Anon., 1989a).

Capital and maintenance dredging activity

Dredged materials from the maintenance of navigable channels, channel deepening, port/harbour construction and so on are disposed of under national licence at several coastal (and estuarine) locations in the North Sea. Table 6.2 provides a rough estimate of the area of seabed being used for this purpose. The disposal activity is monitored at an international level through the Oslo and London Dumping Conventions, and information on quantities and chemical quality is published in Oslo Commission reports. Some 70 million tonnes were deposited at sea (i.e., within 'Convention waters') in 1990, with similar quantities to 'internal waters', especially estuaries.

Guidelines concerning the chemical quality of the material suitable for marine disposal are designed to minimize adverse consequences for the biota. Aside from impacts at the site of dredging activity, the main effects to be anticipated are physical in nature, e.g., through blanketing of bottom deposits and fauna in the immediate aftermath of disposal, and/or associated with organic enrichment, e.g., where soft sediments are dredged from depositional areas within estuaries draining highly urbanised catchments.

In many maintenance dredging operations, there is an element of 're-cycling' involved: the disposal of sediments of similar nature to those of the receiving environment may result in minimal changes to the physical structure of the sediments, or to the benthos, especially in dispersive environments.

Some types of large-particle solid wastes, e.g., from capital dredging projects or mining, may materially alter the seabed within licensed areas, and may render it unsuitable for some types of fishing activity. However, it may be expected that such material would generally be contained within the boundaries of the licensed site. Eagle *et al.* (1979) estimated that some 2% (or 40 km²) of the inshore fishing zone between Blyth and Teesbay off the English northeast coast was in some way affected by colliery waste, fly ash and dredged material disposal. These effects ranged from practically azoic conditions in areas avoided by trawlers, to marginal depletion of benthos and little interference with crab and lobster potting in peripheral areas.

Existing and new approaches to the management, monitoring, and modelling of dredgings disposal in European waters have recently been reviewed in Alzieu and Gallene (1990). It is to be expected that such matters will receive increasing attention in the coming years.

Estimation of scale of effects and consequences of dredging for sand and gravel

The scale of effect of sediment extraction projects and corresponding consequences for marine life and fisheries are dependent on the environmental characteristics of the area (e.g., wave and current regime, geology, turbidity, the nature and extent of the extraction operation, and the time to recover or for the readjustment of the benthos).

In principle, an estimation of the scale of effects and consequences is possible for any given site-specific sediment extraction project (Anon., 1990c). The overall immediate loss of biota in the extracted sediments can be estimated from the areal extent of extraction and an analysis of the biota in benthic samples, and fish/shellfish surveys considered representative for the area. However, our understanding of the contribution of benthic organisms to the diets of other marine organisms, such as benthic-feeding fish species, is frequently not sufficient to provide an accurate assessment of the effects of this loss through each component of the food web. In some areas, a proportion of the benthic biota lost may have a direct, economic value to a commercial shellfish fishery.

The contribution of suspended fines to increased turbidity and possibly reduced primary production in the water column may be approximated by a simple dispersion/settling model using current data applied to the discharged fines. Primary production can be significantly reduced in the zone in which higher than background turbidity levels are predicted to occur. While again one can extrapolate the effects of a loss of primary production through a simple food web model, it can be considered that the expected area of aggregate extraction relative to the total North Sea area of about 575×10^3 sq km is likely to be insignificant. Nevertheless, potentially significant, small-scale effects on sensitive biota (especially filter feeders) may occur.

5.8 Offshore Exploration, Exploitation, and Impacts

In this section, the Skagerrak and English Channel are not included in the quantitative results. There are no hydrocarbon activities in the Skagerrak and no data on activities in the English Channel were available to the Study Group.

Geophysical prospecting

The oil and gas industries initially used explosives for seismic surveys. The use of explosives was abandoned as it caused death to marine animals, especially fish. The air gun was brought into use for two-dimensional (2-D) seismic surveys from 1963. Most of the potential resources of the North Sea have been surveyed by 2-D seismic surveys. However, since 1983 three-dimensional (3-D) seismic surveying has also been used, and all areas with the likelihood of small reserves of oil and gas and all areas not yet covered using 2-D surveys, will be surveyed by 3-D. The greater accuracy and precision of the surveys resulted in an increase in successful exploratory drilling (from 25% to 60% successful). It is estimated that about 10 years will be required to cover all potential gas and oil reserves with 3-D surveys.

The 3-D seismic arrays can only be used during calm weather conditions, mainly from April to September. It can thus be expected that conflict will arise near the coast with breeding colonies of birds and further out with the birth/nursery areas of marine mammals (e.g., seals and porpoises). In the case of marine mammals, the "calving" areas are frequently poorly delineated.

The effects of the air gun depend on the type of bottom (e.g., sand, rock, or mud), but have been reported as a cause of death for zooplankton and larval/young fish in relatively close proximity (generally within 2-3 m) of the emission source (Dalen and Knudsen, 1987).

Oil and gas

This section contains no information on activity in the English Channel, for which data were unavailable to the group, or for the German North Sea Sector, Kattegat or Skagerrak which have no emissions.

Emissions/discharges

Significant oil and gas production started in the North Sea in the late 1960s. Total well drillings to date number 3000 to 4000 (Figure 5.8.1).

There are three main sources of contaminants arising from the offshore mining industry:

- drill cuttings;
- production water; and
- spills.

A major component of the contaminants from these sources is oil. Oil-based mud (OBM) drill cuttings contribute the greatest amount of oil (Figure 5.8.2) (Anon., 1992f). From 1988 there has been a decline in the amount of oil via cuttings because of current regula-

tions, according to PARCOM (Paris Commission) decisions. The relative contribution of production water to the total amount of discharged oil has increased in recent years, both because of the decrease in the amount of oil released from cuttings and because of the increase in the amount of production water. Accidental spills result in only minor amounts of oil entering the North Sea (Figure 5.8.2).

A large amount of cuttings are produced every year. For example, in 1985, the estimated amount of (OBM drill) cuttings discharged was 260,000 tonnes. The other components of drilling muds (both oil-based and water-based muds) are numerous, including toxic heavy metals, barite, bentonite, inorganic salts, surfactants and detergents, corrosion inhibitors, lubricants and biocides (UNEP, 1985).

Production water is a very complex chemical mixture and will vary in composition from field to field and within one field depending upon its duration and operation. Its toxicity is likely to be due to the presence of biocides (Law and Hudson, 1986). The amount of production water discharged annually into the North Sea has increased during recent years (Anon., 1992f).

The pipeline is still the safest transport system available for oil and gas. Only about 1% of all the oil spilled world-wide into the sea originates from pipeline leakages and ruptures.

Contamination

Contaminants enter both the water phase and the sediments (Figure 5.8.3). The amount of oil in sea water, arising from discharges of OBM drill cuttings and production water, as well as flaring operations, is generally low (Anon., 1989a). Sheens of oil observed on the sea surface may come from flaring operations and offshore mining discharges or from shipping. During overflights, an average of 1-2 oil sheens per flight hour have been observed over the last few years (Anon., 1989b; Anon., 1991h).

The accumulation of contaminants in the sediments mainly arises from discharges of OBM drill cuttings, but, depending on the current regime and depth, it may also arise from production water discharges in (turbid) shallow waters (Reiersen *et al.*, 1989; Gray *et al.*, 1990; Figure 5.8.3).

Background concentrations of oil in North Sea sediments, at reference sites away from the influence of drilling sites, are in the range of 0.2 - 15 mg oil/kg dry sediment, depending of the sediment type (sandy < silt) and analytical methods used (Law and Fileman, 1985; Massie *et al.*, 1985). Elevated oil concentrations in the sediments, as high as 100 g oil/kg dry weight, have been

found near discharge sites (Davies *et al.*, 1984; De Jong *et al.*, 1991; Van Het Groenewoud, 1991). Depending on the level of input and the hydrography, elevated oil concentrations have been found from 750 m to 12 km from the drill site (COWICONSULT, 1989; Reiersen *et al.*, 1989; Kingston, 1992). Once drilling has ended long-term (>6 years) oil contamination around drilling sites is observed. This is due to the rather low biodegradation rates of oil and the redistribution of oil-contaminated sediments (Daan *et al.*, 1990b; De Jong *et al.*, 1991). Taking into account the total amount of oil discharged through cuttings up to 1990, it has been estimated that as much as 8000 km² of seabed in the North Sea has sediments with elevated levels of oil (QSR Expert Group, 1992). Elevated levels of heavy metals (Cd, Cr, Cu, Hg, Pb, Zn), associated with discharges from drill cuttings, have been found around several drilling platforms (Reiersen *et al.*, 1989; COWICONSULT, 1989).

Impacts

Toxicants originating from offshore mining discharges find their way from the sediment and water phase into various organisms (Figure 5.8.3). Mussels have been found to be contaminated with oil up to 6 km from an oil platform (Anon., 1988a; Van Het Groenewoud, 1991; De Jong *et al.*, 1991).

For most macrobenthos, the NOEC (No Observed Effect Concentration) is about 10 mg oil/kg dry sediment. Above this level, a variety of effects become visible, such as a reduction in sensitive species, an increase in some opportunistic species, and an overall reduction in macrobenthos (Daan *et al.*, 1990b; Gray *et al.*, 1990; Kingston, 1992). In about 20% of the 8000 km² of the seabed which show elevated levels of oil, the oil concentration may be high enough to give rise to an overall reduction in macrobenthos. Affected areas are not evenly distributed throughout the North Sea, but rather concentrated in certain subareas (Figure 5.8.1). Sessile benthic organisms have been shown to become smothered by cuttings (Davies *et al.*, 1984).

Experiments with caged mussels placed around drilling sites showed a reduced scope for growth, but no effects on the survival of mussels (Van Het Groenewoud, 1991). For infaunal species, such as *Amphiura filiformis*, sediments from up to 500 m from rigs have been found to be toxic (Figure 5.8.4, De Jong and Zevenboom, 1991).

Adverse effects of oil at the sea surface and in the sea water depend on the co-occurrence of the location of oil spills and seabirds. Seabirds are at risk from floating oil due to damage to the waterproofing of their feathers (Figure 5.8.3). Sea mammals, particularly those that spend long periods at the water surface, are also at risk (Dicks *et al.*, 1987).

6 COMPARING THE DIRECT EFFECTS OF FISHING WITH THE EFFECTS OF OTHER ANTHROPOGENIC INFLUENCES AND NATURAL PROCESSES

6.1 Introduction

In this section, we attempt to compare the direct effects of fishing with other anthropogenic influences and natural processes. The major anthropogenic influences considered are described in Section 5 and appropriate details on natural processes are elaborated upon here.

6.2 Impacts on the Physical Structure of the Sediment and Abiotic Processes

Other human activities which might be expected to influence the structure of the sediment in a way similar to fisheries include sand and gravel extraction, pipeline and cable laying, dredging, and oil platform construction. In contrast to fishing, however, these activities are much more localised. Table 6.2, modified from Anon. 1992g, summarizes the areas influenced by the different human activities, and shows that the area influenced by fisheries is more than one order of magnitude larger than areas influenced by any of the other activities.

The impact of fishing activities should also be compared to disturbance caused by natural processes and events. In some areas, strong currents and storms may have effects that are at least equivalent to those caused by fishing. In shallow areas (<25 m), storms will change sediment characteristics in a way similar to that described for fisheries (see Section 4.3). There are only few data available to compare these effects. A theoretical approach comparing the energy input of beam trawling *versus* wave action during a severe storm showed that the intensity of energy transfer per unit of time during the passage of the trawl is a factor of 100 or more greater than during a storm, actually indicating the incomparability of these events. However, on an annual basis the energy transfer of beam trawls is far less than that of waves (de Groot and Vogel, working doc. 3). In deeper areas, the influence of wind events decreases, thus increasing the relative influence of fisheries. However, Cacchione *et al.* (1987) conclude that also in deeper water (85 m) winter storms may be a major factor controlling the distribution of surface sediments.

The effect of trawling on nutrient exchange rates has been measured for otter trawls in the Kiel Bight (Krost, 1990). Considering that the phosphate gradients in the sediment in the Kiel Bight are similar to or slightly steeper than those in sub-area 4 in the North Sea, we may assume that the effects in this part of the North Sea are the same or less. We estimated that in NSTF sub-area 4 (otter trawled surface area 749 km²), the trawl doors enhance the phosphate release from the sediment

by maximally 70-380 tonnes per year. Beam trawlers (trawled surface area 132,000 km²) enhance the phosphate release from sediments by maximally 10,000 - 70,000 tonnes per year. These additional releases are pulses largely compensated by lower fluxes after the trawl has passed, and therefore the net result is almost zero (see Section 4.2). In comparison, Lindeboom (1975) found in the same area a phosphate release from the sediment of over 4,000 tonnes as a result of one storm event. Both processes are a recycling of phosphate within the system, while the river input amounts to 26,000 tons per year (PARCOM, 1992).

Comparing the effects of fisheries with the effects of natural events, it may be concluded that the relative impact of fisheries on abiotic processes is small.

6.3 Mortality

6.3.1 Benthos

To compare possible effects of fisheries, the level of mortality from the various sources must be estimated. For many sources of mortality, the spatial and temporal scales over which they operate are not comparable. This means that efforts to rank the importance of an intense local effect compared to a more subtle effect which occurs over a wide area, requires value judgements which we are reluctant to make. Notwithstanding these difficulties, some coarse but robust conclusions can be drawn for some sources of mortality.

Table 6.2 provides estimates of the total area of the North Sea affected by various activities and indicates that the area swept by those parts of fishing gears which have a major impact on the seabed is at least one order of magnitude greater than that covered by any other activity.

In many areas, activities other than fishing occur, but the areas affected are much more localised. For example, in NSTF sub-region area 4 changes attributable to the offshore oil and gas industry occur over less than 8% of the total area, of which only 2% will show serious benthic effects (see Section 5). In contrast, beam trawling is widespread, probably covering almost all fishable grounds in the subregion. Mortalities in the immediate vicinity of point source discharges, such as oil platforms or dredged material disposal grounds, are high and effects on the community are much more profound than any effect that has been documented for fishing. One possible exception to this is the case of reef-building taxa which have been destroyed by fishing gears. On the other hand, the spatial scale over which many fishing activities operate is much greater than the scale of other activities.

Assessment of the significance of pollution presents some problems: evidence from field surveys points to the

localisation of gross effects around discharge points, at least in coastal and offshore areas (e.g., Rees and Eleftheriou, 1989). Recent wide-scale surveys of the North Sea (e.g., Eleftheriou and Basford, 1989; Kunitzer, 1990) and historical comparisons (e.g., Pearson *et al.*, 1985; Rosenberg *et al.*, 1987) provide no indication that pollution-related mortality has resulted in whole-scale changes in species occurrences or community structure. (Note that the effects of eutrophication are considered separately.)

Areas which experience temporary anoxia can be large and benthic mortalities can be high (see Section 5.4). The mortalities recorded for benthic taxa may be even greater than those caused by fishing in these areas. As with point-source pollution, however, comparing processes operating over different spatial and temporal scales is problematic.

Changes in the spatial and temporal distribution of benthic communities also occur as a result of natural processes. Of these, the natural physical disturbance of sediments by currents, waves and storms is, perhaps, the most obvious. Natural sediment disturbance and the benthic mortalities that may be associated with it will clearly correlate with depth and sediment type, and shallow (<20 m) sandy sediments are likely to be most affected. Data on net bed-load movement are clearly available, but for the question of fishing disturbance, it is the intermittent bed-load events resulting from storms that are likely to be most important (Eagle, 1975; Rees *et al.*, 1977; Yeo and Risk, 1979; Dobbs and Vosarik, 1983).

Daan (1991) offers an approach to assess the relative importance of fishery-induced mortality, which uses average production/biomass (P/B) ratios of benthic species as a measure of the instantaneous natural rate of total mortality. These average P/B ratios were compared to the estimates of mortality caused by the passage of fishing gears, assuming that all individuals in the path of the gear are killed. He concluded that trawl-induced mortality was minor relative to other sources. However, there are limitations regarding the methods described above, which are readily acknowledged by the author. First, the P/B ratios used are estimates for the community as a whole. Thus, the analysis considers the gross attributes of benthic communities and does not address the question of effects on particular taxa which may be sensitive to fishing. In addition, the analysis assumes that the community comprises a homogeneous assemblage of individuals such that rates of growth or mortality do not change with size or age. Thus, the effects of fishing on the size structure of populations and the potential for consequent changes in the P/B ratio are not explored. Moreover, P/B ratios will be dominated by small-bodied taxa and are relatively insensitive to changes in the abundance of large individuals. For a more complete account of the utility of P/B ratios, see Section 9.2.

We have made efforts to quantitatively assess the effects of fishing on the North Sea ecosystem. It should be recognised, however, that the level of analysis we have been able to apply is coarse and is severely limited by available data. Table 6.3.1 summarizes the outcome of a literature review concerning life-cycle traits and sources of mortality for local populations of nine benthic species. Each species was scored (on a scale from 1 to 4) according to perceived risks of local extinction of populations (Rees and Dare, working doc. 7). For example, the consequences of toxic algal blooms or anoxia can be locally severe for *Echinocardium cordatum*. It should be clearly understood that its vulnerability will depend on the magnitude and geographical location of the event. However, for the North Sea as a whole there is no evidence that this benthic species is at risk (see Section 7.2.1).

Echinocardium and *Arctica* are two conspicuous species for which damage by trawls has been noted (BEON, 1991). *Echinocardium* lives for 5-10 years in the southern North Sea (Beukema, 1985), while *Arctica* have been reported to live for 100 years or more off the north-eastern USA (Thompson *et al.*, 1980; Ropes, 1984) and approximately 20 years in the western Baltic Sea (Brey *et al.*, 1990). Recruitment success in both is likely to be very variable between years. Trawl-induced mortality may be locally significant for a number of other species, especially in intensively fished areas.

6.3.2 Fish

The known causes of mortality in wild populations of fish include fishing, predation, disease, thermal shock, anoxia, and poisoning. Mortality in most populations appears to be a gradual process of attrition rather than a series of episodic events. However, mass mortalities do occur at times following local pollution incidents, disease outbreaks or occasionally freak natural events such as incursions of water of abnormal temperature. These are usually local and transient in effect.

In some cases habitat degradation may affect fish, for example, by destroying the spawning grounds of demersal spawners. It has been suggested, for example, that the extinction of the population of spring-spawning herring in the Firth of Forth may have been caused by dumping of colliery waste on the spawning beds (Johnston and Davies, 1975). There is no evidence of habitat degradation on the spawning grounds of herring elsewhere in the North Sea at present, although this is one of the considerations in the granting of concessions to explore for hydrocarbons. Outside the North Sea area as such, a spawning ground for herring in the Firth of Clyde is closed to all active forms of fishing during the spawning season to protect the beds of spawn. Other cases of habitat degradation include the direct impacts of hypoxia and anoxia on fish communities, e.g., in the

southeastern North Sea and the Kattegat (see Section 5.4).

In the major fish populations in the North Sea, mortality has been modelled by a combination of fishing mortality, predation mortality and other mortality. The model ("Multispecies Virtual Population Analysis", MSVPA (Anon., 1991e; Sparre, 1991)) includes eleven of the most important commercially exploited fish species in the North Sea for which time series of the age composition of the catches are available.

To estimate mortality rates attributable to different causes, the model partitions mortality into:

- i) fishing mortality,
- ii) predation mortality caused by cod, haddock, whiting, saithe and mackerel (the 'MSVPA' predators), and
- iii) other mortality, combining estimates of predation by birds and mammals with estimates of other sources of mortality.

To provide a frame of comparison, mean values estimated by the model of each of the above mortality factors over the period 1985-1989 have been converted into proportions of the annual production of that stock (Table 6.3.2). From these estimates it can be seen that the primary source of mortality differs widely between stocks. In the relatively long-lived stocks cod, saithe, plaice and sole, fishing mortality is by far the most important factor, whereas in stocks of small, relatively short-lived species, other causes of mortality predominate. Predation mortality by the 'MSVPA' predators in the model is quite important in the case of Norway pout, sprat and sandeel, but quite a high proportion of the mortality in some species is caused by other factors.

The annual production of fish in the North Sea by the eleven species included in the model (Table 6.3.2) has been estimated at 7.5 million t. Of this, fishing takes 2.5 - 3 million t, the 'MSVPA' predators take 2 million t and other sources of mortality account for about 3 million t. Of the "residual" mortality, it has been estimated that seabirds take in the region of 0.35 million t (Bailey, 1986), although a similar estimate made by Tasker and Hislop (in Anon., 1991e) included discards and offal taken by scavenging seabirds.

Taking into account the fish species not included in the 'MSVPA', the total annual fish production in the North Sea has been estimated at between 10 and 13 million t (Bailey, 1986; Sparholt, 1990). This figure is rather higher than some earlier estimates based on estimates of primary production and assumed transfer coefficients (e.g., Steele, 1974).

While the above estimates are appropriate for the North Sea as a whole, seabirds clearly take a much higher proportion of the production in smaller areas. In the Shetland area, for example, Bailey *et al.* (1991) estimated that the seabird consumption/production ratio of sandeels (*Ammodytes marinus*) was in the range 0.28-0.33.

To summarize, it may be concluded that the main causes of adult mortality of fish differ between species and that, whereas fishing is the main cause of mortality in the larger species used for human consumption, predation by natural predators is the most important factor in the small or short-lived species and for juvenile fish in general.

6.3.3 Seabirds

Comparison of the various impacts of fishing, other anthropogenic activities, and natural processes on seabird populations in the North Sea (and elsewhere) is hampered by several aspects of seabird biology.

Most seabirds have a long period of immaturity, followed by a relatively long breeding life compared with many marine organisms. In general fecundity is low. Sources of mortality can thus have a different effect on a population of seabirds, depending on when they operate. Mortality early in life will have relatively less effect on populations than early in adulthood. It is therefore difficult to compare the importance of direct fishery-related causes of death, such as entanglement, with other causes of death.

Chemical pollution has reduced the population size of seabirds at least once. Effluents from a pesticide factory near Rotterdam from 1964 to 1968 led to over 70% mortality in terns, herring gulls, eiders and other birds (Duinker and Koeman, 1978; Rooth, 1980; Smit, 1981). This resulted in a steep decline in the breeding populations of these species in the Wadden Sea; some of these populations have yet to recover fully. Becker *et al.* (1991) showed that common tern eggs which failed to hatch had significantly higher levels of DDT and PCBs than eggs collected at random in the same colony.

Several schemes record the number of birds that wash up dead onto beaches, with some schemes recording apparent causes of death. Averbek *et al.* (1992) recorded that over an eight-year period (1983 to 1990) 9524 birds were found dead on a sample of 25 km of beach on the German North Sea coast surveyed at least twice a month. Of these, 28% were victims of oil pollution; in the winter of 1991, this proportion fell to 15% (C. Averbek, pers. comm.). The remainder died of unknown causes. However, in another study on 65 km of German beaches surveyed regularly between 1983 and 1988, 78 birds were found dead entangled in plastics or litter, of these 17 were in litter/netting attributable to the

fishing industry (Hartwig *et al.*, 1992). Further north on the North Sea coast of Denmark, over 80% of birds found dead are oiled, with very few killed by nets (Danielsen *et al.*, 1990). Birds which drown in nets are often difficult to separate from birds which have died naturally, an exception to this is when wings are removed when clearing nets.

When bird rings are returned from birds found dead, the cause of death is often reported. Of 460 first-year guillemots ringed in Shetland between 1959 and 1990, Heubeck *et al.* (1990) found that 46% were killed in fishing nets. A further 29% were 'found dead' - a category likely to include some birds drowned in nets. Equivalent figures for 304 older birds were 31% and 40%. Most of the fishing net casualties were from Scandinavian coasts. Mead and O'Connor (1980) analysed all recoveries from the coasts of Britain and Ireland from 1967 to 1978 (Table 6.3.3). Seabirds that feed mostly from the air were less affected by fishing nets and oil than the diving species. Those species that feed in inshore areas tended to suffer more from net entanglement than the offshore species.

6.3.4 Marine mammals

Recent mass mortalities of striped dolphins (Aguilar and Raga, 1990), Baikal seals (Grachev *et al.*, 1989), and harbour seals (Dietz *et al.*, 1989) have highlighted the potential importance of infection from pathogens. Anthropogenic activities, in addition to fishing, which could affect marine mammal populations include hunting, contamination by pollutants and disturbance, both at sea and at haul out sites. Hunting for marine mammals no longer occurs in the North Sea. However, a small number of harbour porpoises are occasionally killed in the Faroe Islands, where there is still a hunt for pilot whales.

6.3.4.1 Seals

Natural processes have had the greatest impact on seal populations in recent years.

The outbreak of phocine distemper virus (PDV) in 1988 caused the deaths of some 18,000 common seals in the North Sea (Dietz *et al.*, 1989). The impact of the epizootic varied geographically, with populations on the eastern coasts of the North Sea and in the southwest of the region being among the hardest hit (Dietz *et al.*, 1989). Phocine distemper virus antibodies are still recorded in common seals, harp seals and grey seals, indicating the continuing presence of the virus and highlighting the potential for the future reinfection of common seals from other species (Markussen and Have, 1992).

High contaminant levels, mainly polychlorinated biphenyls (PCBs) and other organochlorines, are found in seals in the Wadden Sea and Skagerrak areas. These contaminants may cause reduced fertility (Helle *et al.*, 1976a) as well as being implicated in causing diseases (Helle *et al.*, 1976b), such as paradontitis and alveolar exostosis (Mortensen *et al.*, 1991). High PCB levels have been recorded in common seals in the Wadden Sea and experiments have suggested that these may have been high enough to affect reproductive success (Reijnders, 1986, 1988). Despite this, the number of common seals in the Wadden Sea has increased in the three years following the epizootic (Anon., 1992d).

We lack reliable data on numbers of deaths due to contaminants and fisheries. Therefore, we cannot place these mortality sources in a context relative to each other, or to the overall dynamics of North Sea seal populations.

6.3.4.2 Cetaceans

Polychlorinated biphenyl contamination gradients have been demonstrated in North Sea harbour porpoises, with contamination levels highest in the Kattegat area, intermediate in the northern North Sea and lowest further north. No indication of any abnormalities in ovary and uterus function or structure were observed (Bjørge *et al.*, 1991).

The presence of bottlenose dolphins in the Moray Firth has generated considerable interest and boats now operate taking people out to see them. The effect of this disturbance impact has yet to be determined.

Until reliable estimates of the additional mortality due to by-catches are available, it is not, in general, possible to evaluate their importance relative to other anthropogenic factors (Anon., 1992d).

6.4 Discards and Offal as Food

6.4.1 Seabirds

Seabirds gather and feed around fishing boats in all parts of the world. Observations in the North Sea have shown that at various times of the year, fulmars, large gulls, gannets, great skuas and kittiwakes have distributions closely associated with fishing vessels (Tasker *et al.*, 1985a, 1985b, 1987). Direct observations from fishing vessels in the northern North Sea showed that each seabird species specialises in the type and size of fish taken and that there are feeding hierarchies between species and between age classes of individual species. Between 60% and 70% of discarded gadids and some pelagic fish near Shetland were consumed, while flatfish and red gurnard were largely ignored (Hudson and Furness, 1988). Up to 75% of all fish discarded around

Shetland may be taken by seabirds (Furness *et al.*, 1988). In the southeastern North Sea (NSTF area 5 south), Hüppop and Garthe (1992) found by experimental discarding that 88% of all roundfish discarded and 38% of all flatfish were consumed by seabirds. Not all of these observations were made near seabird colonies, but we have no way of knowing how representative they are of the North Sea as a whole.

Furness *et al.* (1988) combined information on the calorific value of discarded fish, and estimates of the energy requirements of birds, in order to estimate the numbers of scavenging birds that use discards that could be sustained throughout the year in the North Sea (Table 6.4.1). These figures could be updated to fit NSTF areas once further information on discard rates becomes available. Based on information presented in Section 4.5, the figures in Table 6.4.1 are probably minima.

The equivalent of about 12.5% of the mass of roundfish (Section 4.5) and about 6.5% of flatfish (Furness *et al.*, 1988) processed for market is discarded as offal at sea. These figures, ICES fish catch statistics summarised in Section 3.3.4, information on the calorific value of offal, and estimates of the energy requirements of fulmars (the main consumer of offal in the northern North Sea) (Furness *et al.*, 1988) were combined to estimate the numbers of fulmars that could be sustained throughout the year in the North Sea (Table 6.4.2). The estimates of numbers of fulmars supported in the North Sea can be compared with a total of 1,600,000 fulmars present on average offshore in the North Sea (Tasker *et al.*, 1989).

Although the number of birds potentially supported is less than the total populations of these scavenging species in the North Sea as a whole, this additional food for seabirds must have a substantial effect on totals present in an area. In addition, the location of fisheries influences the distribution of seabirds at sea, and is likely to affect the location of some seabird colonies. These alterations may influence the wider ecology of an area, and may bring seabirds into areas where they are at greater or lesser risk from other anthropogenic factors, such as oil pollution.

6.4.2 Other animals

Discarded fish and offal are only accessible to seabirds at or near the water's surface. Items not consumed immediately may quickly sink beyond their reach. The fate of this material is unknown, but it must be a potential source of food for pelagic and demersal fish, and benthic invertebrate scavengers. Daan (1973), for example, has considered larger specimens of plaice and sole in the stomachs of cod to be discards from fishing vessels rather than wild-caught prey. On the other hand, it seems likely that the size range of fish that are damaged by trawls may be rather larger than those normally

eaten by many predators. In this case, the fish involved may not be consumed.

Spatial and temporal variations in the abundance of discarded material available to non-seabird scavengers will be primarily determined by fishing practices; the type of fisheries involved, and where they occur will have a profound influence. For example, near Shetland, an area where seabird densities are high, only 25% of discarded material was allowed to sink beyond the seabirds' reach (Section 6.4.1.). However, in regions with lower seabird densities, a larger proportion of discarded material might be expected to drop through the water column. Discarded flatfish and gurnards are largely ignored by seabirds feeding near fishing boats (Section 6.4.1.). Material discarded from boats at flatfish fisheries will, therefore, mainly be exploited by other animals.

On occasion, the quantity of fish discarded, for instance from spilled catches and dumping from klondykers, may swamp the seabirds' ability to respond and large quantities may pile up on the seabed. This may have a variety of short- and long-term effects on the local habitat. In the rather unusual circumstances of a mass mortality of herring in the shallow part of a fjord in Norway, Oug *et al.* (1991) reported far-reaching effects on the benthic community with a recovery time of three years.

Invertebrate animals are discarded from virtually every bottom trawl haul. These are largely ignored by seabirds. Fragile animals damaged through capture may provide a further food source for submarine scavengers.

The fate of discarded material not exploited by seabirds has not been examined in any great detail. No quantitative data exist to allow us to estimate the actual quantities involved.

6.5 Lost Gear and Other Fisheries Debris

Because the data are so limited, it is difficult to evaluate the relative importance of lost fishing gear, as opposed to other anthropogenic activities and natural processes, as a cause of mortalities to marine organisms. One can, however, to some extent document the level to which marine litter and debris due to man's activities is present in the marine environment and the relative amount attributable to fishing. What follows relates primarily to those items of debris, fishing generated or otherwise, having a direct and potentially life-threatening impact on marine life. It is important to note that much of the debris found in the oceans may be either inert (glass, etc.) or biodegradable (paper, organics, etc.) and, while aesthetically unpleasing, of little consequence to marine life.

As noted in Section 4.4, the only quantitative information on lost fishing gear in the North Sea is the unpublished MAFF Cirolana 7/91 data (Pope, pers. comm.). The trawl used in this survey collected 2.17 kg/hr of debris, of which 1.07 kg (49%) was directly attributable to fishing activities. Discarded or lost netting made up 24% (0.528 kg/hr). In addition, a paper, based on visual observations, describing the overall and relative distributions of floating marine litter in some areas of the North Sea indicated that nylon netting and rope represented only a small proportion of the debris encountered (Dixon and Dixon, 1983). Other categories of items (metal and glass, man-made wood items, paper and cardboard, and plastics) were all more frequently observed. As most lost nets would be submerged, this observed frequency must greatly underestimate their true abundance.

It should be recognized that some portion of the debris which cannot be documented as of fishing origin does, in fact, come from fishing activities. Much of the packaging and wastes from the maintenance and operation of fishing vessels and from the at-sea sustenance of their crews is known to be discarded at sea. Although unaesthetic, and whether degradable or not, little of the debris can be shown to be harmful to marine organisms. Exceptions to this, noted in Section 4.4, involve entanglements in nets, packing straps, and other plastic debris and the ingestion of debris by some organisms.

The Canadian Department of Fisheries and Oceans (DFO) data provides some insight into the abundance and hazards of marine debris to fish (Rowell, working doc. 9). For 1,070 sea trips on the Scotian Shelf, onboard observers recorded a total of 260 incidents where marine organisms were either internally or externally affected by plastic debris (98% fish, 2% shellfish). Nearly half of these incidents involved porbeagle sharks (*Lamna nasus*) with strapping bands about their bodies. Over a 38-day period, one Faroe Islands long-liner, directing for porbeagle sharks, captured 26 with packing or strapping bands on them. All showed some level of damage to the skin and in two cases the bands had cut deeply enough for the skin to heal over them. The majority of other cases involved fish which had ingested pieces of plastic bags or rope or were entangled in discarded monofilament nets. For the Southern Ocean, Slip and Burton (1991) noted the entanglement of fur seals (*Arctocephalus gazella*) in plastic strapping as well as plastic ingestion by seabirds. The same authors report fur seals being observed entangled in fishing gear.

It should be noted that for many items, such as packing straps, there is no evidence as to their fisheries or non-fisheries origin.

Canadian "Marine Plastic Debris Occurrence Reports", collected over two years and representing 1,070 fishing trips by 11 countries, give some information on the

dumping of plastic debris on the Scotian Shelf and on the rate at which changes in practice can be achieved if fishing companies, fishermen's groups, and government agencies decide to stop such practices. Over a two-year period, the percentage of Canadian fishing vessels dumping plastic debris at sea dropped from 99% to 42%. This was the result of a corporate decision by one large fishing company to bring all garbage ashore, a fishermen's union educating and encouraging its members to do the same, and provision, by government, of dumping facilities on wharfs.

7 LONG-TERM EFFECTS OF FISHING ACTIVITIES

In the North Sea the direct effects of fishing, other anthropogenic activities and natural processes will act in concert and, in the longer term, give rise to indirect effects. Fishing activities may create indirect effects in several ways. There may be changes in the feeding relationships of organisms, changes in the genetic make-up of populations and other changes, such as changes in the habitat. In the longer term, the mixture of direct and indirect effects makes it extremely difficult to establish causal relationships between the amount of fishing and observed long-term population changes.

7.1 What Can be Known About the Long-Term Effects?

7.1.1 Effects of fishing on feeding interactions

Fishing can affect feeding relationships by causing changes in prey for predators, in predation pressure on prey, or in populations of potential competitors (Gulland, 1971). The direct effect of fisheries is to reduce populations of target or by-caught species. Hence fisheries initially decrease populations of predators, prey, or competitors. However, the immediate reductions in populations caused by fishing can be followed by indirect increases in other populations, or even the same ones. These changes can occur because the fish removed by the fishery no longer consume prey or compete with other fish. Effects need not be direct; linkages can stretch across several interacting populations. Effects are also not always consistent with intuition.

If fishing alters the abundance of populations, there almost certainly will be indirect consequences on other species or groups of species. Food web theorists and community ecologists have shown that it will be nearly impossible to predict what the consequences will be *a priori*, and nearly impossible to attribute changes in an individual prey, predator, or competitor population to fishery-induced changes in a target stock. The cases when it may be possible to link dynamics of predator and prey (or competitor) populations occur when a fishery

changes a population greatly over a short time. Then the immediate trophic effects may be apparent. Unfortunately, the immediate effects are transient, and the final outcome (increase, decrease, or no change) cannot be predicted with certainty for any of the populations, including the one being fished.

Several features of marine food webs contribute to making it impossible to predict indirect (or long-term) effects of fisheries. One is simply that for all but the upper levels, the North Sea food web is poorly known (Anon. 1988b, 1989c, 1991e). Differential grouping of species for different parts of a food web places many restrictions on the inferences which can be drawn from analyses of the web (Hall and Raffaelli, 1991; Schoenly and Sugihara, in press). These are data-based limitations, but there are also other problems.

Important species, including many fish stocks, feed on different types of prey at different ages, sizes, or life-history stages (Osman *et al.*, 1990; Tschumy, 1982, many others). Because of this life-history omnivore, feeding loops and indirect predation are common. That is, a predator on one life history stage may be prey of another. Potential competitors eat each other. One life history stage may eat competitors of another life history stage. More complex relationships are plausible, and certainly occur (Polis *et al.*, 1989).

Several different modelling approaches to these complex relationships all show that if one investigates the consequences of a change in one population, the results are indeterminant. Consequences will vary depending on initial abundances of all the species, the structure of the model formulations, and specific parameter values for their interactions. Models indicate that although the initial responses of populations may be predicted subsequent population responses may amplify, mitigate or even reverse the initial responses (Polis *et al.*, 1989; Pimm and Rice, 1987; Yodzis, 1988; Hastings and Wollkind, 1982; Werner and Gilliam, 1984).

The analytical studies have shown that intuition may be misleading as a way to investigate indirect effects of fishing on other populations. Nonetheless, intuition will continue to be used. It is important to set clear limits on its utility. For a Group of interacting predators, prey, and competitors, it may be possible to predict the distribution of outcomes (i.e., the proportion of species which will increase, decrease, or not change), but not the long-term response of any individual species (Sugihara *et al.*, 1989; Werner and Gilliam, 1984; Polis *et al.*, 1989; Pimm and Rice, 1987; Pimm, 1991). This suggests that long-term effects of fishing may be reflected by summary attributes of groups of species (i.e., species diversity, community size composition), but not by attributes of single populations.

An important exception to the rule of limited predictability is the likely elimination of a vulnerable species by sustained over-exploitation, but in general the ability to examine effects on single species is limited. It is certainly possible to predict the short-term effects of fishing on the species harvested, and on other species for which they are important prey, predators, or competitors (where "important" is poorly defined, but dependencies must be very strong). Studies of press perturbations (Bender *et al.*, 1984) of model systems suggest the short-term effects expected to last at most twice the average life span of species experiencing mortality before other indirect effects may dominate the initial direct ones. These indirect effects may be even in the opposite direction from the initial effect (Yodzis, 1988). Moreover, fishing is not the only influence on the numbers of animals. The little regularity to be expected from marine food webs will be diminished by other factors affecting recruitment levels, growth rates, and survivorship (Wollkind *et al.*, 1982; Polis *et al.*, 1989).

7.1.2 Effects of fishing on the genetic make-up of populations

Genetic effects arise from selective harvesting of populations. Such selection may reduce the genetic diversity of the populations harvested. The genetic changes might alter life history characteristics, and thereby the ecological requirements of the populations over time.

Fish or invertebrate stocks which are subject to heavy fishing pressure have seldom declined to such low levels that the loss of genetic diversity has become a question of prime importance. Recently, however, there is evidence, both from the field and the hatchery, that genetic diversity may be reduced by heavy harvesting even at high population levels. Smith *et al.* (1991), in a study of Orange Roughie in New Zealand, found a loss of diversity as older and larger (more heterozygous) individuals were selectively removed from the population.

To study the loss of genetic diversity of populations due to harvesting, it is necessary to know something about the genetic make-up of the stock. This information is rarely available, although information is accumulating, particularly for salmon stocks. There the evidence of loss of diversity is clear, but it is unclear how much is due to harvesting and how much is due to many other human interventions such as salmon farming, enhancement projects, and hatcheries.

Law and Grey (1988, 1989) have examined the evolution of yields in age-structured populations. Blythe and Stokes (1990) considered the effects on both fitness and yield of size selective harvesting. There is now much theoretical interest in this field; particular emphasis has been placed

on the way in which size selective fisheries may cause changes in growth rate and age/size at maturity.

These processes feed back into the effects of size selection on sustainable yield. Such theoretical work has implications for viable harvesting strategies. The work also may provide a means by which we can characterise the dynamics and health of fish stocks.

In parallel to theoretical work, there is an increasing amount of laboratory and field work (e.g., Rowell *et al.*, 1989, and the recent conference "The Exploitation of Evolving Populations", Julich, 1992) in which attempts are being made to measure both selection differentials and selection responses. Interpretations are not simple, however. Gene/environment interactions are common and important, so correlation studies can be inconclusive. Also harvesting changes densities, as well as gene-pool characteristics, so life history traits may alter simply due to different densities of populations, rather than different gene frequencies.

Sections 3 and 6 show that fishing is a major source of mortality on adults of many fish stocks. Studies of fish population genetics are just beginning, but there are indications that such high levels of fishing mortality are likely to alter genetic compositions of fish populations. Such alterations are important for at least two reasons. First, when the genetic make-up of populations is changed by fishing, stocks will not return to pre-fishery conditions if fishing pressure is reduced. Second, altered genetic structures make it likely that the stocks have reduced ability to cope with other challenges, due to the strong directional selection for characteristics compatible with high levels of exploitation. Both possibilities represent important indirect consequences of fishing, but the field has not progressed far enough to evaluate the possibilities rigorously for any wild fish stock.

7.1.3 Other effects of fishing

Other effects of fishing are reviewed in Sections 4 and 6. They include resuspension of sediments, altering nutrient availability, and physical alteration of the sea floor including removal and/or displacement of benthic organisms.

Long-term cascading changes in community structure may occur if 'keystone' populations are adversely affected by fishing, leading to marked changes in the pattern of predation and/or competition. Competition can be for space as well as for food (Paine, 1974; Roughgarden, 1986), so the distinction between strictly habitat effects and trophic effects is not perfect. For example, cascading effects may occur when the intensity of predation is reduced by the removal of fish predators. If there is a strong linkage between fish and their benthic prey, their removal may lead to an increase in the

numbers of a competitively dominant species which would otherwise be eaten. Such 'keystone' species effects have been demonstrated experimentally for some rocky shore communities (Paine, 1974, 1980), but manipulative experiments on soft-bottom benthic habitats have generally failed to produce such dramatic cascading responses (e.g., Summerson and Peterson, 1984; Hall and Raffaelli, 1991; Hall *et al.*, 1992). Indeed, from available evidence to date, it would appear that the majority of linkages between species in food webs are feeble (Paine, 1992; Lawton, 1992) and the probability of a dramatic cascading interaction after trophic links are broken appears, therefore, to be low.

It is important to distinguish between the effect of breaking strong functional links between species and effects resulting from simple changes in the availability of secondary space. As examples of the latter, the removal of any reef building species, such as mussel beds or *Sabellaria*, will inevitably result in large changes in the species associated with the reef structure itself. Changes in the structure of primary space may also be important and most obviously occurs when boulders and rocks are removed or disturbed by fishing gears. Reef building species represent one end of a spectrum from sediment stabilisers to de-stabilisers. Because changes in sediment stability can indirectly affect other members of the community, it is possible that fishing removes species which are important in this respect, and that this leads to knock-on effects on other community members. In general, it seems likely that sediment stabilising species will be most affected by fishing and that the resultant destabilisation of sediments may then lead to other community changes. At present, however, such changes remain purely speculative and, even if fishing effects are left aside, categorical statements about the role of many benthic taxa for sediment stability are difficult to make (Nowell and Jumars, 1984; Jumars and Nowell, 1984). Even classical hypotheses which invoke sediment stabilisation as an important mechanism to explain observed patterns in benthic communities (e.g., "Trophic Group Amensalism", Rhoads, 1974) have been difficult to extend beyond the habitat for which they were originally formulated (Posey, 1986, 1990; Thrush, 1986a,b; Commito, 1987; Commito and Boncavage, 1989).

In the case of trawling in offshore habitats, it is now incontrovertible that some macro-benthic animals get killed (e.g., BEON, 1990). However, the effects of this mortality on the populations of the affected species are less clear. Short-term studies can be expected to yield reasonable estimates of mortality rates for species of interest, but the persistence of such local effects will be determined by the re-invasion of adults (either by passive transport or active migration from unaffected areas) and by larval recruitment. The degree to which these latter processes of recolonisation and recruitment will mitigate the effects we observe after trawling depends critically

on the population dynamics of the affected species (e.g., generation time and life-history strategy), the behaviour of the affected species, and the patchiness of disturbance. This latter factor determines the extent of the areas from which re-establishment can take place. The frequency of disturbance in relation to the rate of recovery is another critical factor.

One general effect of fishing disturbance that has been suggested for benthic communities is that overall productivity may increase. This change may arise through the direct effect of fishing on populations of long-lived slow-growing taxa, leading to replacement by smaller faster-growing taxa whose populations are better able to respond numerically to continued disturbance (Reise, 1982). It is often suggested that such shifts also lead to changes in other community parameters such as species diversity which is, for many, an important intuitive yardstick for measuring environmental effects. It is important to point out, however, that not all levels of disturbance will necessarily result in lower community diversity. On the contrary, current ecological theory would support the idea that intermediate levels of disturbance would result in an increase in the values for this parameter (Connell, 1978; Huston, 1979).

7.2 Long-term Changes in the North Sea

7.2.1 Benthos

7.2.1.1 Changes

The possibility that disturbance by fishing can have direct long-term consequences for North Sea macro-benthic populations or communities cannot be discounted. On local scales, particularly in inshore habitats, there are documented cases of the loss of target populations, such as oysters, the loss of especially sensitive solitary species such as seapens or cup corals, and the destruction of reefs built by species such as *Sabellaria* or calcareous algae (Riesen and Reise, 1982; Fowler, 1989; Brown, 1989). In the case of reef destruction, indirect effects on the faunal community associated with the reef will probably be profound (Section 7.1.3) and recovery is likely to be protracted. There is also evidence for localised habitat changes where encrusting epifaunal populations are affected by the removal and disturbance of rock and boulder substrates (Section 4).

While the changes outlined above undoubtedly occur and are a legitimate concern in local areas of specific conservation interest, the evidence for more widespread and general regional effects is unclear. One suggested long-term consequence of fishing disturbance at regional scales is a decline in the populations of taxa sensitive to physical damage and an increase in the populations of less fragile taxa. For example, it could be argued that species such as hermit crabs and sea stars will increase

in abundance because they can scavenge on the animals which are left damaged after fishing. There appear to be no investigations of this type of possible long-term impact of fishing.

It should be recognised that while some progress is being made in estimating the frequency and patchiness of disturbance (see, for example, Rijnsdorp *et al.*, 1991), a quantitative mechanistic understanding of the processes underlying recolonisation and recovery is lacking. Without such mechanistic understanding, effects observed at practical scales of observation are difficult to relate to possible effects at the larger scales of interest to environmental managers. The need to address the problems of appropriate time and space scales has led to a number of research programmes which have looked for differences between benthic communities differing only in the level of fishing disturbance they experience. This approach has been adopted, for example, in BEON (1991) where a sandy area was compared with an adjacent area ('Borkum Reef') in which the sandy substrate is interspersed with boulders, preventing or limiting fishing effort. This study failed to detect any significant differences in the benthic community of the area which could clearly be attributed to beam trawling. However, the "untrawled" Borkum Reef was later found to have been trawled heavily as well (BEON, 1992).

The disappearance of *Sabellaria* reefs from localized inshore areas of the North Sea provides another example. *Sabellaria* is a specialized polychaete worm, usually associated with gravelly substrates. In favourable conditions, it can form extensive colonies consisting of rigid sand-tubes, which may persist for several years. Riesen and Reise (1982) ascribed their disappearance from a location in the German Wadden Sea to destruction by shrimp-fishing trawls. Mussel beds have now encroached over parts of a nearby area subject to similar change (Riese and Schubert, 1987). The longer-term consequence, therefore, appears to be a major shift from one assemblage type to another. The same explanation has been offered for a reduction in reefs off parts of the UK coast (Mistakidis, 1956; Warren, 1973; see also Graham, 1955; Rees, 1982).

The logical expectation for a locality with a persistently high frequency of trawl disturbance would be the favouring of r- over K-selected species, as noted by de Groot (1984), among others. In this respect, intermittent effects on populations of 'resilient' species might be considered intrinsically less significant than for *Sabellaria* and *Arctica*. In the former case, the re-development of reefs may take several years, while the re-establishment of mature populations of the latter would similarly be an extended process, by virtue of its long life-span. In both cases, variability in recruitment success adds a further element of unpredictability to recovery rates. Boesch and Rosenberg (1981) predicted that, in

general, recovery times for the macrobenthos of temperate regions would be less than five years for shallow waters (including estuaries) and less than ten years for coastal areas of moderate depth. Clearly, the recovery of communities characterised by the presence of long-lived species with low recruitment (exemplified by the bivalve *Arctica*) would be much longer.

The magnitude and persistence of dredge effects clearly vary with many environmental and operational factors. The most vulnerable habitats and benthic communities are associated with very localized inshore rocky areas; the least vulnerable are those on intertidal mobile sandflats.

The above examples necessarily dealt with local impacts of fishing on benthos. An alternative approach to assessing the potential effect of fishing disturbance was used by Daan (1991) and is summarized in Section 6. The results suggest that on a regional scale fisheries have a relatively small impact on the biomass of benthic populations, compared to other sources of mortality.

7.2.1.2 Perspectives

As described in Section 3.1, there are no time-series data on benthos which allow long-term trends (i.e., spanning several decades) in the North Sea as a whole to be identified or interpreted.

Inshore species and communities are vulnerable to a wide range of natural and anthropogenic sources of mortality but, for those inhabiting sandy substrates in exposed areas, they are generally the most resilient. The consequences of anthropogenic influences for the fauna of stable deposits (inshore and offshore) may be considered to be more significant, because of prolonged rates of recovery due to the longer life-spans and intermittent recruitment success of a number of the characterising species.

Assessments of the distribution and abundance of conspicuous species relative to known sources of mortality are useful means to give a perspective on the importance of fishing practises. As such, the selection of appropriate 'indicators' for future quantitative study is to be recommended. The Study Group did not discuss how they should be identified or monitored. There has been particular interest in *Echinocardium cordatum* and *Arctica islandica* (see Section 6.3.1) where the scale of fisheries-generated mortality may be such as to far exceed other sources of mortality. Swept area estimates for the North Sea suggest that in a number of sub-areas, fishing pressures should be sufficient to significantly reduce populations of these two species. However, reference to the results from the 1986 North Sea Benthos Survey (Figures 7.2.1.1 and 7.2.1.2) shows that populations of both are widespread and - at least for

Echinocardium cordatum in the central southern North Sea - consistent with earlier studies (e.g., Ursin, 1960).

However, alternative effects of a more subtle nature may also be significant, e.g., regular disturbance may favour opportunistic or physically robust species. This may be reflected in changes in community structure or function, with consequences for higher trophic levels, but not necessarily in significant mortality - and certainly not global extinction - of conspicuous species. It is clear that further work is required on the processes determining population and community responses to physical perturbations before definitive statements can be made. Direct mortality rates have been measured for few species and on local scales. We know little about trophic linkages of benthos, about benthic-pelagic coupling, about habitat specificity of benthic populations and factors which influence recruitment and recolonization.

Moreover, it is not entirely clear what an "appropriate" scale for analysis would be because we know so little about the larger-scale processes involved in recruitment and recolonization in benthic communities. What is clear, however, is that documenting effects with reference to single trawl tracks is insufficient and that the time and space scales on which the ecological effects of fishing need to be assessed do not match the scales on which most studies have been conducted. Longer-term studies are required if the recovery of affected communities is to be assessed.

Species occurrences, or community types, are largely determined by the nature of the physical habitat; criteria by which to judge the significance of anthropogenic influences should therefore take account of locality- and habitat-specific factors.

7.2.2 Fish

7.2.2.1 Trophic effects

As summarized in Section 7.1.1, the catholic taste of many predators, and large suite of species feeding on most types of prey make it difficult to quantify ways in which effects of fishing reverberate among species related by feeding. Moreover, variance in biological populations in response to a dynamic physical environment provides a noisy setting in which to investigate such effects. Two avenues have some promise. One is to use empirically parameterized models which include only the dynamics of those species whose feeding relationships are quantified. Here the starting conditions and parameters are known, so models can provide useful information about interactions among fish populations, if only over time scales of a few years to a few decades. The other approach is to examine summary statistics of the fish assemblage, without trying to resolve what caused

observed changes in individual species, or forecast how individual populations will change in the future.

7.2.2.1.1 MSVPA

Multispecies Virtual Population Analysis (MSVPA) procedures have been applied to the North Sea fisheries by the ICES Multispecies Assessment Working Group. The methods and results are presented in recent reports of that Working Group (Anon. 1988b, 1989c, 1991e). MSVPA and MSFOR (the forecasting version of MSVPA) allow quantitative investigation of many problems associated with the indirect effects of fishery practices on North Sea fish populations. In particular, MSFOR predicts the responses of prey populations (by age) to changing abundance of dominant fish predators, and the responses of predator populations (by age) to changes in their prey. Typical outputs are illustrated in Table 7.2.2.1, extracted from Anon. (1989c). This table shows the impact of each species of predator on each species of prey and can be formatted to show the sensitivity of each predator to changes in the abundance of each prey.

Presently, the predators modelled are cod, whiting, mackerel, saithe and haddock; prey include cod, whiting, haddock, herring, sprat, sandeel, and Norway pout. Other species of predator and prey can be added, but only if stringent standards for data availability are met. The species included are the ones of major commercial importance in the North Sea.

MSVPA and MSFOR can be used to explore quantitatively how changes in fish populations due directly to fishing harvests may have additional consequences on other stocks, and even on themselves, due to feeding linkages. The quantitative results underscore the points made in Section 7.1 about the indeterminacy of the consequences of perturbations. To illustrate, MSFOR was used to predict the percentage changes in total biomass, spawning biomass and catch, following a 30% reduction in fishing mortality by the roundfish fishery (Gislason, in prep.). Even the four species harvested directly by the roundfish fleet did not show matching population responses to identical reductions in F . Cod biomass increased greatly, saithe, somewhat less, whiting changed very little and the total biomass of haddock actually decreased (Table 7.2.2.2). When a modest amount of recruitment variation was added to the forecasting runs, even the direction (increase or decrease) in which biomass changed in response to the reduced F s on roundfish was undetermined for haddock, sprat and sand eels. Only the prey populations of herring and Norway pout were reduced consistently when their predators experience higher survivorship (Table 7.2.2.3). When effort was reduced by 10% in all fisheries, biomass still declined for haddock and Norway pout. Among the other stocks there is a five-fold difference in

the magnitudes of response to the consistent reduction in F (Table 7.2.2.4). The 10% reductions were forecast with constant recruitment. The five-fold difference among stocks would increase had recruitment variation been added to the runs.

These examples illustrate two important points. The first is the value of MSFOR as a research tool to investigate how fisheries affect stocks in the long run. The second is a validation of the warning in Section 7.1 that intuition and qualitative models can mislead when used to evaluate indirect consequences of fishing. MSVPA and MSFOR for the North Sea constitute one of the few instances with sufficient data to specify starting population sizes and linkage parameters. Even the directions of response of key prey populations are indeterminate and magnitudes of responses differ greatly in response to a fixed change in harvest. Moreover, even these results oversimplify reality. MSFOR contains no provision for density dependent growth processes, although such processes may be important in real populations. Adding such further complexities will only increase the range of outcomes possible, given some simple change in pattern or intensity of harvest.

7.2.2.1.2 Measures of assemblage/community structures

Ecologists have developed an array of summary measures of characteristics of communities or assemblages. The number and variety of indices that have been applied to species abundance data, e.g., diversity indices, have been reviewed in several texts (Pielou, 1975; Washington, 1984; Magurran, 1991). The concept of species diversity contains two components: species richness and the evenness of species abundance. The numerous indices used in the literature vary in their sensitivity to one or other of these two components. Consequently, no one index can convey all the information required to distinguish all possible changes in community structure that can occur over time, or be found between habitats. A suite of carefully chosen descriptors is required, each chosen because of its specific response to a particular type of variation in community structure. In this way, the dynamic response of an assemblage of species to fishery-induced changes in the abundance of target species may be captured. These indices should call attention to noteworthy alterations in assemblage structure, if the value of a summary index changes abruptly.

Analyses of the size spectrum of biomass has been useful in limnological applications (Sprules and Munawar, 1986). They convey information about changes in the distribution of biomass among size categories. In particular, the slope may reflect system productivity, transfer efficiency, and how fishing pressure has altered community structure (Sheldon *et al.*, 1972; Kerr, 1974;

Murawski and Idoine, 1989). This approach has been applied to the fish assemblages of several marine regions. Compared to Georges Bank, or the Faroes Bank, the North Sea has relatively less biomass in the large size groups (Pope and Knights, 1982; Pope *et al.*, 1988; Pope, 1989) (see Figure 7.2.2.1). This has been explained as a possible direct consequence of more intensive harvesting in the North Sea. Because of the relationship between fish size and diet (Daan, 1973; Basimi and Grove, 1985), differences in the slope of the species aggregated number at length relationship imply differences in trophic structure in the fish communities of the three regions. Such analyses do not forecast what further changes would occur if fisheries for individual species were to change, or what the unfished biomass spectrum would look like.

In Figure 7.2.2.2, the log-length frequency distribution of fish caught during the English Ground Fish Survey are presented for two years (1977 and 1991) for all NSTF areas separately in comparison with the overall North Sea slopes. Such data refer typically to the fish community caught by the bottom-trawl gear and not necessarily to the entire fish community, because the pelagic species and sandeels are not effectively caught. From these data, it would appear that there are consistently more larger fish and relatively fewer smaller fish in the northern areas compared to the southern areas. Also, there appears to be a difference between the earlier and the later year, with fewer large fish in the recent year. This might be indicative of a change in exploitation regionally and temporarily. However, more extensive analyses are required to validate these conclusions.

There are many multivariate methods for the analysis of patterns of communities. These analytical tools, such as CANOCO (ter Brack, 1989), can also reveal changes in the organization of groups of species in response to external influences (e.g., Iglesias, 1981; Mirza and Gray, 1981), including fishing pressure. For illustrative purposes, these various analyses were applied to the International Young Fish Survey data collected by Scottish fisheries research vessels over the period 1972 to 1991 within three regions of the northern North Sea. Among the results (see Section 9.1), species diversity decreased further away from the British northeast coast and may have cycled between 1977 and 1991; the fish assemblage of the most offshore area had a larger percentage of bigger fish than either of the two other areas from 1972 to 1984, but not from 1986 onwards; patterns of species abundance varied between 1972 and 1983, but became more stable thereafter. The underlying reasons for these spatial and temporal differences in pattern remain unclear. However, they point out specific times (1983-1984) when the structure of the fish community appears to have changed, and where boundaries between different communities may exist. The data are from only part of the North Sea, and are subject to any

sampling deficiencies of the survey. Nevertheless, the analyses show the value of these methods to identify when and where some structuring processes have operated.

Most assemblage analysis methods are methods for describing patterns. As with MSVPA, these quantitative and qualitative tools for analyzing patterns in biocoenosis structure have potential; they also have their limitations. The first major limitation is that process cannot be implied directly from pattern. All the processes which might link fishing practices to changes in the fish community will remain unknown after application of such analytical tools; however, the analyses should aid in the design of the necessary process-orientated studies. A further limitation with this type of analysis is that it presupposes that appropriate data exist. As reviewed in earlier sections, there are serious data gaps. Without filling these gaps, only the most speculative of analyses can be undertaken.

7.2.2.1.3 Analyses of "Key" or "Indicator" species

There may be species of special interest in the North Sea. The Study Group did not discuss how they should be identified or monitored.

7.2.2.1.4 Food web models

Food web models have been proposed and discussed for the North Sea (Jones, 1982; Steele, 1974). As noted in Section 7.1.1, however, they are unlikely to provide clear and useful insights into specific effects of fishing on the dynamics of interacting populations. Where there is detailed knowledge of trophic linkages, such as for linkages between commercially exploited species of fish, MSVPA already provides a rigorous and quantitative tool for examining the effects. Different predators in MSVPA have been shown to have different slopes for their size preference functions (Anon., 1989c). Differences in slopes of size preferences have large impacts on feeding interactions, so simplifying assumptions of general feeding models are likely to be wrong in important features. Much work must be done before food web models can be used to evaluate the effects of fisheries on North Sea fish populations.

7.2.2.1.5 Alteration of life history characteristics

Within ICES, much effort has been devoted to the study of variation in life history characteristics of exploited fish species. Many abiotic as well as biotic factors have been documented to be correlated with growth or maturity parameters of North Sea fish species. In general, it has not been possible to relate any of these parameters to indirect effects of fishing with any certainty (Daan *et al.*, 1990a). The Study Group decided that there was little to

be learned from this type of approach to investigating the indirect effects of fishing.

7.2.2.2 Habitat mediated effects of fishing

The Study Group was unable to address the quantification of habitat mediated effects of fishing practices on fish. Data on the impact of fishing gears on the physical environment are incomplete (Section 4.3). Even with better data on habitat alterations, very little is known of the habitat affinities of many North Sea taxa. Even in cases where something is known of the patterns of habitat usage, actual habitat requirements have not been documented. Both data bases and analytical tools are inadequate for quantifying the habitat - stock relationships for any fish species in the North Sea. However, incubation or rearing habitats may be sensitive to disturbance by local fishing activities, with possible consequences for recruitment.

7.2.2.3 Genetic effects

There are a number of studies which indicate that fishing has altered the life-history characteristics of North Sea stocks. The observed changes in growth, maturation and fecundity of North Sea plaice are consistent with the levels and patterns of fishing mortality (Rijnsdorp, 1992). Rowell (1992) suggests that there is a decrease in size at maturity for North Sea cod, and that selection due to fishing explains this trend better than alternative explanations. Other analyses of cod growth and maturation provide different estimates for the portion of any changes that can be attributed to genetic effects.

There is a developing ability to use quantitative population genetics to help to identify optimal, or even viable, harvesting strategies for fish (i.e., the conference on "The exploitation of evolving populations", Julich, 1992). Where they have been applied, they indicate that current levels of harvesting are much too high for some key North Sea stocks, and could be causing large and undesirable changes in the life histories of the species being harvested (Stokes, 1992).

The integration of quantitative population genetics with fisheries management is a field in its infancy. It may provide powerful tools for evaluating the long-term consequences of fishing practices. Even at this early stage it is providing warning that harvesting levels are dangerously high for some stocks. The models do not make clear what the population genetics response would be, though, were fishing pressure reduced.

7.2.2.4 Perspective

A few direct effects of fishing on fish populations are clearly evident. The changes in age composition of fished stocks in the North Sea have been conclusively linked to

harvesting. The evidence that fishing has changed other life history attributes, such as growth and maturation rates, is less complete, but fairly strong and growing. The slope of the biomass at length plot is steeper for the North Sea than that for other seas which have been examined. That could be interpreted as indicating that the combined biomass of fish has been fished more heavily in the North Sea, but it could also be the consequences of basic differences in community structure (Anon., 1991e).

Despite the long history of relatively heavy fishing mortality, there are only a few population collapses which can be linked directly with fishing (Section 3). The decline in mackerel in the late 1960s was almost certainly caused by very large removals. The collapse of the North Sea herring in 1976-1978 has been linked to over-harvesting. However, the recruitment failure which followed can be explained by anomalies in North Sea circulation affecting recruitment processes, as well as by depressed spawning biomass. Whatever the cause, the stock rebounded strongly less than half a decade later. Fishing mortalities on both cod and plaice have been increasing steadily for recent decades, to values twice as large as they were in 1960. Sole showed a comparable rise in F_s in the 1960s. Plaice and sole have remained strong, but current levels of effort on cod and haddock are high enough to cause serious concern in ACFM. There are fears that further reductions in spawning stock biomass could lead to recruitment failure. Fisheries may have also changed the abundance of non-target species, taken as by-catch, such as skates (Purdom and Garrod, 1990).

There is a lot of variation in all the fish populations, including those harvested by the industrial fisheries. However, few of the fluctuations can be attributed directly to fishing. Most variation seems recruitment driven, rather than harvest driven. Not a single strong stock recruitment relationship has been found for a North Sea stock, with the possible exception of herring and mackerel. Therefore, even if removals by fisheries have reduced some stock biomasses, it would be difficult to link the direct effect of removals to longer term changes in recruitment. The major fluctuations in recruitment (and subsequently in biomass) seem more successfully linked to physical oceanography effects than to biological consequences of fish harvests. This applies particularly to the major changes in biomass levels for species groups, such as the "gadoid outburst" and the shift in small pelagics in the 1980s (Jakobsson, 1991). However, alternative explanations based on changing predator levels (Andersen and Ursin, 1977) and food supply (Jones, 1982) have been proposed as well.

The data available are inadequate to demonstrate major and consistent impacts of fishing on population dynamics of even the individual stocks harvested, although some

life history attributes have unquestionably changed. Given the difficulties in seeing or predicting indirect effects of population changes (Section 7.1), we should not be surprised at the paucity of documented indirect changes. Certainly if we add to our understanding of the biological and physical processes operating in the North Sea (and how they interact), we may be better able to identify and study indirect and long-term effects of fishing. Perhaps the collection of more and better data will allow more long-term effects to be seen. Perhaps whatever major changes did result from fishing occurred many decades ago. Then, with current levels of fishing effort comparable to historic levels for most roundfish, at least, perhaps we should not expect fisheries to make major new changes in fish communities now. This is not arguing that a new equilibrium has necessarily been established. Rather the present levels of perturbation constitute the normal condition for the duration of our data series. With that reasoning, it would be appropriate to monitor new fisheries, and fisheries where effort shows major changes.

7.2.3 Seabirds

Seabird numbers are relatively easy to count in comparison to fish numbers. As a consequence, their populations have been tracked for a century in some cases (Section 3.1). Numbers of 15 of the 24 species of seabirds breeding in the North Sea have increased in recent years. Numbers of 7 species have remained constant, or have not been surveyed adequately. Only two, herring gull (*Larus argentatus*) and roseate tern (*Sterna dugale*), have declined in number (Lloyd *et al.*, 1991). Adverse effects of any anthropogenic processes are, therefore, either relatively small or are masked by positive effects of other processes on a North Sea scale. Seabirds are affected by variation in many natural processes, so it is difficult to isolate effects of fishing from other factors.

7.2.3.1 Trophic effects

Significant alteration of the structure of marine fish populations by man's fishing activities will impact bird populations. Seabirds feed on intertidal and nearshore benthos, pelagic invertebrates, pelagic and young demersal fish, and fishing discards and offal. The quantification of man's indirect effects is complex, however (Section 7.1.1). The inherent complexity of the problem is exacerbated by the lack of knowledge of offshore seabird diet in the North Sea and the known plasticity of the diet elsewhere. There is some knowledge of chick diet of the most important species during the breeding season, determined through observations at colonies. In addition, some researchers have examined stomachs of dead birds washed up on beaches, or have collected birds at sea, usually in relatively inshore and sheltered sites. Very few studies have been carried out further to sea.

Knowledge of estuarine bird diet is reasonably good; relatively few species prey directly on mussels, cockles or other harvested species. In local situations, these prey may be important for some species.

Fisheries may compete directly for the same prey as seabirds. In the North Sea, sandeels, sprat and young gadids are taken both by many birds and industrial fisheries. In most cases where fish stocks have declined in a linear fashion (as opposed to catastrophically), fisheries become uneconomic before seabirds are severely affected. Inshore fisheries for mussels and cockles compete with bird populations feeding on these shellfish. Fishing has thus been connected with the apparent collapse of the mussel and cockle stocks in the southern Wadden Sea in the winter of 1990/1991 which led to starvation and mass mortality of eiders in the area (de Kuip, 1991).

Catastrophic declines in fish stocks can affect both seabirds and fisheries. There are numerous cases overseas. The effect of the El Niño/Southern Oscillation in the Pacific is perhaps the best known. Here the collapse in the anchoveta fishery was paralleled by a collapse in seabird populations which have never recovered to their pre-collapse levels. Decreases in fish recruitment affecting seabird populations have occurred in the Shetland sand eel stock, and the North Sea sprat stock. Blake (1984) considered that the collapse in this latter stock was partly responsible for a mass mortality of auks in the western North Sea in winter 1982/1983.

The proximate cause of the above collapses in pelagic stocks seems to be recruitment failure. Such stocks rarely show strong stock recruitment relationships, but do show marked environmental correlates with the production of very weak (and very strong) year classes. Therefore, it has proven extremely difficult to isolate the contribution of harvesting to the collapses, although harvesting may play a role in accelerating or prolonging the collapse. Hence, although seabirds may serve as useful monitors of the status of some fish stocks, at least locally, they are not monitoring the impacts of fisheries on the stocks (Rice, 1992). Moreover, because collapses occur even in absence of fishing (Souter and Isaacs, 1974; Sherman *et al.*, 1981; Southward *et al.*, 1988), it is impossible to manage fisheries in ways which guarantee that pelagic stocks will never collapse. Questions about the impact of fisheries on seabirds should address whether or not fisheries increase the frequency, duration or magnitude of collapses. More generally, it is important that fisheries for those species on which seabirds (and other marine predators) depend are managed with other components of the ecosystem in mind. Most of these fisheries are industrial, which are among the least known and managed in the North Sea.

Changes have occurred in many fish populations in the North Sea (Section 3.1). Some of these changes in fish populations have had consequences for seabird populations. However, until the linkages among marine species, and the factors which determine recruitment levels of fish, are better known, it will not be possible to quantify the indirect effects of fisheries on seabirds. Even modelling has a limited ability to relate fisheries in the North Sea to changes in fish community structure or to seabird populations. At present, general systems and energetics models are too simple and insensitive, and there is insufficient information on seabird diet to include them in most of the empirical models that are available.

There has been some dispute over the proportion of the fish population in an area taken by seabirds, with Furness (1978) considering that seabirds took about 29% of the pelagic fish stock around Shetland, and Bourne (1983) claiming a figure about one-tenth of this value. Tasker *et al.* (1989) modelled the entire North Sea and found that about 5% of the pelagic fish production was taken overall by seabirds, but that there were hot spots, both of bird predation and pelagic fish production. The ratio between quantities of offal and small fish taken by fulmars was a key factor in determining seabird impact on the North Sea ecosystem.

The spatial and temporal patchiness of seabird predation is widely recognised. Tasker *et al.* (1989) estimated that seabird consumption may differ by an order of magnitude among parts of the North Sea and between months. Impacts may be even greater on more local scales (Furness, 1978, 1982; Furness and Monaghan, 1987; Schneider, 1989).

The complexities of interpreting interactions among seabirds, fish stocks and fisheries is illustrated by the interactions around the Shetland Islands. Table 7.2.3.1 shows the numbers of breeding seabirds in Shetland in 1985-1987. Populations of seabirds have been monitored with varying levels of effort for two decades (Lloyd *et al.*, 1991; Cramp *et al.*, 1974; Bullock and Gomersall, 1981; Monaghan *et al.*, 1989). In general seabird populations expanded through the 1970s and early 1980s. A fishery on the local stock of sand eels, the dominant prey for many of the seabirds, commenced in 1974 and rose to a peak of 52,000 tonnes in 1982. In 1984 Arctic terns showed a near total breeding failure. Through the rest of the 1980s, poor breeding success and other effects of poor food supply were noted for kittiwakes, fulmars, arctic skua, puffin, great skua, razorbill, and black guillemot: all species which forage relatively near the surface. Breeding success of guillemot, gannet, cormorant and shag, which all can dive more deeply, has generally not been affected as seriously. The fishery showed marked declines over the same period, and was terminated in 1991.

Declines in seabird breeding and fisheries were both related to significant declines in the sand eel stocks (Monaghan *et al.*, 1989, 1992). The declines in sand eels resulted from a series of recruitment failures which began in 1982, although spawning biomass remained high for several years thereafter (Bailey, 1991). Environmental effects, including hydrography and predation by other pelagic fish stocks, have been suggested as causes of the recruitment failure. Possibly seabird numbers and the sand eel fishery increased in parallel with an increase in stock size for sand eels, although the status of the sand eel stock was not quantified until the fishery was underway. The subsequent recruitment failures and decline in stock size cannot be automatically taken as evidence of direct impacts caused by fisheries and the case illustrates that even for a relatively local system, the interplay among the hydrography, fish populations, predators, and fisheries is complex.

In general, it appears that of the offshore fisheries that may be in direct competition with seabirds, the industrial fisheries for sand eel, sprat and Norway pout are the most important. To understand how fisheries may be affecting seabirds, it is therefore particularly important to study the harvesting practices of the industrial fisheries.

7.2.3.2 Discards and offal

The enormous growth in numbers of scavenging seabirds in the North Sea has often been attributed to extra food provided by man from fishing vessels (Fisher, 1953; Cramp *et al.*, 1974; Lloyd *et al.*, 1991). This seems reasonable, but the growth could alternatively have been caused by other factors, such as reduction in human exploitation.

Spaans (1971) found that while adult herring gulls fed predominantly on intertidal invertebrates in the Netherlands, chicks were fed mainly on fish. Between 20% and 33% of this fish came from fishery discards, and Spaans argued that this was likely to be important in determining breeding success of the population.

Furness *et al.* (1988) studied the dominance hierarchy in those species of seabirds feeding around fishing vessels in Shetland waters. Larger species, such as gannets, often excluded smaller species, such as herring gulls and kittiwakes. Gannets took mostly whole discarded fish, and could take almost all sizes of discard. Gulls also took whole discarded fish, but could not handle the largest sizes discarded at that time very effectively or rapidly; as a consequence, these were often robbed by other birds. Great skuas preferred to feed on sand eels when these were available, but could utilise discards. The large increase in numbers of great skuas around Shetland during this century may have been sustained by the availability of small haddock and whiting discarded from

the local fishery (Furness and Hislop, 1981; Furness, 1987).

In the southern North Sea, Noordhuis and Spaans (1992) showed that lesser black-backed gulls out-competed herring gulls near trawlers, and have forced herring gulls to switch to other food supplies. This change in food supply has contributed considerably to a decrease in herring gull breeding success since the late 1960s. Although absolute numbers of herring gulls have not fallen, numbers of lesser black-backed gulls have grown proportionately faster.

Some changes might be expected in the future. The decline in whitefish catch (and presumably discards and offal) since the 1970s (Section 3.1), coupled with the continued increase in the more aggressive dominant species of seabirds in the northern North Sea, may have put the less competitive species in this area (herring gull, lesser black-backed gull and kittiwake) under increased pressure to obtain this food. The decline in numbers of these three species in the northern North Sea (Section 3.1, Lloyd *et al.*, 1991) may indicate that this is happening already. Future changes in fisheries practices which affect the production of discards and offal can be expected to affect seabirds which depend on this source of food.

7.2.3.3 Perspectives

Despite the evidence of some local effects of fishing on birds, there is no evidence of a reduction in bird populations around the North Sea due to entanglements, even at a colony level. Such evidence is difficult, sometimes impossible, to collect. It is unlikely that small changes in seabird populations could have been detected if they have occurred. Inshore gill-net fisheries appear to be the most likely to cause seabird mortality. Due to the small boats prosecuting this fishery, it is also one where monitoring and regulation are particularly weak.

Most seabird biologists attribute some of the increase in at least some species of seabirds directly to increased food supply from offal and discards from fisheries. Even here, population changes may also have been influenced by decreases in hunting pressure as well, so even the direct effect of fishing on seabird food cannot be demonstrated unequivocally. Both seabirds and fisheries have suffered when some specific stocks of small pelagics have collapsed (e.g., the Shetlands). However, the collapses in stocks, and corresponding losses of food by seabirds, cannot be shown to be a consequence of fishing. Rather, hydrographic influences seem strongly implicated in the recruitment failures. Because they harvest the same species, some industrial fisheries present the greatest potential for competition with seabirds. Nonetheless, the most recent evaluation of

industrial fisheries indicates that present levels of harvest are sustainable (Anon., 1991c).

7.2.4 Marine Mammals

The life history of marine mammals is such that environmental effects will show up first in reproductive attributes. Consequences of changes in feeding conditions include effects on pup size, milk production, growth rates, size at weaning, birth rates, infant mortality rates and other features (Harkonen and Heide-Jorgensen, 1991; Anon., 1992d). Marine mammals have a life expectancy of a decade or more, and relatively low fecundity, so population consequences of poor recruitment for several years can persist for a long time.

Marine mammal reproduction is also sensitive to contaminant levels (Reijnders, 1986, 1988). There is concern that if contaminant levels become elevated in North Sea marine mammal populations, these populations may become more vulnerable to any additional stresses, including any stresses that may be caused by fishing practices.

7.2.4.1 Seals

The diet of seals in the North Sea has been studied in recent years mainly in response to questions concerning the effects of seals on fisheries. Work by the Sea Mammal Research Unit has concentrated on using analyses of otoliths from faecal samples to estimate the proportion by weight of different species in the diet of grey seals (Prime and Hammond, 1990; Hammond and Prime, 1990). Results show that grey seals eat a wide variety of fish, but tend to concentrate on a few species. In the North Sea, the most prevalent species are sand eels and large gadids, especially cod and, to a lesser extent, whiting. In the southern North Sea, and in Orkney, flatfish also contribute significantly to the diet. The proportions of these species in the diet varied by area and by time of year. There is no information on the local abundance of prey species, so it is not known whether this variation reflects a preference for certain species or predation in proportion to relative abundance.

Aberdeen University, in conjunction with SOAFD, have undertaken a study to investigate the ecology and diet of harbour and grey seals in the Moray Firth, Scotland. Various methods for evaluating diet were compared. The food of both species consisted mainly of inshore fish, but included some offshore fish and cephalopods. Seasonal and yearly changes in the diet reflected changes in the local abundance of different fish species; feeding sites also changed. Seals fed predominantly on small clupeids in winter and principally on sand eels in summer (Thompson *et al.*, 1991). A marked increase in the importance of cephalopods in the summer diet has been observed over the period 1988-1991 (Anon., 1992d).

There is little information on how seal diet may have changed over time as a result of fishing activities. Northridge (1990) has investigated the relationship between grey seal diet and the size of demersal fish stocks off the east coast of the UK for a number of years within the framework of an optimal foraging model. The results suggest that grey seals do exert some degree of preference for certain species; that is, they do not simply eat fish in proportion to their abundance. This analysis assumes, among other things, that the relative abundance of fish available to the seals is reflected by the relative abundance of fish observed on surveys in that area.

The observed diet of seals (especially grey seals) in the North Sea indicates that seals are able to respond to changes in the relative availability of prey species. Fishing in the North Sea may affect foraging distribution and diet composition of seals but, to date, there is no evidence that this has had any significant effect on populations.

Numbers of both grey and common seals have increased in the recent decades, with the exception of the epizootic outbreak in common seals in 1988 (see Sections 3, 4, and 6). The principle cause of this increase is assumed to be the protection from hunting (Reijnders and Lankester, 1990). Some local population declines have been documented (Reijnders and Wolff, 1982; Wug, 1986). Local populations have been affected by contamination with toxic chemicals (see Section 6). The impact of this over the long term is presently unknown.

7.2.4.2 Cetaceans

Relatively little is known about the species interactions involving cetaceans in the North Sea. The harbour porpoise was believed to prey preferentially on herring and gadoid fishes (Ray, 1965, 1973; Gaskin *et al.*, 1974). However, over the period when herring abundance was low, no evidence of a decline in body condition or fecundity was observed (Clausen, 1991), suggesting that they were not short of food. Recent studies of stranded and by-caught porpoises suggest that demersal fish species may be more important in porpoise diets than was previously thought (Martin *et al.*, 1990; Lick, 1991). Opportunistic sampling of stranded animals is likely to provide only biased estimates of porpoise diet; better records would be provided using by-caught animals (Anon., 1992d).

7.2.4.3 Perspectives

Among marine mammal species, seals are the most readily surveyed and population census data are available for most North Sea regions for between one and three decades (Reijnders and Lankester, 1990). Over this period, grey seal populations in the North Sea have either been stable in size, or have increased. The situ-

ation is similar for common seals up to, and in the years following, the sharp decline caused by the phocine distemper virus in 1988 (Heide-Jorgensen and Harkonen, 1988; Anon., 1992d). There is little evidence therefore to suggest that by-catch mortality, or other mortality sources, have had any significant impact on seal populations. However, where mortality has been documented, it frequently involves juvenile animals (Anon., 1992d) and this may hamper the potential recovery of a population reduced by some other trauma, for example, the epizootic in common seals.

The long-term trends in small cetacean populations in the North Sea are unclear due to the lack of quantitative information on historical and present abundance. Albeit still abundant in the northern North Sea (Björge and Øien, 1990), porpoise numbers in the southern North Sea have declined markedly over several decades (Verway, 1975; Kayes, 1985; Evans, 1987; Reijnders and Lankester, 1990). Similarly, bottlenose dolphins are now rare in the southern North Sea, where they used to be relatively abundant (Verway, 1975). There is no evidence to suggest that food shortages arising through fishing have been a contributory factor in these declines.

By-catches of dolphins and porpoises in fishing gears continue to be a matter of public concern; however, the population impacts of this are currently unknown. There is no consensus regarding the actual number and population structure of cetaceans in the North Sea. It is, therefore, impossible to assess the impact of any additional mortality above that resulting from natural processes. However, it has been pointed out that "small cetacean populations, and harbour porpoises in particular (Woodley and Read, 1991), are probably unable to sustain a large additional mortality because they have a low annual reproductive rate and high adult mortality" (Anon., 1992d).

8 POSSIBLE MANAGEMENT OBJECTIVES, CONFLICTS AND SOLUTIONS

Even if the full extent of the ecosystem effects of fishing remains unclear, the exploitation of the living resources in the North Sea undoubtedly affects the structure and functioning of the ecosystem and must therefore be viewed against other management objectives. At the same time, fisheries are likely to be affected by other human activities having an impact on the ecosystem and by the overriding climatic factors over which man has little control. This section discusses potential conflicts and possible ways to resolve them. The scope for integrated management depends largely on the definition of an appropriate set of objectives for the different uses of the North Sea. Therefore, it seems appropriate to discuss first the present situation in this respect.

8.1 Fisheries Management Objectives

The responsibility for fisheries management in the North Sea Task Force area lies within the bordering countries. Since member states of the EC have handed over their responsibility to the Fisheries Commission, fisheries management has to be agreed between the EC, Norway and Sweden. In developing its policy, the EC has to take into account the procedures within member states and possible legal, economic and social consequences of these. The Common Fisheries Policy (CFP), agreed upon in 1983, is based on the free access principle of all member states to the 200-mile zone of the Community waters, with the exemption of some parts lying within the 12-mile zone.

The overall objective of the CFP is the long-term rational development of the fisheries in the member states. Although this objective is primarily an economic one, it is clearly acknowledged that, in order to maintain viable fisheries, steps are taken to guarantee the continued existence and productivity of commercial fish stocks. In the interests of both fishermen and consumers, rules have been introduced for controlling the level of exploitation. These rules are backed up by technical measures to improve the exploitation patterns, special measures to satisfy the needs of coastal fisheries, and enforcement measures.

The basis for controlling fishing effort has been the Total Allowable Catch (TAC) system. The advantage of this system is that it is relatively easy to split fishing rights among member states once the key for partitioning TACs according to historic catch levels had been established. However, a disadvantage is that it is an indirect measure of controlling fishing effort, which does not take into account discrepancies between actual catches at sea and landings (e.g., discards). In practice, enforcement has caused severe problems, which resulted in a deterioration of catch statistics and, consequently, in a reduction in the reliability of the scientific advice. Moreover, the TAC system does not cover all fisheries. In addition to TACs and technical measures, the Fisheries Commission has already developed guideline plans for effort reduction in its Multiannual Guidance Plan. These are likely to be emphasized when the CFP is reviewed in 1993 (Commission of the European Communities, 1991).

Although decision making is essentially a political process, the initial proposals for TAC's made by the Fisheries Commission are based largely on scientific advice. This advice is provided annually by the Advisory Committee on Fishery Management (ACFM) of ICES. Because clear guidelines in respect of the management objectives of the policy makers were lacking, ACFM has developed its own set of scientific criteria for rational exploitation on which the advice was based. The resulting TAC advice was generally in the form of a single

value. A matter of particular dispute in the past has been to which extent economic criteria must be incorporated in the advice. The biological and economic objectives of fisheries management are so interlinked that this problem has never been solved satisfactorily. Recently, there is a tendency for ACFM to become less concerned with the economics of fisheries and to leave it to the managers as to whether they want to prolong a situation of overfishing as long as the stocks are not suffering recruitment failure. As a consequence, the advice has become less definite and the range of possible options from which the managers may choose has broadened. In 1991, ACFM reconsidered its objectives, tasks and the form of its advice (Annex 3), because it was felt that the new attitude towards management of the sea in general required some essential adjustments.

The agreed objective of ACFM is now to provide the advice necessary to maintain viable fisheries within sustainable ecosystems, where an ecosystem is defined as sustainable if management actions (including no action) do not result in irretrievable loss of any component of the system. This is of course not to say that losses may not occur regardless of human activities. Although the sustainable ecosystem is now explicitly incorporated in the basis of the ACFM advice, the reality is that there are at present few objective means to assess the ecosystem in relation to fisheries effects. Traditionally, the advice has been based on single species and sometimes multispecies assessment, and this is not likely to change in the near future. However, ACFM has stated that it is prepared to take ecosystem considerations into account when formulating its advice.

One aspect worth mentioning here is that, in preparing its advice, ACFM is guided by a set of quantitative sign posts, which allow an objective assessment of the impact of the fisheries on individual fish stocks. What is required is an extension of such sign posts for evaluating ecosystem effects.

8.2 Ecosystem Management Objectives

Ecosystem management is aimed at an integration of the various uses of the sea, with particular emphasis on nature conservation or possibly restoration to some perceived desirable state. Sustainability of the development of human activities and the wise use of the sea are an important issue in this respect (Bruntland report). The precautionary approach has been explicitly introduced in the policy adopted in the Declarations of the Second and Third International Conferences on the Protection of the North Sea. The implication is that, in case of doubt as to the effects on the marine environment, preventive or remedial action is to be taken, i.e., any decision should err on the safe side (Hey, 1991). The precautionary approach has been made applicable to all inputs of substances that are persistent, toxic and liable to bioaccu-

multate and was extended to apply to inputs of nutrients and pollution from ships. Fisheries effects do not seem to have been considered in this context so far.

Clearly, the ultimate objectives of ecosystem management are not yet well defined. The North Sea Task Force is in the process of formulating an appropriate set of ecological objectives, but these have not yet crystallized (TF 5/5/15-E). During a recent session of the sub-group on ecological objectives, the following draft definition of 'ecological quality' was arrived at: "an expression of the structure and function of the ecological system taking into account natural physiographic, geographic and climatic factors as well as biological, physical and chemical conditions, including those resulting from human activities". This definition recognizes the local (and frequently habitat-related) scale of practical management problems that are likely to arise.

The Advisory Committee on Marine Pollution (ACMP) of ICES has, rather than formulating objectives, elaborated on a framework for environmental protection (Annex 4). This framework is based on the concept that the combined benefits of the various uses to society must outweigh the combined deficits or detriment. Although science may play a role in deriving limits on the environmental changes that are permitted to occur and also in suggesting actions that will minimize unnecessary environmental changes, setting the ultimate objectives is clearly, just as in fisheries, within the province of managers. Science can only help in defining and selecting appropriate objectives.

8.3 Management Conflicts

In the context of ecosystem effects of fishing activities, the possible links between ecological objectives and fisheries management objectives are particularly relevant. Because fisheries management is founded on the concept of sustainable yields, the ecological and fisheries objectives run to a large extent parallel and contrary to those of some other users. For instance, certain contaminants and mining operations may have negative effects on both the quality of the marine environment and on fish production or, specifically, the quality of fisheries products.

On the other hand, there may be opposite interests, because large-scale fishing operations will undoubtedly have an effect on the fauna. Firstly, any significant reduction in the biomass or change in the relative size composition of a fish stock will necessarily affect the flow of energy through the marine food web. Although it is possible in principle to investigate possible and likely effects of fishing on the system through modelling studies (Andersen and Ursin, 1977), it is generally agreed that the quantitative knowledge about the components lower in the food chain is insufficient to have

confidence in answers from those studies. Secondly, fishing operations may have unintentional side effects (physical changes in habitat, discards, accidental by-catches, etc.), which may cascade through the system. Also in this case, the scale of such effects relative to natural changes in the environment remains largely unknown. Moreover, there is a set of parameters describing ecosystem characteristics (diversity, reversibility, structure in space and time) which at present are difficult to relate to direct or indirect effects of fishing. Thirdly, fishing operations might lead to (near) extinction of species which are taken as a by-catch. The chance for such effects to occur depends on the scales of the fishing operations in relation to the scale of the dynamics of the species under consideration.

In some respects, fisheries, like other human activities, may favour particular components of the ecosystem. This applies for instance to certain groups of birds which thrive on discards and the interest of which runs parallel to present fishing practices. It is quite obvious that conservation and restoration measures may cause changes in the ecosystem which are not entirely satisfactory to all members of society concerned. Recreation of the pristine condition of the North Sea is an utopia. Any description of a reference situation is arbitrary, because the system is not static. Therefore, one may wonder to what extent science can help in developing objective criteria for integrating various types of uses of the marine environment.

8.4 Possible Solutions

Fisheries are sustained by natural productivity. Wise management acknowledges that exploitation is constrained by the potential of the resource to balance the losses due to fishing by growth and reproduction. The unintentional effects on non-commercial fish and benthos stocks must be viewed along the same lines. Wise management, therefore, should also prevent detrimental side effects on other components. The well-managed exploitation of fisheries, therefore, does not necessarily run counter to objectives set for the conservation of other parts of the ecosystem.

Pope's (1984) minimum sustainable whinge model shows the nature of the trade-offs between a selection of possible long-term social, economic and biological objectives. It stresses the avoidance of 'worst' conditions rather than the attainment of 'best' conditions. Such thinking could readily be extended to include the avoidance of environmental 'worst' conditions.

Optimisation of the long-term economic outputs of the fishery generally requires the use of the economically most efficient gear. Such a gear may have greater or lesser effects on vulnerable components of the ecosystem than would a less efficient gear. Thus, in some cases the

ecosystem might be impacted less by employing low levels of an economically efficient gear, whereas in others it might be preferable to use higher levels of a less efficient gear. In a rational world, knowledge of these trade offs could provide managers with an additional set of considerations to balance when deciding on the appropriate measures. In short, we might consider reducing side effects (benthos kills) or waste (discards) as one of the objectives of fisheries management to be balanced with objectives related to catch levels, profitability and employment. A number of traditional fisheries management approaches could be considered in attempts to meet the appropriate objectives. These would include combinations of effort reductions, gear restrictions, closed areas, closed seasons, etc., all of which could, if applied appropriately, reduce the impact on a variety of elements of the ecosystem. The relevance of these measures in the present context is discussed below.

8.4.1 Effort reductions

Fisheries science is based on the concept that the impact of a fishery on a fish stock depends on the relative magnitude of the mortality caused by natural processes (diseases, predation) and by fishing. With the exception of those stocks which present a special case due to their schooling behaviour (e.g., pelagic fish), the fishing mortality experienced by a fish stock is proportional to the fishing effort (e.g., the number of hours trawling corrected for horse power of the vessel). This should apply similarly to most species constituting the unintentional by-catch. Therefore, effort reductions appear to be extremely effective as a means to reduce side effects of fishing.

8.4.2 Gear restriction.

Different fishing gears select various species and sizes of fish in different proportions. The exploitation pattern for any particular species may, therefore, be improved by the modification of certain fishing gears. For instance, mesh size regulations have been introduced in the past for the particular reason of protecting juvenile fish. Also, certain gears have been prohibited (e.g., explosives, long drift nets).

Although gill nets present exceptions in particular circumstances, because they can be employed at locations with high fish abundance where towed gears would become damaged (rocky bottoms, wrecks), active gears are generally more efficient than passive gears. As a consequence, the less efficient passive gears have often been out-competed for economic reasons (e.g., herring drift net fishery).

Different gears vary in their selectivity and also catch non-commercial fish, benthos, birds and mammals in different proportions. Bottom trawls will particularly

affect the benthos, whereas gill nets have a higher chance of entanglement of birds and marine mammals. Each gear could thus be characterized by the differential side effects they produce on non-target species. These effects may also vary depending on where and when the gear is deployed. For instance, the accidental by-catch of birds and mammals in gill nets may vary in relation to the time and place of their usage. Therefore, it would appear that, given enough information on the by-catches, it might be possible to weigh the pro's and con's of the different gears and advise on appropriate restrictions of the various gears in time and space.

8.4.3 Gear modifications

Apart from restrictions on the use of available gears, there is also scope for further development of gear modifications which try to minimise negative effects. For instance, the square mesh has been developed as an alternative to diamond mesh to improve the selectivity of trawls. Investment in research aimed at increasing the selectivity of active trawling gears might be expected to result in significant improvements in this respect, but at present this type of research is mostly concerned with improvements in exploitation patterns of the commercial stocks. However, in order to reduce the by-catch of cetaceans in fishing gear, the development of acoustic equipment has been initiated that deters these animals from the neighbourhood of fishing operations.

8.4.4 Closed areas

Area closures appear to be particularly suitable for reduction of some unwanted side effects. Indeed, within the array of technical measures of the EC fisheries management scheme, area closures, in combination with closed seasons, have been applied efficiently to safeguard nursery areas against exploitation, to reduce catches of undersized fish, and to improve exploitation patterns. Areas closed to all forms of fishing have also been proposed for reasons of nature conservation with particular reference to benthic communities. A problem of nature reserves in the marine environment is that neither the water mass nor the animals are constrained by arbitrary boundaries imposed by man, or at least much less so than in the terrestrial environment. Biological processes are thus not confined to particular localities but depend on inputs and outputs over the borders.

Area closures may be aimed at preserving a particular species, a specific habitat or a community. Therefore, in selecting an appropriate area, the local and regional habitat distribution and migration patterns of the species concerned must be taken into account. The required size must be evaluated against the particular objectives and cannot be generalized.

It should be noted that, if the introduction of a closed area is not associated with a reduction in fishing effort, a direct effect would be a shift in the distribution of the effort. The associated negative effects in unprotected areas might, at a North Sea-wide scale, counteract the positive effects expected in the protected area.

The concept is, apart from possible difficulties in enforcement, attractive from an experimental perspective, because monitoring of such areas could provide insights into the role of fishing as a structuring force. It should be noted, however, that, although any observed changes in community structure would demonstrate that fishing had an effect, they would not necessarily reveal the underlying processes. Only when monitoring studies are integrated with process studies would the experimental closure provide a sound basis for further management considerations. Therefore, further consideration should be given to precisely how closed areas should be used as a research tool.

8.4.5 Closed seasons

Some effects of closed seasons are similar to those of closed areas and they are, therefore, often applied as part of an integrated set of measures. They may be particularly effective in protecting ecosystem components during critical stages in their life history (e.g., spawning time, migration, etc.). In the present context, they might for instance be effectively used if accidental by-catches of marine mammals were related to specific times of the year, when their distributions overlap with those of fishing activities.

8.4.6 Other solutions

In some special cases, economic incentives might present an effective tool in reducing side effects of fishing. These methods are being explored elsewhere (e.g., North Pacific halibut fisheries), but more work is required to identify the scope of this type of solution for management conflicts arising in the North Sea.

Training and awareness of fishermen offer another possibility to reduce the impact. Education has become an integrated part of the CFP and consideration might be given to include the aspect of avoidance of unwanted by-catches as part of the programme.

8.5 Conclusions

From the foregoing discussion, it appears that there is not necessarily an essential contradiction between fisheries management objectives and ecosystem management objectives. Conflicts arise only if the respective protagonists are not prepared to compromise. If conservation of the North Sea is seen as an absolute return to the pristine state, then there is no room for fishing or

any other anthropogenic activities. Similarly, if fisheries are only concerned with maximizing short-term profits and do not care about the effects of their activities, then it is not unlikely that the North Sea ecosystem will change in an unpredictable way and that some of its components might even become extinct.

Fisheries management has not been particularly effective in controlling fishing effort in the North Sea. Fisheries science predicts that the long-term profitability would increase, if the effort were halved in most fisheries. Even if all the present impacts of fisheries are not known, the magnitude of the effects will also be reduced, when effort is reduced across the board. Therefore, effort control offers itself as a direct means to reduce the impact of the North Sea fisheries on the ecosystem. In addition, a range of technical measures is available that might solve specific problems, but more research is required to select the appropriate measures.

Because our knowledge of the ecosystem effects of fishing is still very limited, there is an urgent need to plan more research in this respect. The introduction of one or more closed areas for scientific purposes is expected to enhance considerably our understanding of the effect of fisheries as a structuring force on the ecosystem, particularly on the benthic community.

9 FOOD FOR THOUGHT

9.1 Changes in the Fish Assemblage Within Three Regions of the North Sea

A range of the types of analyses described in Section 7.2.2.1.2 were applied to the International Young Fish Survey data collected over the period 1972 to 1991 within three regions of the North Sea by Scottish fisheries research vessels (Figure 9.1.1).

Four diversity indices were used: the Shannon-Weiner index (H') and the reciprocal of Simpson's index ($1/D$), and two indices obtained by fitting the data to the log-normal (λ) and log-series (α) models. The equations for these indices, and methods for fitting the theoretical models, are given in Magurran (1991). In over 40% of samples, the log-series model provided an inadequate description of the observed data. Annual variation between the Shannon-Weiner index and the reciprocal of Simpson's indices was strongly correlated in all three areas ($P < 0.001$ in each case) and both suggested a significant trend for decreasing diversity moving northeast into the North Sea away from the northeast coast of the UK. The log-series index suggested that there was little difference in species diversity between boxes 1 and 2, but that species diversity in box 6 was significantly higher than in both offshore boxes (Figure 9.1.2a). Significant between-year variation was

only indicated by the log-series index, which suggested the possibility of a cyclical event (Figure 9.1.2b), however additional years of data are required to test this possibility.

The log of the number of fish belonging to 5 cm length classes was plotted (Figure 9.1.3a) and the gradient of the slope (regression coefficient) over the length range 20 to 65 cm, where the relationship was linear, was determined. The slope of the relationship observed in box 1 was shallower than those observed in boxes 2 and 6; the fish assemblage in box 1 contained a larger proportion of bigger fish. No difference was detected between boxes 2 and 6. No significant between-year effect was noted; however, examination of the annual variation in the slope gradient within each box (Figure 9.1.3b) suggested that the gradient was shallow in box 1 up to 1984, but between 1984 and 1986 it steepened, so that from 1987 onwards the fish assemblages in all three boxes were indistinguishable by this method.

The Bray-Curtis dissimilarity index (Bray and Curtis, 1957) was applied to the species abundance data and a cluster analysis was applied to the resulting similarity matrices. Multi-dimensional scaling plots obtained for each area suggested that the fish communities present from 1983 onwards contained similar species assemblage patterns. These later years were clustered closer together on the plots (Figure 9.1.4) and this was particularly apparent in box 6.

To conclude:

- Species diversity decreased further away from the British northeast coast and may have cycled between 1977 and 1991;
- The fish assemblage of the most offshore area had a larger percentage of bigger fish than either of the other two areas from 1972 to 1984, but not from 1986 onwards;
- Patterns of species abundance varied between 1972 and 1983, but became more stable thereafter.

The underlying reasons for these spatial and temporal differences in pattern remain unclear. However, they point out clear moments in time (1983-84) where changes in the structure of the fish community have occurred, and where boundaries between different communities (box 2 - box 1) may exist, thus indicating when and where structuring processes must have operated. Other data sets, for example hydrological data, must now be examined to see whether factors responsible for these changes in species assemblage patterns can be identified.

9.2 Productivity (P/B) Ratios (Life History Characteristics) and Benthos Mortality

Although the biomass (standing stock, expressed as weight per unit area or volume, B) of a population is an important measure of abundance, it is the production (P , the change of biomass per unit time) and productivity (P/B , or turn-over rate of biomass) which determine the potential for the population's mass growth and renewal. Predators or fisheries harvest, *sensu stricto*, production and productivity (both are dynamic rate measures) rather than biomass (a static measure). Because the production and productivity of a population determine its capacity for exploitation, recovery, and control, these factors are among the key terms one wishes to predict (Peters, 1983).

Allen (1971), working on the assumption of a stable age structure, asserted that the annual population P/B is equivalent to the annual instantaneous rate of total mortality (Z), provided that mortality is exponential and that individual growth can be approximated by the von Bertalanffy growth function (VBGF). Thus, given these provisions, *indirect estimates* of P/B can be extracted from the literature describing mortality. In this context, "turn-over" (P/B) is obviously highly correlated with mortality and body mass characteristics generally (Beverton and Holt, 1959; Hoenig, 1983). There is a general relationship between the annual P/B ratio and life history: this indicates that the P/B ratio increases with the number of generations produced per year (Waters, 1977), but decreases with the longevity (Zaika, 1970; Robertson, 1979) and age/size at maturity (Banse and Mosher, 1980; Heip *et al.*, 1982) and individual body weight per size class (Schwinghamer *et al.*, 1986). An obvious bias of P/B by demographic parameters (e.g., age structure, individual growth rate and survivorship) may lead to departures from the typical allometric exponent (-0.25) of size in equations describing specific rates (Peters, 1983). A study of population P , B , and P/B and mean individual weight (W) of marine macro-benthic invertebrates showed a highly significant dependence of P on B and W , whereas P/B depended only on W and not on B (Brey, 1990).

Despite the importance of these types of scalar productivity studies, a number of limitations exist regarding their applicability to determinations of the mortality effects of fishing activities on benthos. Although regression of the data on size at maturity (W_{sm}) shows that P/B declines as $W_{sm}^{-0.37}$ (Banse and Mosher, 1980; Peters, 1983), it is clear that the confidence limits of the log-log relationship are considerable, reflecting the large degree of inter-taxa variability governing both the slope and elevation of the regression. The main taxonomic groups in the benthos (e.g., molluscs, polychaetes, crustaceans) exhibit significantly different regression equations (Brey, 1990). As the faunal composition and dominance of benthic commu-

ities may vary markedly in time and space, their P/B ratios will vary accordingly. The P/B ratio of communities is dominated by the contribution of the smaller/younger size fractions (Calder, 1984; Reiss, 1989), thus emphasising that there must be a particularly intense differential removal of older/larger individuals before the community P/B will be changed, by either predators or fishing gear.

A promising approach for examining the effect of fishing mortality on the benthos is to determine the variability in P/B and associated life histories of relevant benthic species with respect to environmental parameters (e.g., ambient temperature, geographic and latitudinal gradients). Unfortunately, few direct calculations of P or P/B have been carried out for given species at different localities spanning a broad range of environmental characteristics. However, this approach has been used in examining variability in life history traits (including B, P, and P/B for cohort, age-group and population) for *P. borealis* at a single locality (Balsfjord, northern Norway) over a decade, as well as over the geographical range of the species (Hopkins *et al.*, 1990; Nilssen and Hopkins, 1991; Clarke *et al.*, 1992). The results, *inter alia*, underlined the susceptibility of the species towards the extremes of its range of distribution. Towards the southern and warmer end of the distribution, age-at-maturity at the lower physiological extreme coupled with semelparity resulted in a situation prone to heightened mortality rates. Towards the northern and colder end of the distribution, a high age-at-maturity requires a high degree of iteroparity to sustain the population, thus making such populations vulnerable to elevated mortalities. These basic considerations apply to poikilotherms in general (Cole, 1954; Stearns, 1976; Calder, 1984).

Alterations of a population's reproductive characteristics through differential exploitation or environmental change may reduce population fitness (Leggett and Carscadden, 1978). Although the mechanism is primarily steered through mortality, it should be envisaged as part of the realm of density dependent regulation. The potential difference in natural mortality rates between populations may lead to varied responses to specific exploitation rates. Differences in life cycles, must be taken into account when developing prudent management strategies (Stearns and Crandall, 1984).

9.3 Managing Fish Stocks while Considering Wildlife - Problems of Scale

The problems that have become apparent near Shetland (Section 7.2.3.1) in the past few years are instructive in showing the difficulties of managing fish stocks at the scale important to wildlife. The Shetland sandeel 'stock' and fishery are trivial in terms of total North Sea stock and catch, but in terms of the North Sea's seabirds it is

very important. Fisheries managers do not have the resources to collect information on a small scale. This meant that the stock was defined in terms of the area of the fishery in whole ICES Statistical Rectangles, rather than as a known biological unit. Recent evidence shows that the Shetland stock is not unitary. As a consequence, stock assessment for both fishery management and, potentially, to evaluate the stock's importance to wildlife is extremely difficult.

In the case of the Shetland stock, it was fortuitous that some statistics were collected from the fishery when the stock was increasing to a peak in the early 1980s. Without these statistics, it would have been very difficult to determine what had happened to cause the seabird breeding failures. On a wider scale, if we are to be sure that fisheries are not affecting wildlife, it is important that sufficient research be done on key fish stocks at a scale relevant to wildlife's usage of the same stocks.

Of the fisheries that may be in direct competition with seabirds, the industrial fisheries for sandeel, sprat and Norway pout are likely to be the most important. While accepting that the largest factor in determining the size of the stocks of these fish may be variation in recruitment due to environmental factors, it is alarming that the information on stock size and distribution is particularly poor. In the most recent report available from the ICES Industrial Fisheries Working Group, it was found that "in 1990 ... sampling of industrial landings in the North Sea decreased to an unacceptably low level. This decrease was particularly serious in the sampling for age composition." In the eastern North Sea, fewer than 50 samples were taken from industrial landings in 1990; it should be remembered that this fishery comprised around 50% of the fish biomass landed in that year (Section 3). If any sort of assessment is to be made of the potential impact of this very large and important fishery on seabirds, then not only must information be available on a relevant scale, it must first be collected!

There has been a great growth in information on which areas (and, to a certain extent, which fish/fish stocks) are important to birds in recent years. Some areas are relatively fixed, for instance those near major colonies. At other times of the year, seabirds can move further in search of food, but may survive better if certain fish stocks are large. If fisheries are to be managed while taking other wildlife into consideration, perhaps these known areas and stocks should be the first to be considered. Even if stocks are mostly controlled by variations in recruitment due to environmental factors, managers should be sure that fisheries are not exacerbating the frequency of recruitment failures, depleting stocks following such failures, or restricting the ability of the stock to recover.

10 RECOMMENDATIONS

The Study Group on Ecosystem Effects of Fishing Activities recommends that

Recommendation 1

the Study Group should work by correspondence during 1993, with a view to meet in 1994, in order to:

- a) analyze existing data on discards and offal to study temporal and spatial variations in the amounts that are produced by different fisheries and their utilization by and effects on different components of the system;
- b) evaluate methods of assessing impacts of groundfish fisheries on the benthic infauna and epifauna with particular reference to P/B-related approaches;
- c) analyze existing survey data in terms of appropriate summary parameters for species assemblages, with a view to initiating a study of changes in community structure;
- d) consider the attributes that would be appropriate to define indicator species for the evaluation of long-term impacts of fishing in order to initiate a review of information on a variety of marine species that meet these attributes.

Recommendation 2

the report on Ecosystem Effects of Fishing Activities in the North Sea (Editor, H. Gislason) should be published in the ICES Cooperative Research Report series.

Recommendation 3

ICES should support initiatives to establish areas of appropriate size that are closed for all fishing in order to monitor the response of benthic communities in heavily fished areas and allow process studies that could help to understand the impacts of fisheries.

Requests to other ICES Working Groups

The Benthos Ecology Working Group is requested to develop criteria by which to judge the significance of fishing and comparable anthropogenic influences on benthos, taking account of locality- and habitat-specific factors in order to permit the Study Group on Ecosystem Effects of Fishing Activities to identify appropriate indicator species for the evaluation of long-term impacts of fishing.

The Industrial Fisheries Working Group is requested to provide information on the industrial fish stocks, their

age structure, and their fisheries on an ICES Statistical Rectangle basis, in order to allow the Study Group on Ecosystem Effects of Fishing Activities to evaluate the quantitative effects of industrial fisheries on the ecosystem.

The Study Group on Seabird-Fish Interactions is requested to coordinate research on interactions between fisheries and birds in the North Sea and adjacent areas with special emphasis on:

- a) diet studies on species that scavenge on discards and offal;
- b) quantitative observations on rates of entanglement in different fishing gears;
- c) diet studies particularly outside the breeding season and away from the coast.

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Table 3.1.1 Percentage changes in numbers for those species of seabirds for which reliable comparative data exist on British North Sea coasts between 1969/1970 and 1985/1987 (Lloyd *et al.*, 1991). * = over 80% of North Sea population breeds on British North Sea coasts.

Fulmar	+92%*	Herring gull	-43%
Gannet	+96%*	Great black-backed gull	-2%
Cormorant	-11%*	Kittiwake	+21%*
Shag	-8%*	Sandwich tern	+54%
Arctic skua	+222%*	Roseate tern	-87%*
Great skua	+150%*	Common tern	-20%
Black-headed gull	-20%	Little tern	+74%*
Lesser black-backed gull	+102%*	Guillemot	+108%*

Table 3.3.2 Timetable of major events which have influenced the fishing effort in the North Sea.

DATE	EVENT	DEMERSAL ROUND FISH	DEMERSAL FLAT FISH	PELAGIC	INDUSTRIAL	DATE
1870		Smacks beam trawling				1870
		Steam trawling				
1880		Otter boards				1880
	Mechanical ice production					
1890						1890
				Steam drifters		
1900		Smacks sole fishing southern North Sea only			Interest in fish products as animal feed	1900
	ICES					
1910						1910
	WW 1 Some reduction in fishing	Half remaining smacks sunk				
	Russian Revolution Emergence of Eastern Bloc fishing fleet					
1920				Herring trawling		1920
		Bridles & Bobbins Anchor and Fly Seining				
					Fish meal production	

DATE	EVENT	DEMERSAL ROUND FISH	DEMERSAL FLAT FISH	PELAGIC	INDUSTRIAL	DATE
1930	Radio telephone Direction finders Echo sounders	Powered line and gill net haulers				1930
1940	WW 2 70 % reduction in catches	Smacks all gone		Torry kiln increased smoking		1940
1950	DECCA Echo sounders as fish finders Synthetic fibres Asdic	Shipboard freezing			Danish Bloden herring fishery	1950
					Danish Sandeel fishery	
1960					Norwegian Sandeel fishery	1960
1970	Icelandic Cod wars also in 1975/6					1970
1980						1980
1990						1990

Table 3.3.3 Fish stocks for which time series of fishing mortality data are available from ICES working groups.

Stocks	First year	Last year	Source
Cod	1963	1990	Anon. 1992a
Haddock	1960	1990	Anon. 1992a
Whiting	1960	1990	Anon. 1992a
Saithe	1960	1990	Anon. 1992a
Plaice	1947	1990	Anon. 1992b
Sole	1951	1990	Anon. 1992b
Herring	1947	1990	Anon. 1991c
Sprat	1967	1984	Anon. 1985a
Mackerel (N.S. Stock)	1969	1984	Anon. 1985b
Sandeel (North)	1972	1989	Anon. 1990a
Sandeel (South)	1971	1989	Anon. 1990a
Sandeel (Shetland)	1976	1991	Anon. 1991d
Norway pout	1974	1989	Anon. 1990a

Table 3.3.4 Total international fishing effort by gear, NSTF area and quarter in 1989 as estimated from ANOVA of STCF data on catch and effort by fleet (see text and Annex 2 for further details).

a. Beam Trawlers. Total International Effort. (Excluding Belgium.)

Total effort in units of net beam trawl > 300 HP days absent from port.

NSTF Area	Quarter	1	2	3	4	Total
1						
2a						
2b		2	348	246	100	695
3a				2	33	35
3b		1,058	519	1,513	1,680	4,770
4		6,093	7,197	7,853	5,308	26,451
5		4,327	2,731	1,094	1,398	9,550
6		59	129	265	158	611
7a		575	3,069	1,593	1,457	6,694
7b		3,524	3,449	4,448	4,686	16,107

b. Otter Trawlers. Total International Effort.

Total effort in units of Scottish otter trawler hours fishing.

NSTF Area	Quarter	1	2	3	4	Total
1		19,868	13,675	18,620	18,996	71,158
2a		17,553	22,055	21,511	9,742	70,860
2b		6,971	18,614	10,831	9,601	46,016
3a		10,583	16,812	13,629	25,958	66,982
3b		15,571	15,608	8,143	12,798	52,120
4		6,975	5,593	3,285	15,207	31,061
5		3,155	6,473	2,543	17,066	29,237
6		5,409	8,722	9,505	4,527	28,162
7a		3,820	6,823	11,235	4,001	25,879
7b		2,516	5,746	11,670	4,335	24,266

c. Pair Trawlers. Total International Effort.

Total effort in units of Scottish pair trawl demersal hours fishing.

NSTF Area	Quarter	1	2	3	4	Total
1		10,971	8,227	6,553	4,787	30,538
2a		8,307	32,425	5,493	2,560	48,785
2b		2,371	18,726	12,162	1,868	35,127
3a		2,367	10,513	17,745	7,795	38,420
3b		2,422	5,489	9,171	3,303	20,385
4		20,461	13,172	13,270	43,472	90,375
5		3,030	5,432	3,720	3,928	16,110
6		32,953	10,727	12,583	6,436	62,699
7a		9,099	15,683	15,525	12,456	52,763
7b		3,087	3,850	9,635	1,873	18,445

Table 3.3.4 (cont'd)

d. Industrial Otter Trawl. Total International Effort.

Total effort in units of Danish industrial trawl >100 GRT days absent from port.

NSTF Area	Quarter	1	2	3	4	Total
1		799	418	516	607	2,340
2a		3	175	19	1	198
2b		1,650	902	423	1,061	4,036
3a		14			10	24
3b		11	617	107		735
4		5	213	190	13	422
5		45	3,121	2,204	149	5,519
6		1,789	4,275	2,032	5,745	13,841
7a		1,607	7,622	1,074	556	10,859
7b		625	7,833	2,035	502	10,996

e. Industrial Pair Trawlers. Total International Effort.

Total effort in units of Danish industrial pair >100 GRT days absent from port.

NSTF Area	Quarter	1	2	3	4	Total
1		24	1	36	14	75
2a						
2b		36		49	1	86
3a			3			3
3b						
4						
5		27	15	148	36	226
6		389	524	2,009	467	3,390
7a		33	31	147	27	237
7b		55	1	11	16	83

f. Seines and Ringnets. Total International Effort.

Total effort in units of Scottish seine net hours fishing.

NSTF Area	Quarter	1	2	3	4	Total
1		46,085	31,073	58,262	44,202	179,622
2a		3,521	4,773	3,922	4,817	17,033
2b		4,570	25,955	19,980	13,686	64,191
3a		6,475	10,564	17,600	7,719	42,358
3b		209	540	7,208	101	8,059
4		151	443	140	74	808
5		1,012	6,684	8,057	4,193	19,948
6		4,516	21,758	3,210	4,813	34,297
7a		7,605	45,576	27,534	10,157	90,872
7b		5,626	8,682	13,639	5,197	33,144

Table 3.3.4 (cont'd)

g. Pelagic Trawlers. Total International Effort.

Total effort in units of French HSF, pelagic trawlers hours.

NSTF Area	Quarter	1	2	3	4	Total
1		0	0	16,608	12,613	29,220
2a		0	0	15,859	1,124	16,983
2b		0	0	5,120	0	5,120
3a		0	1,499	3,746	0	5,245
3b		21,689	0	10,990	37,990	70,668
4		31,083	0	15,882	86,427	133,392
5		0	0	0	0	0
6		0	0	0	0	0
7a		0	0	624	20,725	21,350
7b		0	0	0	77,096	77,096

h. Purse Seines. Total International Effort.

Total effort in units of Danish purse seiner days absent from port.

NSTF Area	Quarter	1	2	3	4	Total
1		628	1,792	11,065	1,003	14,489
2a		0	26	1,129	323	1,478
2b		59	11	151	195	416
3a		0	0	92	7	99
3b		0	0	13	0	13
4		0	0	0	0	0
5		0	0	0	0	0
6		327	548	1,005	1,797	3,677
7a		2	1	153	110	266
7b		0	0	2	0	2

i. Gill Net. Total International Effort.

Total effort in units of Danish gill net days absent from port.

NSTF Area	Quarter	1	2	3	4	Total
1		0	7	12	0	19
2a		4	134	6	0	144
2b		4	97	52	5	158
3a		139	40	61	156	396
3b		207	354	129	250	941
4		1,254	1,557	167	311	3,289
5		696	1,809	750	735	3,990
6		198	409	269	121	997
7a		841	1,081	969	252	3,143
7b		278	1,048	492	133	1,952

Table 4.2.1 Types of fishing gear, parts in contact with the ground, penetration depths and areas fished per 100 ship hours. (Sources: Anon., 1988c; Anon., 1990c; Margetts and Bridger, 1971; Van der Hak and Blom, 1990.)

Type of gear	Gear parts in contact with ground		Typical width of gear parts (m per ship)	Towing speed (knots)	Penetration depth of gear parts (cm)		Estimated area fished (km ² /100 h)	Fishing area
	Minor	Major			Soft ground	Hard ground		
Beam trawl (flatfish):								
a) outside 12 mile zone	-	Trawl shoes + Tickler or Chain mats	12.0 x 2	6	>6	6	31.1	Extensive
b) inside 12 mile zone	-		4.0 x 2	5	?	?	8.9	Coastal
Shrimp beam trawl (Crangon)	Ground rope with rollers	Shoes	0.20 x 4	4-5	?	?	0.7	Estuarine/ Coastal
Otter trawl	-	2 otter doors	1.5 x 2	3-4	8-10	<5	2.0	Extensive/ Coastal
	Ground rope	-	30	3-4	?	?	22.2	
Industrial single trawl	-	2 otter doors	1.5 x 2	3.5	8-10	<5	1.9	Extensive
	Ground rope	-	25	3.5	?	?	16.2	
Industrial pair trawl	Ground rope	-	25	3.5	?	?	16.2	Extensive
Pair trawling:								
a) Pelagic	-	-	-	3	-	-	-	Extensive
b) Demersal	Ground rope	-	40	3	?	?	26	Extensive
Seine	Ground rope	-	-	-	?	-	0.05	Extensive
Mussel dredge	-	Blade + belly	1.75 x 4	2	5-25	-	2.6	Estuarine/ Coastal (Wadden Sea)
Cockle dredge	-	Suction head	1.0 x 2	2	≥5	5	0.7	Estuarine/ Coastal
Scallop dredge								
a) English (Newhaven)	-	Tooth bar + belly	0.75 x 16	3	3-4	2-3 ?	6.6	Coastal
b) French	-		2 x 5	3-4	<10	NA	7.2	
French clam dredge		Blade + belly	0.7 x 2	3-4	<15	NA	0.8	Coastal

Table 4.2.2 Factors used to estimate total areas swept and impacted by various gear types.

Factor	Beam trawl	Otter trawl	Industrial pair trawl	Industrial single trawl	Pair trawl	Seine
Area impacted (km ² /100 h)	31.1	2		1.9		
Area swept (km ² /100 h)	31.1	22.2	16.2	16.2	26	0.05
Conv. to Eff. in hours	16	1	16	16	1	1

Table 4.2.3 Total areas swept in 1989 (km²/year) by those parts of the gear which have a major impact on the sea bed. (See Tables 4.2.1 and 4.2.2.)

NSTF area	Beam trawl	Otter trawl	Industrial single trawl	Total
1		1,423	711	2,134
2a		1,417	60	1,477
2b	3,461	920	1,227	5,608
3a	176	1,340	7	1,523
3b	23,735	1,042	224	25,001
4	131,619	621	128	132,368
5	47,520	585	1,678	49,783
6	3,040	563	4,208	7,811
7a	33,311	518	3,301	37,130
7b	80,147	485	3,343	83,975
Total	323,009	8,914	14,887	346,811

Table 4.2.4 Total areas swept in 1989 (km²/year) by towed fishing gear in contact with the seabed. (See Tables 4.2.1 and 4.2.2.)

NSTF Area	Beam trawl	Otter trawl	Industrial pair trawl	Industrial single trawl	Pair trawl	Seine	Total
1		15,797	195	6,077	7,940	90	30,099
2a		15,731		514	12,684	9	28,938
2b	3,461	10,216	224	10,481	9,133	32	33,547
3a	176	14,870	7	62	9,989	21	25,125
3b	23,735	11,571		1,909	5,300	4	42,519
4	131,619	6,896		1,095	23,498	0	163,108
5	47,520	6,491	587	14,331	4,189	10	73,128
6	3,040	6,252	8,803	35,943	16,302	17	70,357
7a	33,311	5,745	616	28,199	13,718	45	81,634
7b	80,147	5,387	216	28,554	4,796	17	119,117
Total	323,009	98,956	10,648	127,165	107,549	245	667,572

Table 4.2.5 Estimates of area swept in 1989 by beam trawl and otter trawl (only boards) in 1,000 km² and in percentage of the size of the corresponding NSTF area.

NSTF Sub-area	Surface of NSTF area 1,000 km ²	Swept area beam trawl 1,000 km ²	Percentage of area swept by beam trawl	Swept area otter trawl boards 1,000 km ²	Percentage of area swept by otter trawl boards
1	92			2.1	2.3
2a	68			1.5	2.2
2b	55	3.5	6.3	2.1	3.9
3a	55	0.2	0.3	1.3	2.5
3b	44	23.7	53.9	1.3	2.9
4	41	131.6	321.0	0.7	1.8
5	31	47.5	153.3	2.3	7.3
6	55	3.0	5.5	4.8	8.7
7a	99	33.3	33.6	3.8	3.9
7b	71	80.1	112.9	3.8	5.4

Table 4.4.1 Types of mollusc dredge fisheries and their initial biological and physical effects (combined).
 (*slight ****severe; arbitrary scale assessed from literature).

Dredge fishery	Substrates				Countries	References
	Rocks/ stones	Gravels/ shells	Sands	Muds		
(A) Non-hydraulic						
scallop (<i>Pecten maximus</i>)	****	**			UK, F	a
i) English		**	*		F	b
ii) French	**	*			UK, F	c
queen scallop (<i>Aequipecten opercularis</i>)		****	**		F	d
clams (<i>Spisula, Venerupis, Glycymeris</i>)		**			UK, N	e
oyster (<i>Ostrea edulis</i>)				*	UK, N, G, D, F	e
mussels (<i>Mytilus edulis</i>)				**	UK	e
clam (<i>Mercenaria mercenaria</i>)						
(B) Hydraulic						
cockle (<i>Cerastoderma edule</i>)			*	***	UK, N	f
clam (<i>M. mercenaria</i>)				***	UK	g
razor clam (<i>Ensis</i> sp.)			**		UK (exploratory)	h

Countries: D = Denmark, F = France, G = Germany, N = Netherlands, UK = British Isles.

References: a = MAFF (unpublished); Chapman *et al.*, 1977; Fowler, 1989; b = Dupouy, 1982; Berthou and Hamon (in prep.); c = Brown, 1989; d = Medcof and Caddy, 1971; Berthou and Hamon (in prep.); e = MAFF (unpublished); f = MAFF (unpublished); Fowler, 1989; Godcharles, 1971; Peterson *et al.*, 1987; h = Hall *et al.*, 1990b.

Table 4.4.2 Mean fishing mortality rates (F) and percentage of stock caught during the year over the period 1986-1990 for the main species exploited in the North Sea. The subscripts refer to the range of age groups to which the estimate applies. O refers to the O-group in their first year of life, 1 to the second year of life, etc. All estimates based on single species assessments carried out in 1991.

Species	Stock area	Juveniles		Main exploited age groups	
		F	% caught	F	% caught
Cod	IV	F1=0.18	11.2	F2-8=0.88	52.1
Haddock	IV	F1=0.11	5.2	F2-6=1.01	54.2
Whiting	IV	F1=0.21	12.3	F2-6=1.01	36.5
Saithe	IV	F1=0.03	2.2	F2-6=0.60	30.4
Sole	IV	F1=0.003	0.3	F2-8=0.49	32.9
Plaice	IV	F1=0.002	0.2	F2-10=0.53	27.4
Herring	North Sea stock in IV, VIIId, IIIa	F1=0.37	20.1	F2-6=0.45	28.8
Sandeel	Shetland	F0=0.07 ¹	4.8	F1-3=0.15	7.1
Sandeel ²	IV N of 56°30'N (excl. Shetland)	F0=0.03 ¹	2.2	F1-3=1.38	37.9
Sandeel ²	IV S of 56°30'N	F0=0.001 ¹	0.1	F1-3=0.42	16.6
Norway Pout ²	IV	F0=0.06 ¹	3.3	F1-2=0.85	17.0

¹Estimates for the second half of the year.

²Mean values over 1986-1989.

Table 4.4.3 Increasing importance of fishing-gear-related auk mortality shown in ringing recoveries. Recoveries are grouped into four classes: FD (found dead), Oil (reported as oiled), Shot (shot or otherwise hunted), Net (in netting or other fishing gear); and two periods: Historic (all recoveries to 1970) and Recent (July 1987 to June 1989). (From: Mead, 1989.)

Species		FD	Oil	Shot	Net	Total
Guillemot	Historic	226	142	112	24	504
	%	45	28	22	5	
	Recent	283	100	35	244	662
	%	43	15	5	37	
Razorbill	Historic	307	109	155	40	611
	%	50	18	25	7	
	Recent	83	22	2	37	144
	%	58	15	1	26	

Table 4.4.4 Catches of harbour porpoises taken in gill nets and other gears from some studies conducted in the North Sea, Skagerrak and Kattegat.

Country	Period	Gill nets	Other	Unknown	Total	% gill nets	Source
Sweden	6/88 - 12/90	151	27	0	178	85	Lindstedt (1991)
Germany	1/87 - 12/90	38	2	1	41	93	Benke <i>et al.</i> (1991)
Denmark	8/80 - 2/81	111	38	0	149	74	Clausen and Andersen (1988); Kinze (1990)
	1/86 - 12/89	147	5	0	152	97	
Eastcoast of Scotland	1960 - 1965	40	12	0	52	77	Northridge (1988)
	1966 - 1986	67	17	0	84	80	
Shetland	1989 - 1990	0	10	12	22	0	SMRU (unpublished)

Table 4.5.1 Range of survival rates reported for fish escaping from various gears.

Gear	Species	Percentage survival	Length of experiments
Otter trawl	Cod, haddock	67-94%	3-4 weeks
Beam trawl	Sole	60-100%	24-96 hours
	Plaice	85%	24-96 hours
	Dab	56-67%	24-96 hours
Pelagic trawl	Herring	60-97%	1-12 days
Purse-seine	Mackerel	10% (slipped fish)	48 hours

Table 4.5.2 Estimated quantities of haddock and whiting discarded in the North Sea, 1971-1990 (Anon., 1992a).

Year	Haddock		Whiting	
	Number (millions)	Weight ('000 tonnes)	Number (millions)	Weight ('000 tonnes)
1971	1,282	177	458	63
1972	760	128	398	67
1973	660	115	659	110
1974	1,091	167	477	85
1975	1,862	260	699	135
1976	788	154	641	136
1977	226	44	547	163
1978	418	77	240	35
1979	286	42	640	77
1980	541	95	466	76
1981	298	60	210	35
1982	181	41	168	26
1983	389	66	360	48
1984	412	75	317	39
1985	458	86	226	28
1986	308	52	572	78
1987	334	59	408	53
1988	362	62	227	28
1989	111	26	275	35
1990	192	33	524	54

Table 4.5.3 Estimates of the percentage by weight of fish and other material discarded in some of the main fisheries in 1987-1989.

Area	Gear	Percentage of fish captured that are discarded			Additional percentage of other material ¹
		Commercial species	Non-commercial species	Total	
Western and Northwestern North Sea	Trawl	45	7	52	n/a
	Seine	49	6	55	n/a
Eastern North Sea	Trawl + seine	12	15	27	9
Southern North Sea	Beam trawl	18	38	56	58

¹ Quantity expressed as % of catch of fish and benthos combined; includes benthos and inorganic matter.

Table 5.2 Foodborne diseases caused by marine microbial toxins (after Taylor, 1988, and Ackefors, 1990).

Disease	Seafoods implied	Toxin-producing microorganism
Bacteria-associated	Scombroid fish	<i>Morganella morganii</i>
Scombroid fish poisoning	Tuna, mackerel jack, bluefish, herring, etc.	<i>Klebsiella pneumoniae</i> and other enteric bacteria
Algae associated	Reef fishes including barracuda, grouper, snapper, and sea basses	Dinoflagellates: <i>Gambierdiscus toxicus</i> , <i>Prorocentrum concavum</i> , <i>P. mexicana</i>
Paralytic shellfish poisoning	Bivalves, mainly mussels, clams, cockles, scallops	Dinoflagellates: <i>Gonyaulax catenella</i> , <i>G. tamarensis</i>
Neurotoxic shellfish poisoning	Bivalves, mainly mussels, clams, cockles, scallops	Dinoflagellates: <i>Ptychodiscus brevis</i> , ("red tide")
Diarrhetic shellfish poisoning	Bivalves, mainly mussels, clams, cockles, scallops	Dinoflagellates: <i>Dinophysis fortii</i> , <i>D. acuminata</i>

Table 5.5 Some physico-chemical properties of sewage sludge. (Unless otherwise specified, these are weighted averages of all UK sewage sludges disposed of to sea in 1987; source: MAFF, unpublished data.)

Constituents	Dry	Wet
Dry solids (%)	-	3.2
Organic matter (%)	64.2	2.0
Organic carbon (%)*	28.2	0.9
Total nitrogen (%)+	5.6	0.2
Total phosphorus (%)+	1.5	0.1
Mercury (ppm)	4.8	0.2
Cadmium (ppm)	13.6	0.4
Chromium (ppm)	547.4	17.3
Copper (ppm)	601.5	19.0
Nickel (ppm)	86.6	2.7
Lead (ppm)	662.0	20.9
Zinc (ppm)	1,816.7	57.4

*Assuming that 44% of organic matter is organic carbon.
+Not available for all arisings.

Table 6.2 Physical disturbance of the North Sea seabed by various anthropogenic activities. The data refer to 1986, except for fishing where the area equals the area swept by major ground-disturbing parts in 1989 (from Table 4.2.3). IOE: Institute of Offshore Engineering, Heriot Watt University, Edinburgh. Modified from Anon. (1992g).

Source	Area	No./amount	Reference/Source
Fishing	346,811 km ² /yr		
Aggregate	180 km ² /yr	30 x 10 ⁶ t	IOE calculation
Dredging disposal	72 km ² /yr	72 x 10 ⁶ t	Calculated from OSCOM 13
Waste disposal	5.5 km ² /yr	5.5 x 10 ⁶ t	Calculated from OSCOM 13
Sludge disposal	5.5 km ² /yr	5.5 x 10 ⁶ t	Calculated from OSCOM 13
Platforms	313 km ²	399	IOE calculation
Wellheads	300 km ²	382	IOE calculation
Pipelines	8,374 km ²	8,374 km	IOE calculation
Cables	7,322 km ²	7,322 km	IOE calculation
Wrecks	284 km ²	7,100	IOE estimate
Cuttings	0.5 km ² /yr	593,741 t	IOE calculation

Table 6.3.1 Assessment of risk of local extinctions through various natural and anthropogenic factors (1 = low; 4 = high). Based on literature review (Rees and Dare, Working Document 7).

Species	Burial/ wash-out	Temperature	Predation	Plankton blooms/ eutrophication	Trawl/dredge effects	Pollution	Other (excl. fishing)
POLYCHAETA							
<i>Pectinaria koreni</i>	4	2	3	2	3	1	1
<i>Sabellaria spinulosa</i>	1	1	2	1	4	2	1
CRUSTACEA							
<i>Cancer pagurus</i>	1	2	1	2	2	2	2
MOLLUSCA							
<i>Abra alba</i>	4	2	3	3	2	1	1
<i>Artica islandica</i>	1	1	1	1	3	2	1
<i>Pecten maximus</i>	1	2	2	1	2	2	1
ECHINODERMATA							
<i>Asterias rubens</i>	2	1	2	2	1	1	1
<i>Echinocardium cordatum</i>	3	2	1	4	3	2	1
<i>Echinus esculentus</i>	2	1	1	3	2	2	2

Table 6.3.2 Estimates of the proportion of the annual production of the main exploited fish species in the North Sea which is harvested by fisheries (Y/P), eaten by 'MSVPA' predators (cod, whiting, saithe, mackerel, and haddock) (Pr/P) and removed by other sources of death (M1/P), as well as production biomass ratios (P/B) and total annual production. Mean values for the years 1985 to 1989 from MSVPA (Anon., 1991e).

	Y/P	Pr/P	M1/P	P/B	Annual production (t x 10 ³)
Cod	0.74	0.03	0.23	1.24	238
Whiting	0.39	0.20	0.40	1.11	392
Saithe	0.67	0	0.33	0.63	271
Mackerel	0.59	0	0.40	0.46	35
Haddock	0.37	0.24	0.39	1.51	578
Herring	0.54	0.14	0.33	0.68	1,229
Sprat	0.16	0.34	0.50	1.22	331
Norway pout	0.11	0.52	0.37	2.05	1,411
Sandeel	0.27	0.28	0.44	1.09	2,776
Plaice	0.77	0	0.23	0.44	218
Sole	0.82	0	0.17	0.57	25
Total					7,504

Y = catch.

P = production.

Pr = quantity eaten by 'MSVPA' predators.

M1 = quantity killed by other sources of mortality including other predators.

B = average biomass during the year.

Table 6.3.3 Seabird recoveries (%) according to finding circumstances in Britain and Ireland, 1967-1978 (from Mead and O'Connor, 1980).

Species	Oiled	Net	Total found
Fulmar	3	6	163
Manx shearwater	3	2	580
Shag	5	10	2,884
Gannet	10	6	381
Eider	17	4	969
Kittiwake	5	3	333
Guillemot	30	9	311
Razorbill	32	13	315
Black guillemot	13	0	24
Puffin	13	2	299

Table 6.4.1 Estimated quantity of fish discarded from whitefish and *Nephrops* boats in the North Sea in 1982 (ICES) together with the numbers of gulls that could be sustained throughout the year by this food (Furness *et al.*, 1988).

ICES fishing area	Tonnes of whitefish discarded		Number of gulls supported
	Whitefish boats	<i>Nephrops</i> boats	
IVa	41,000	3,000	310,000
IVb	30,000	8,800	277,000
IVc	5,300	10	38,000
Total	76,000	11,800	625,000

Table 6.4.2 Estimated tonnes of offal discarded in the North Sea in 1989, by NSTF areas and total numbers of fulmars that could potentially be supported throughout the year. These are compiled from figures presented in Section 3.1. 6.5% of the flatfish (beam trawl fleet) catch (Furness *et al.*, 1988) and 12.5% of the catch by otter trawls, pair trawls and Danish seines/ringnets is assumed to be offal.

NSTF area	Beam	Otter trawl etc.	Total	Total fulmars
1	0	14,658	14,658	227,205
2a	0	10,537	10,537	163,322
2b	165	6,624	6,789	105,225
3a	2	5,777	5,779	89,569
3b	605	2,774	3,379	52,382
4	2,938	2,839	5,777	89,539
5	644	2,655	3,299	51,139
6	128	5,270	5,398	83,667
7a	1,300	6,086	7,385	114,474
7b	3,424	2,732	6,157	95,426
Total	9,206	59,952	69,158	1,071,948

Table 7.2.2.1 Total consumption (TC) in tonnes and predation relative to average prey biomass (P/P; %) by individual predators, based on MSFOR baseline run. From Anon. 1989b.

Prey		Predators					Total
		Cod	Whiting	Saithe	Mackerel	Haddock	
Cod	TC	12,433	1,226	987	263	-	14,909
	P/P	4.6 ¹	0.5	0.4	0.1		5.5
Whiting	TC	40,574	43,842	6,025	-	209	90,650
	P/P	10.3	11.1	1.5		0.1	22.9
Saithe	TC	-	-	-	-	-	-
	P/P						
Mackerel	TC	-	-	-	-	-	-
	P/P						
Haddock	TC	46,617	44,958	97,164	-	817	189,556
	P/P	9.4	9.1	19.6		0.2	38.3
Herring	TC	35,367	51,858	15,562	14,623	84	117,503
	P/P	3.4	5.0	1.5	1.4	+	11.4
Sprat	TC	24,501	187,077	18,334	17,192	2,595	249,699
	P/P	3.5	26.5	2.6	2.4	0.4	35.4
Norway pout	TC	63,623	115,772	805,026	17,319	56,316	1,058,057
	P/P	5.1	9.3	64.7	1.4	4.5	85.0
Sandeel	TC	106,003	349,104	83,100	248,015	121,183	907,405
	P/P	3.9	12.8	3.0	9.1	4.4	33.3
Total	TC	329,118	793,837	1,026,198	297,421	181,205	2,627,778
	P/P	4.8	11.6	14.9	4.3	2.6	38.2

¹Predation/prey biomass as percentage.

Table 7.2.2.2 Percentage change in long-term equilibrium biomass, spawning stock biomass (SSB) and catch predicted by the MSFOR following a 30% reduction of the fishing mortality generated by the North Sea roundfish fishery. Recruitment kept constant at average 1974-1988 level.

Species	Total Biomass	SSB	Catch
Cod	21	61	-8
Whiting	2	3	-11
Saithe	13	34	3
Mackerel	0	0	0
Haddock	-8	4	-24
Herring	-5	-6	-7
Sprat	0	0	0
Norway pout	-8	-12	-21
Sandeel	0	0	0
Plaice	0	0	0
Sole	0	0	0
Total	-1	0	-7

Table 7.2.2.3 Distribution of percentage long-term changes in spawning stock biomass subject to changes in levels of recruitment of +/- 50%. Status quo compared to 30% reduction in the fishing mortality generated by the North Sea roundfish fleet. Results from 512 comparisons.

% Change	Species										
	Cod	Whiting	Saithe	Mackerel	Haddock	Herring	Sprat	N. Pout	Sandeel	Plaice	Sole
80 - 90	0	0	0	0	0	0	0	0	0	0	0
70 - 80	24.0	0	0	0	0	0	0	0	0	0	0
60 - 70	33.8	0	0	0	0	0	0	0	0	0	0
50 - 60	36.1	0	0	0	0	0	0	0	0	0	0
40 - 50	6.1	0	0	0	0	0	0	0	0	0	0
30 - 40	0	0	100.0	0	0	0	0	0	0	0	0
20 - 30	0	0	0	0	16.0	0	0	0	0	0	0
10 - 20	0	0	0	0	27.5	0	0	0	0	0	0
.01 - 10	0	100.0	0	0	13.7	0	21.5	0	30.1	0	0
-.01 - .01	0	0	0	100.0	0	0	1.6	0	1.0	100.0	100.0
-10 --.01	0	0	0	0	17.4	94.1	77.0	48.8	68.9	0	0
-20 - -10	0	0	0	0	3.9	5.9	0	51.2	0	0	0
-30 - -20	0	0	0	0	19.7	0	0	0	0	0	0
-40 - -50	0	0	0	0	1.8	0	0	0	0	0	0
-50 - -60	0	0	0	0	0	0	0	0	0	0	0
-60 - -70	0	0	0	0	0	0	0	0	0	0	0
-70 - -80	0	0	0	0	0	0	0	0	0	0	0
-80 - -90	0	0	0	0	0	0	0	0	0	0	0
Sum	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0

Table 7.2.2.4 Percentage change in long-term equilibrium biomass, spawning stock biomass (SSB) and catch predicted by the MSFOR following a 10% reduction in the level of fishing mortality in the North Sea. Recruitment kept constant at average 1974-1988 level.

Species	Total Biomass	SSB	Catch
Cod	7	20	-3
Whiting	3	4	-5
Saithe	9	24	2
Mackerel	7	16	-1
Haddock	-5	-2	-13
Herring	5	7	-2
Sprat	3	5	-6
Norway pout	-3	-5	-17
Sandeel	4	7	-4
Plaice	6	10	4
Sole	9	13	2
Total	2	5	-1

Table 7.2.3.1 Numbers of breeding seabirds in Shetland, 1985-1987. Counts are of pairs except where otherwise indicated (derived from Lloyd *et al.*, 1991).

Fulmar	236,000
Manx shearwater	present
Storm petrel	present
Leach's petrel	present
Gannet	17,000
Cormorant	400
Shag	6,500
Arctic skua	1,900
Great skua	5,600
Black-headed gull	600
Common gull	2,500
Lesser black-backed gull	500
Herring gull	5,000
Great black-backed gull	3,000
Kittiwake	50,000
Sandwich tern	4
Common tern	1,000
Arctic tern	31,800
Guillemot	163,000
Razorbill	14,000
Black guillemot	12,000
Puffin	182,000

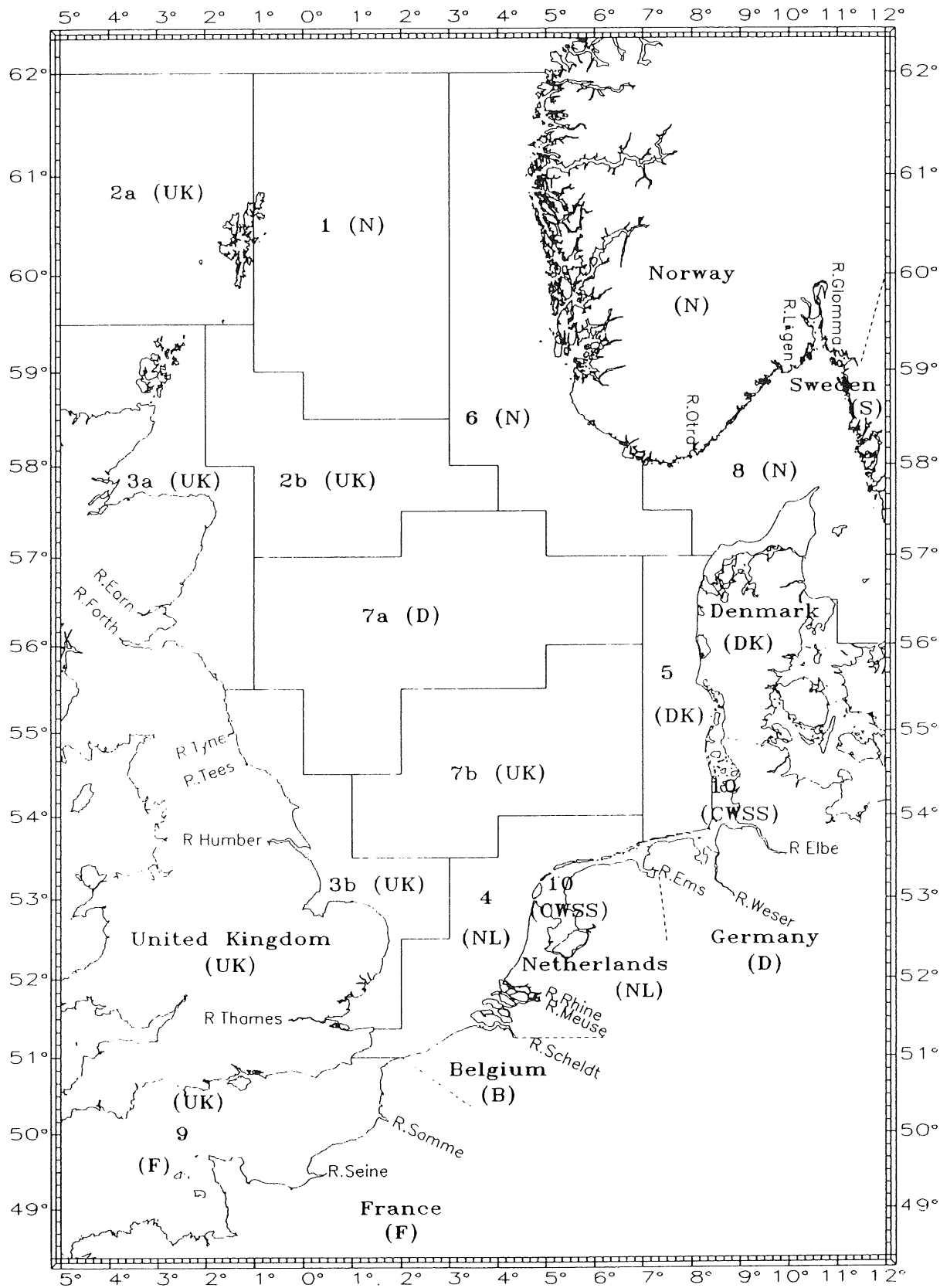


Figure 1 Sub-regions of the North Sea adopted by the North Sea Task Force.

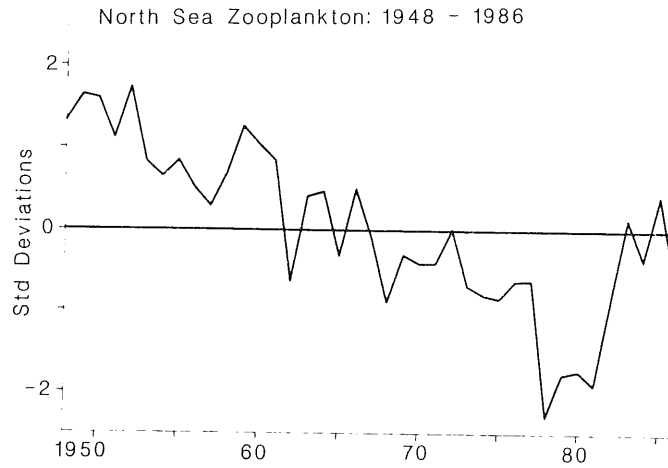


Figure 3.1.1 The dominant pattern of annual fluctuations in the abundance of North Sea zooplankton (from Franz *et al.*, 1991).

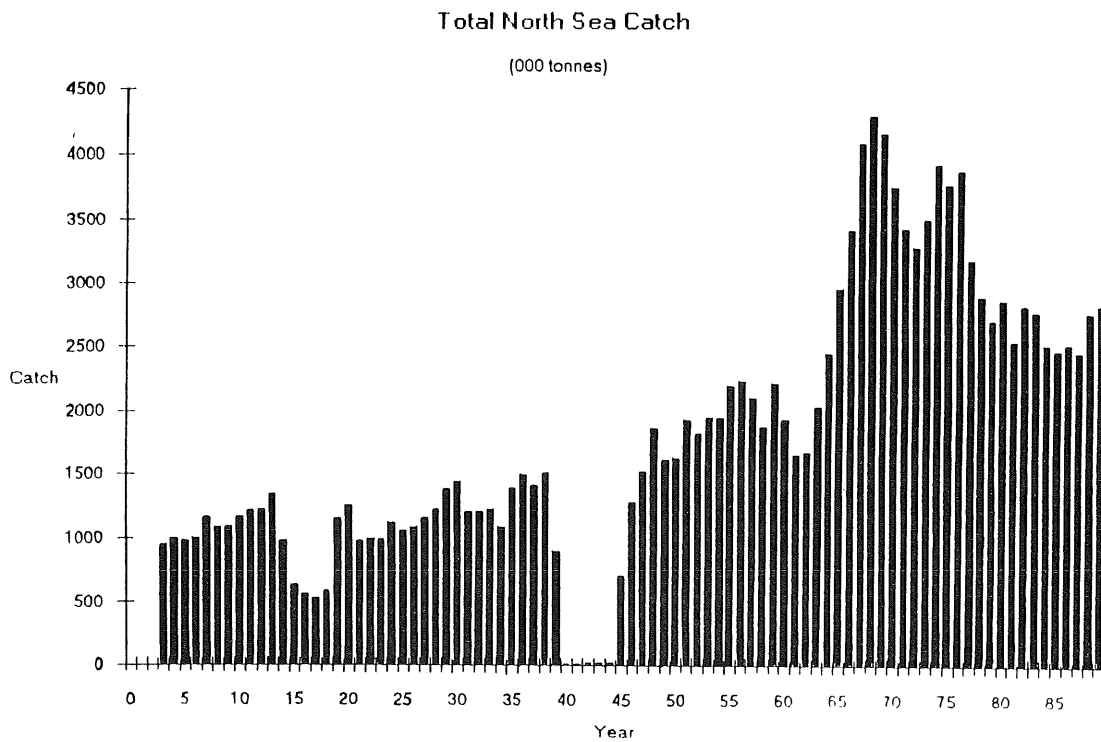


Figure 3.1.2.a Total international catch from the North Sea (ICES Sub-area IV). Data from Bull. Stat. and ICES Assessment Working Group Reports.

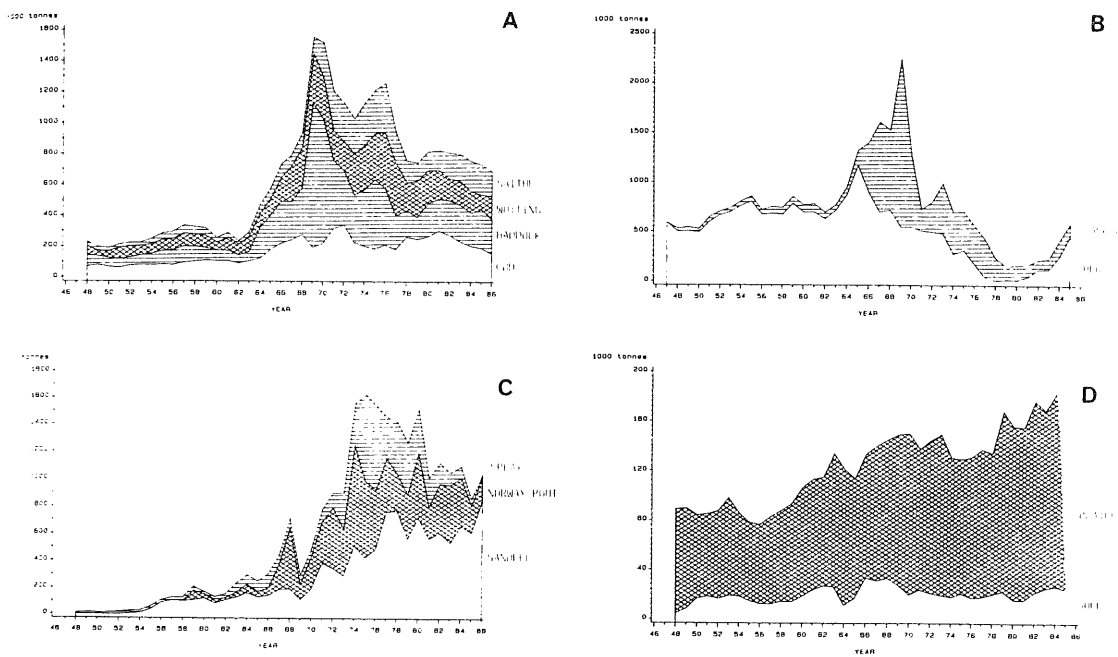
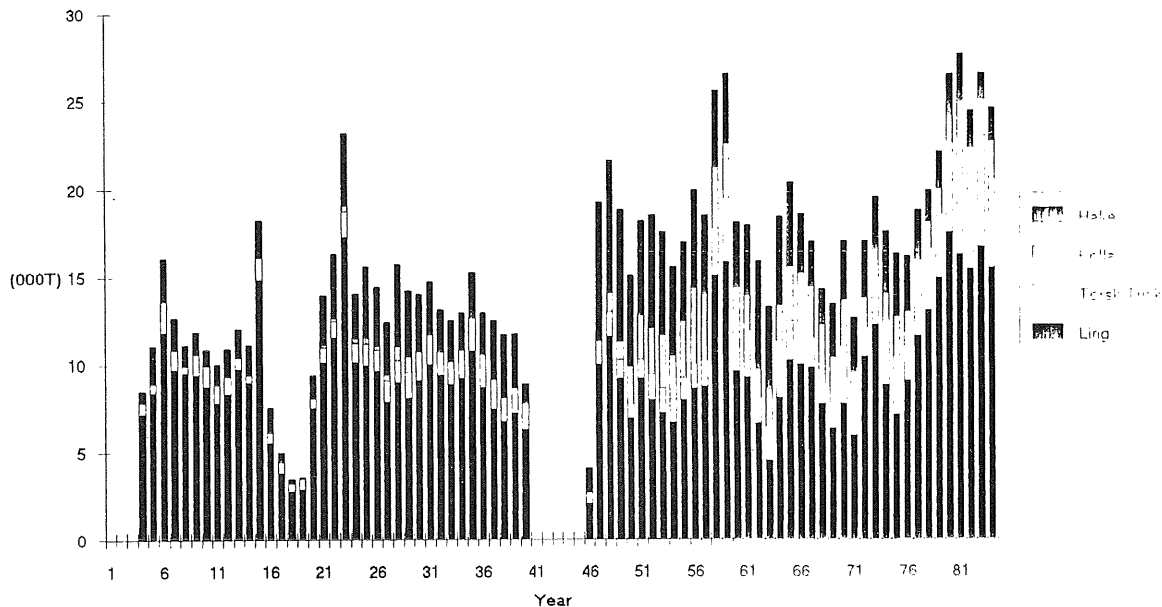
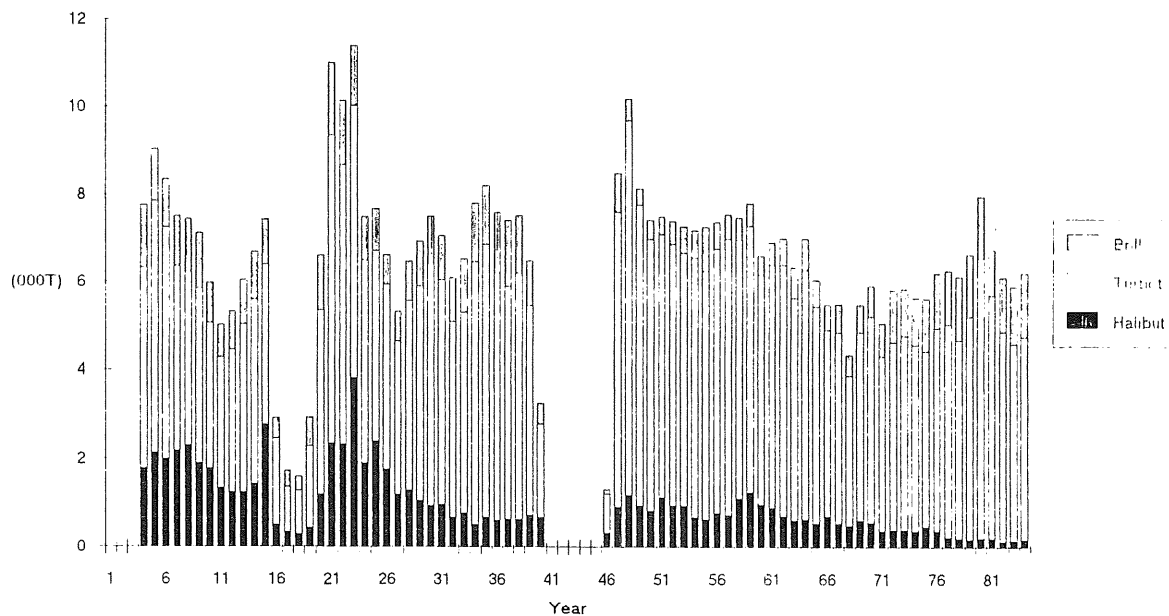


Figure 3.1.2.b Total international catch by groups of species. A: Gadoids. B: Pelagic species. C: Industrial species. D: Flatfish (from Daan *et al.*, 1990).

Trends in catches of Less Common Large Gadoids



Trends in catches of Large Flatfish in the North Sea



Trends in Elasmobranch catches

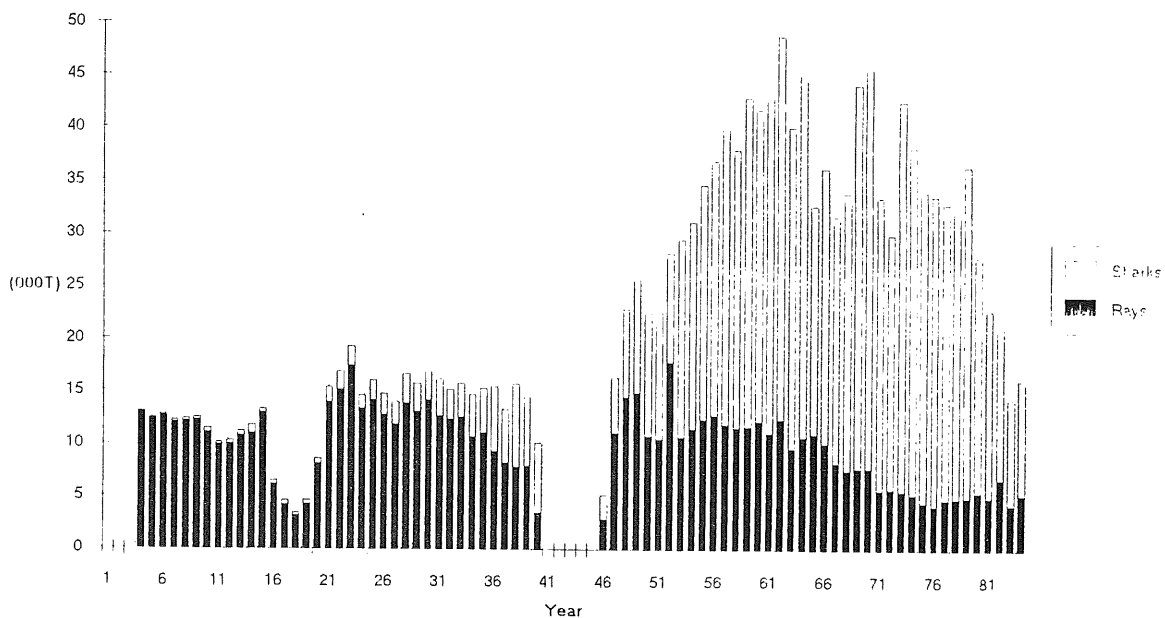


Figure 3.1.3 Reported landings of selected by-catch species of commercial interest (Bull. Stat.)

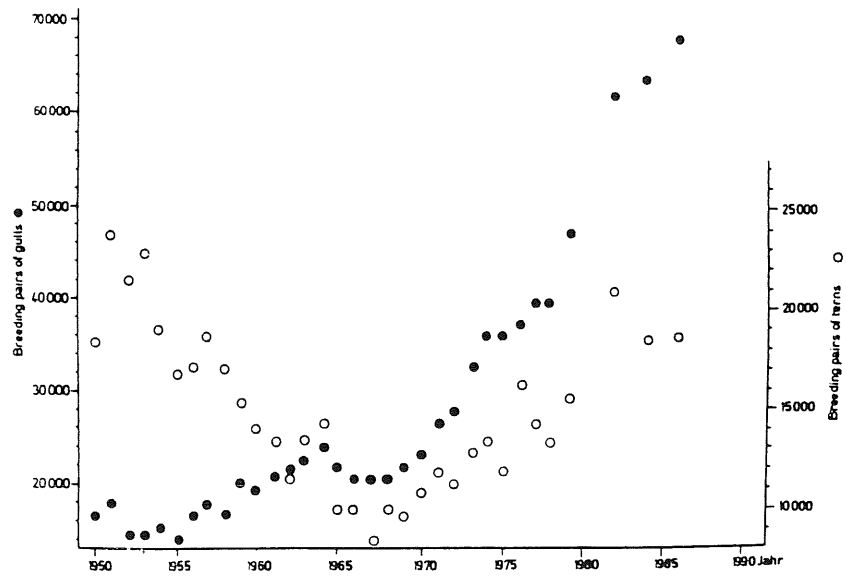


Figure 3.1.4 Trends in numbers of gulls (from Dunnet *et al.*, 1990).

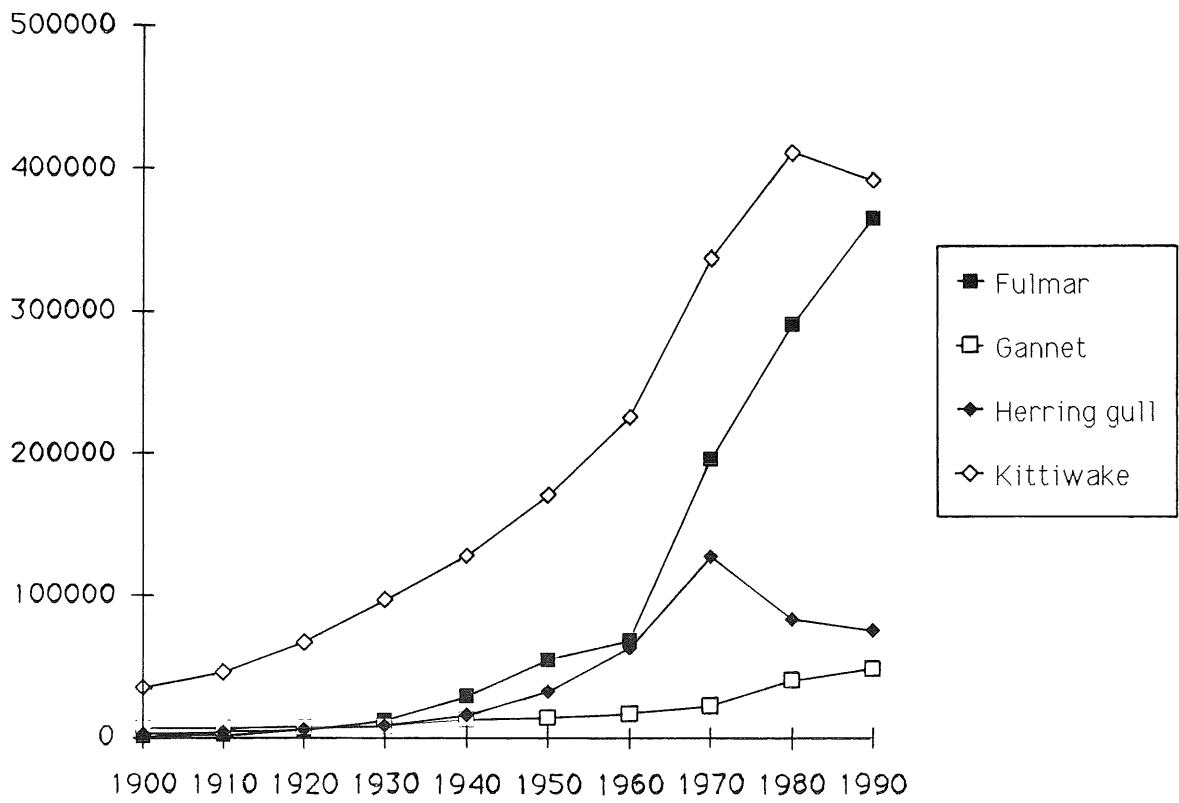


Figure 3.1.5 Trends in numbers of selected sea bird populations around the North Sea (from Furness (in prep.)).

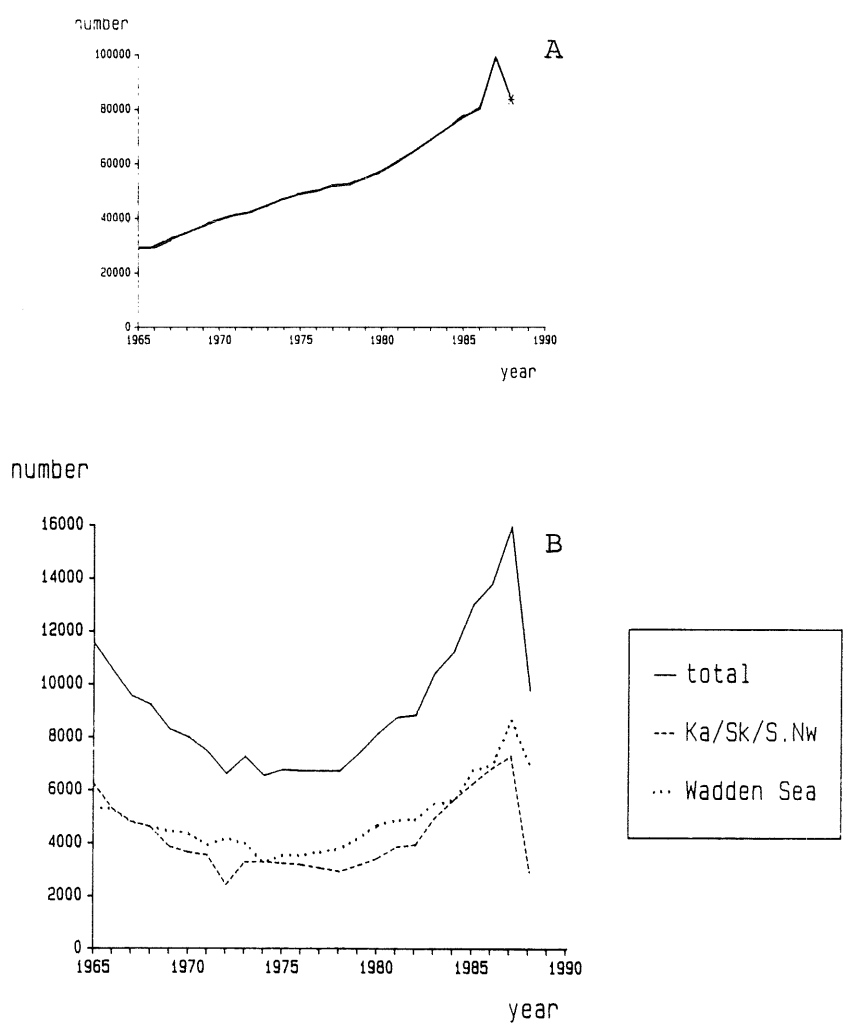


Figure 3.1.6 Long-term trends in seal populations. A: Number of grey seals in Great Britain. B: Number of common seals in the North Sea, excluding UK (from Reynders and Lankester, 1990).

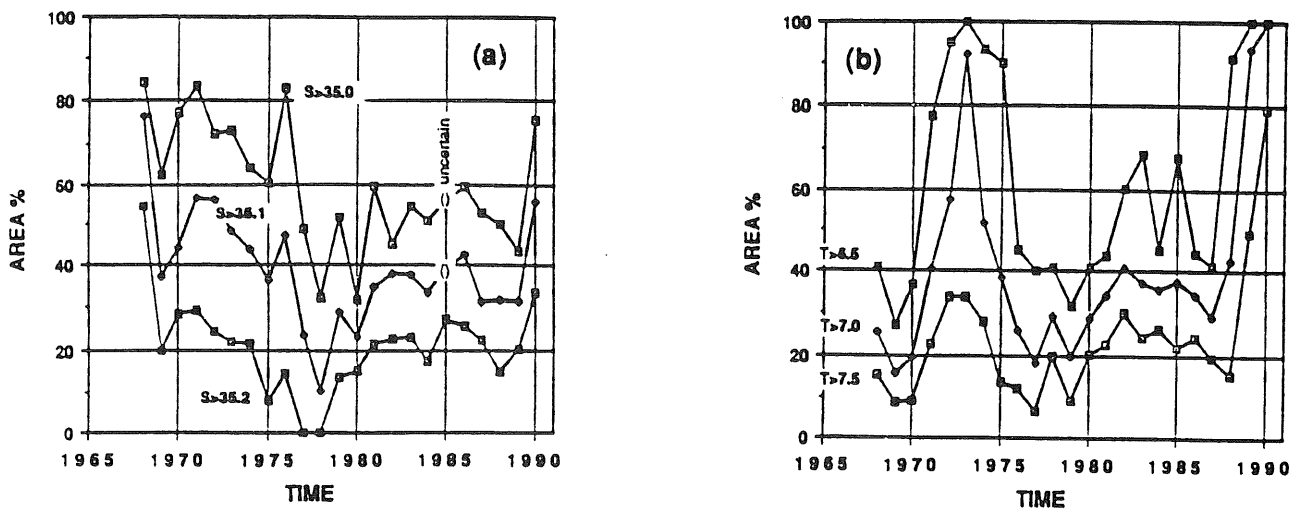


Figure 3.2.1 Time series of the fraction of the area north of 57°N at 50-200 m depth covered with a) mean salinity > 35.0, 35.1 and 35.2; b) mean temperature > 6.5, 7.0 and 7.5°C (from Svendsen and Magnusson, 1992).

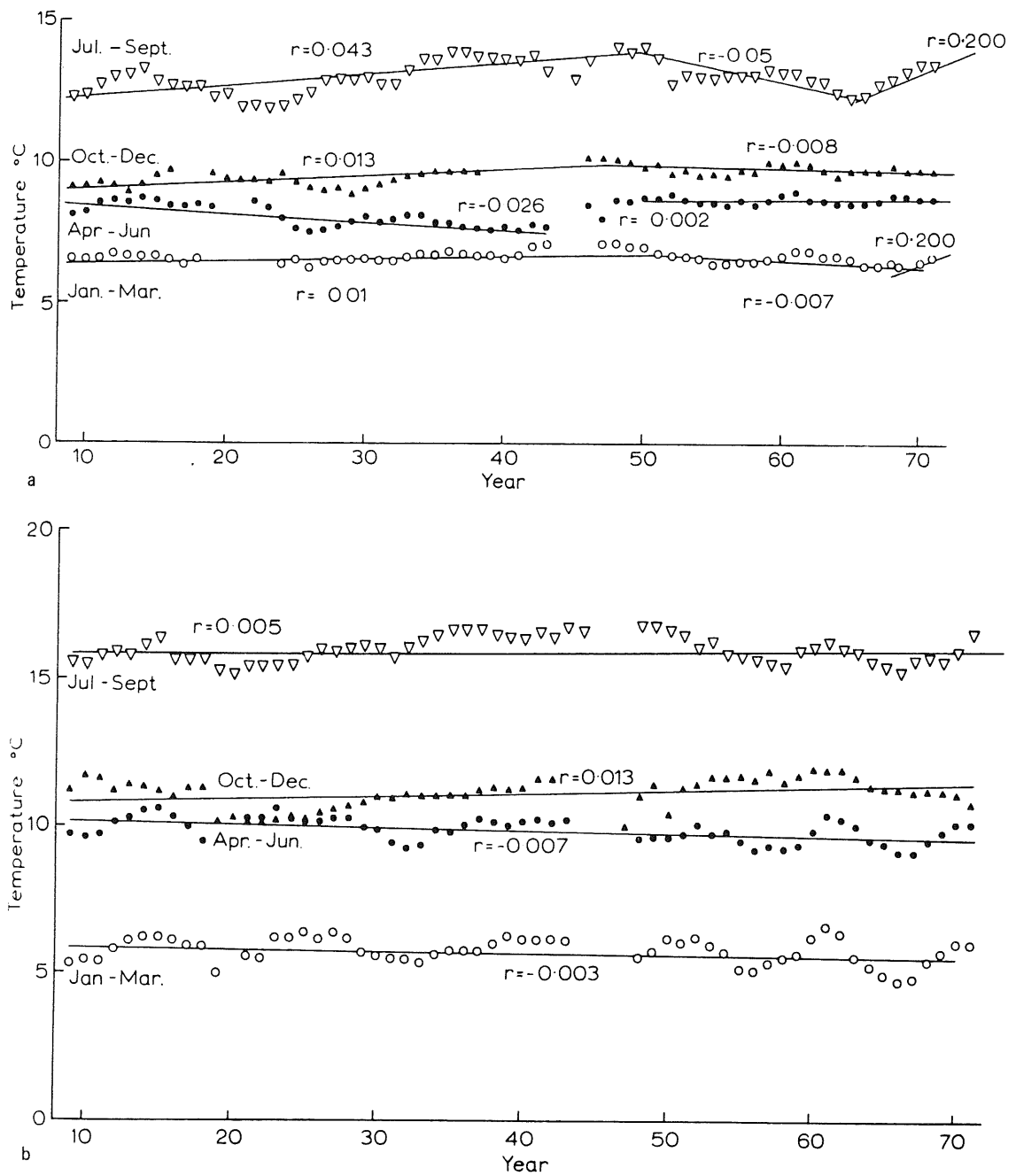
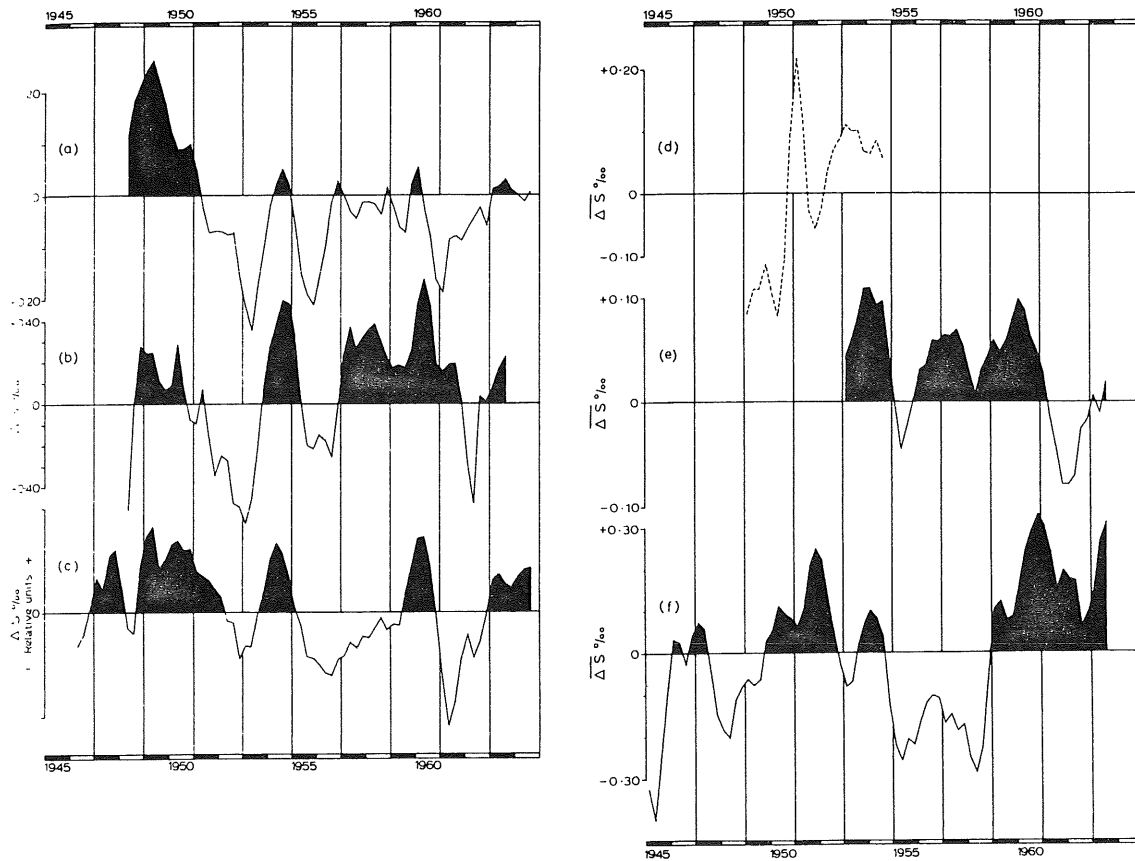


Figure 3.2.2 Running 5-year quarterly means of surface temperature, a) northwestern North Sea; b) southern North Sea (from Otto *et al.*, 1990).



Mean post-war Group salinity trends (mean of all running 3-quarterly means of salinity anomaly for all fields in each Group)
 (a) Eastern Channel Group; (d) Field 47 (Orkney-Shetlands);
 (b) Southern Bight Group; (e) Central North Sea Group;
 (c) German Bight Group; (f) Irish Sea Group;

Figure 3.2.3 Salinities anomalies (running 3-quarterly values) for a) Eastern Channel; b) Southern Bight; c) German Bight; d) Orkney-Shetlands region; e) Central North Sea; f) Irish Sea (from Otto *et al.*, 1990).

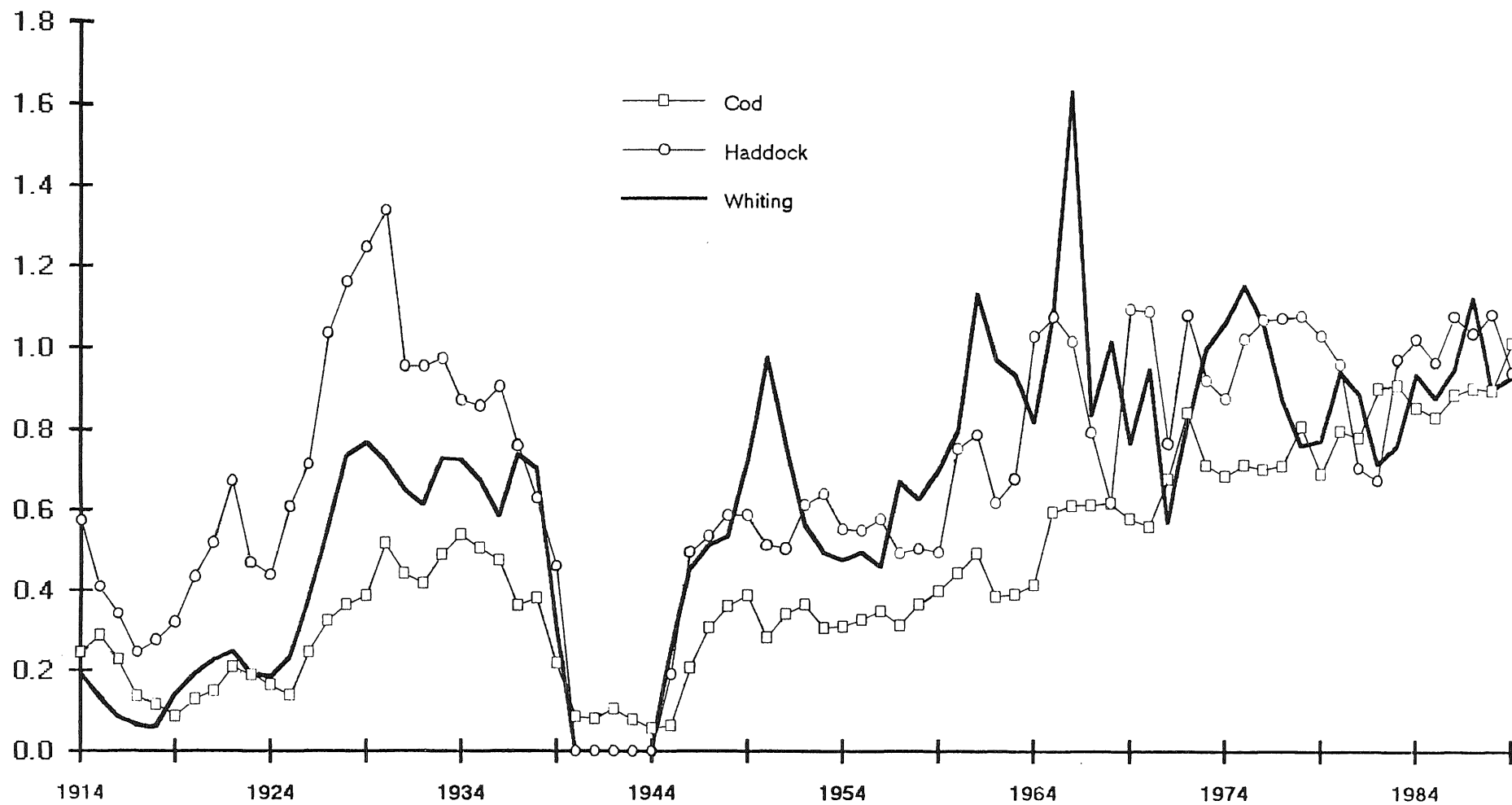


Figure 3.3.3.1 Annual fishing mortality of cod, haddock and whiting caught in the North Sea roundfish fishery. Values from ICES Working Group reports (1961-1989) and from calibrated fishing effort (1914-1960).



Figure 3.3.3.2 Annual fishing mortality of plaice and sole caught in the North Sea flatfish fishery. Values from ICES Working Group reports (plaice: 1952-1990); sole: 1957-1990) and from calibrated fishing effort (plaice: 1907-1951; sole: 1946-1956).



Figure 3.3.3.3 Annual fishing mortality of herring in the North Sea (data from ICES Working Group reports).

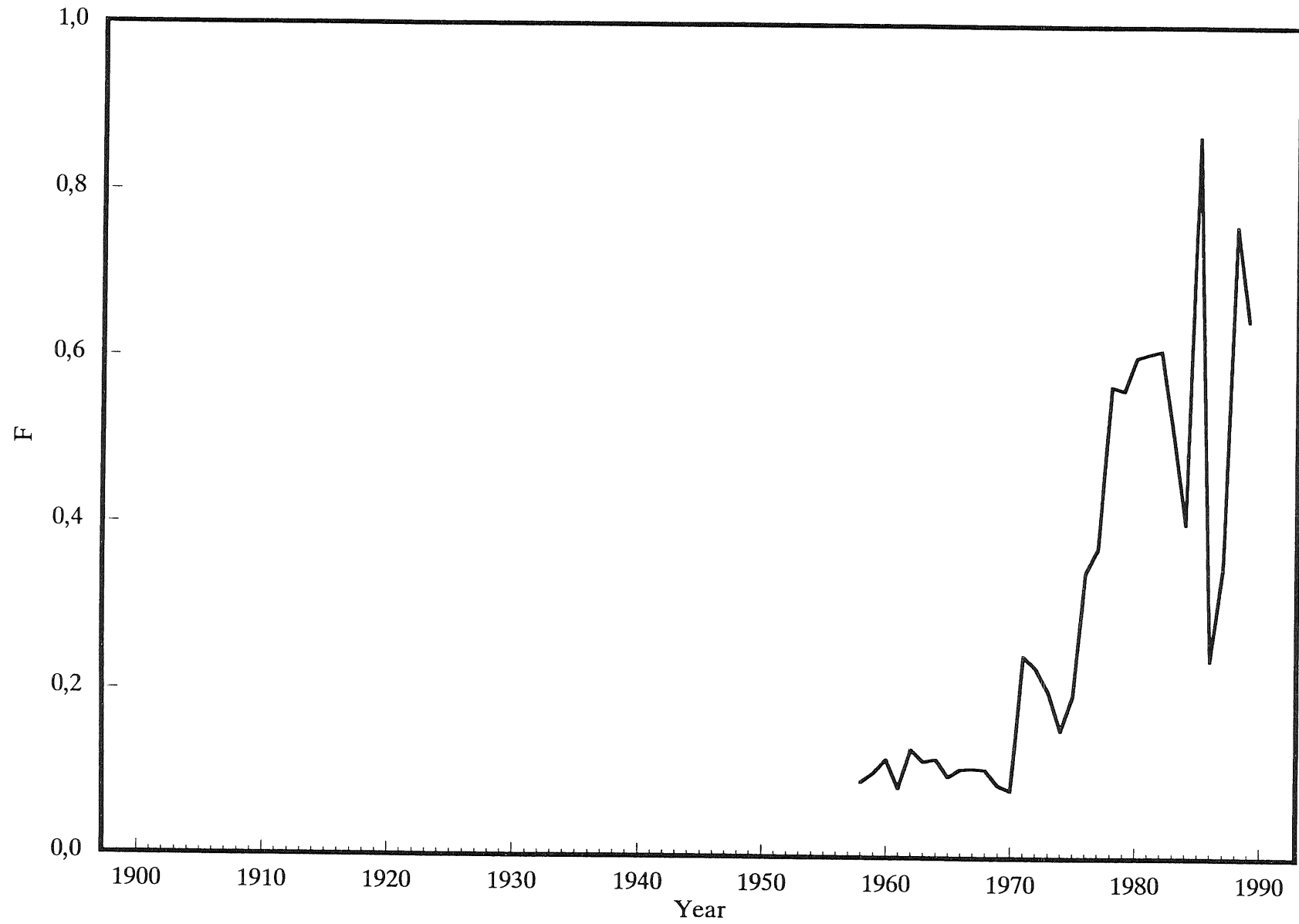


Figure 3.3.3.4 Annual fishing mortality from sandeel caught in the North Sea industrial fishery. Values from Anon., 1991e (1974-1989) and as predicted from fishing effort (1958-1973).

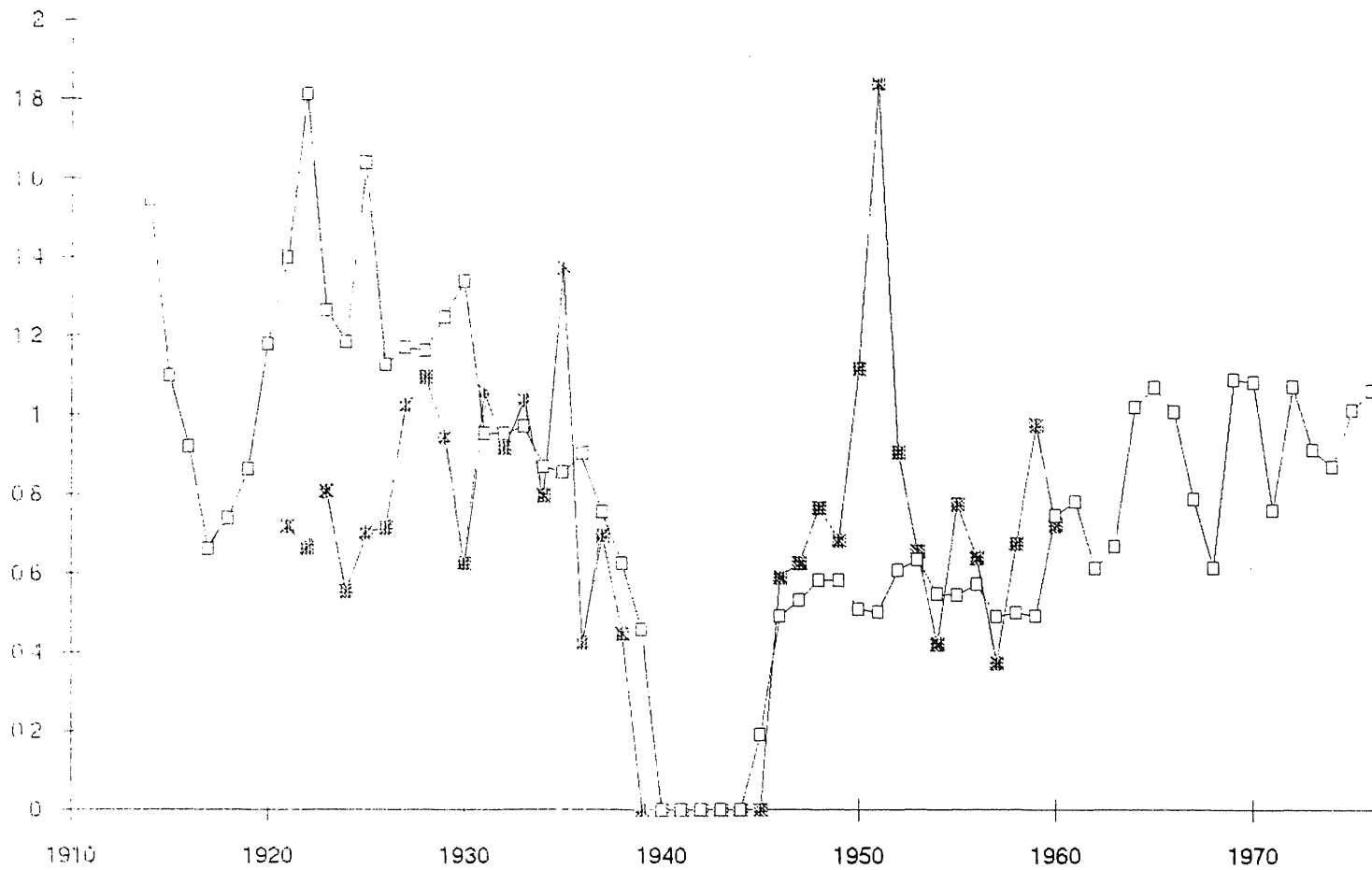


Figure 3.3.3.5 Fishing mortality of haddock generated by uncalibrated effort series and compared to fishing mortality obtained from VPA (Pope, pers. comm.).

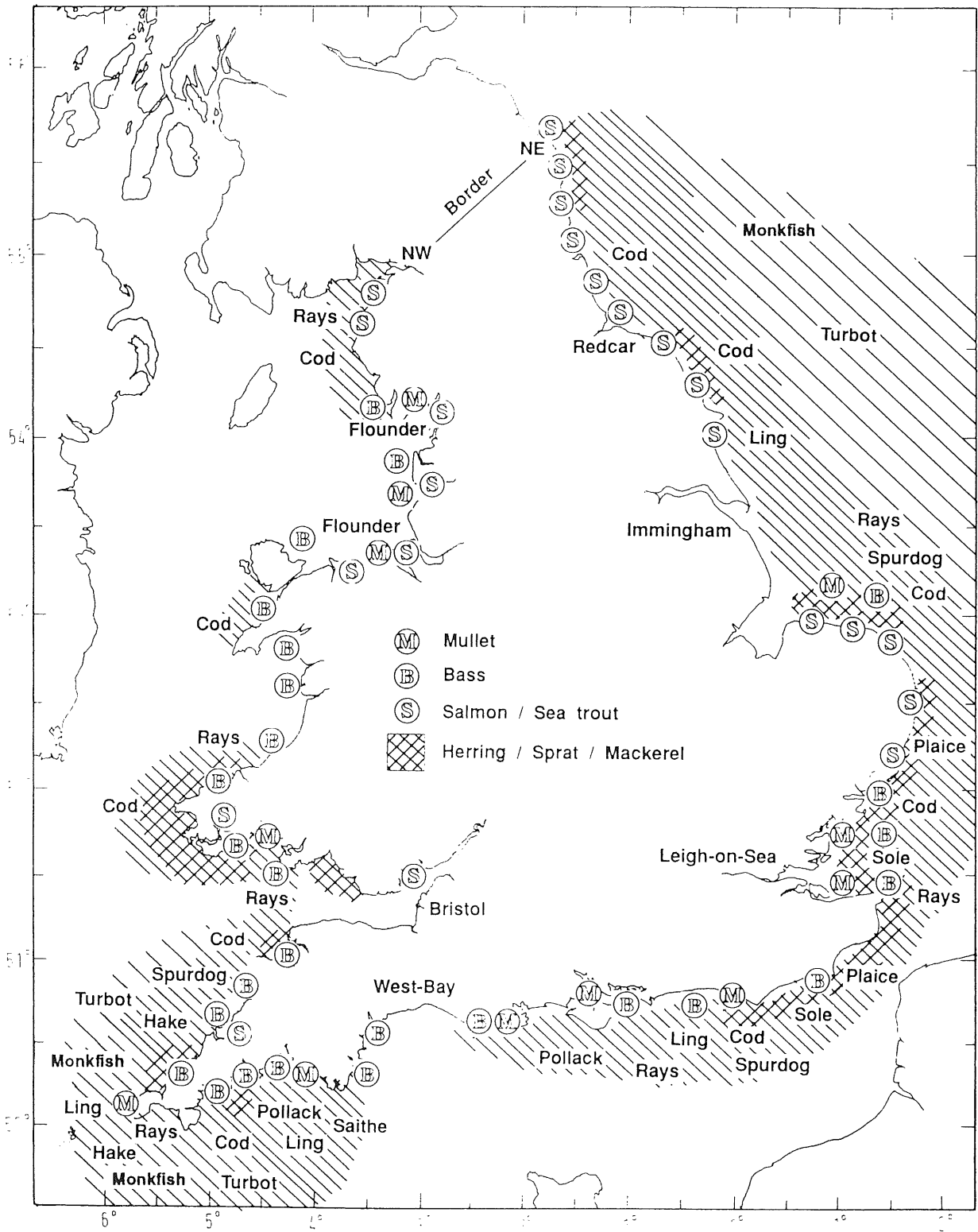


Figure 4.4.3 The distribution of English and Welsh gill-net fisheries in 1989 (from Potter and Pawson, 1991).

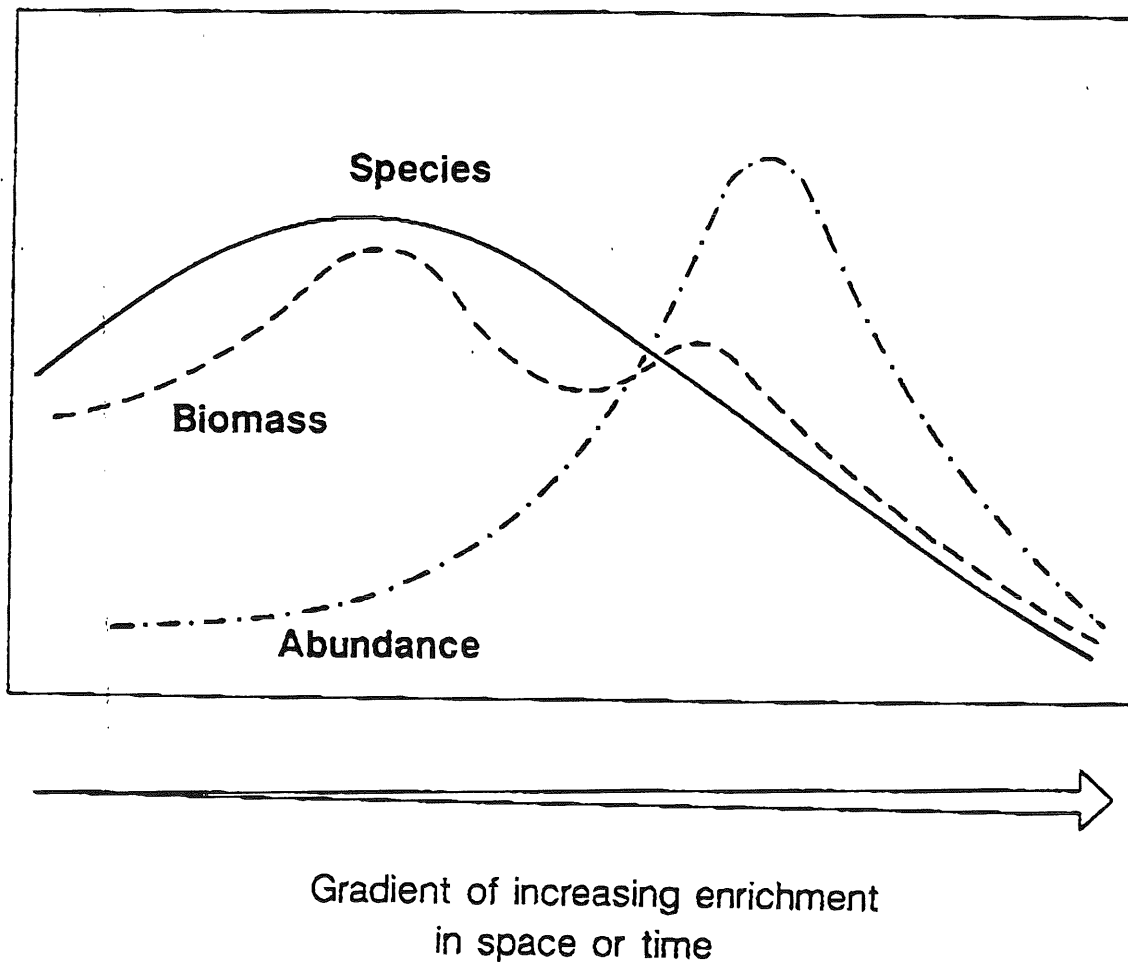


Figure 5.3 Empirical model for changes in the macrobenthos in response to organic enrichment (after Pearson and Rosenberg (1978) from Rees *et al.* (1991)).

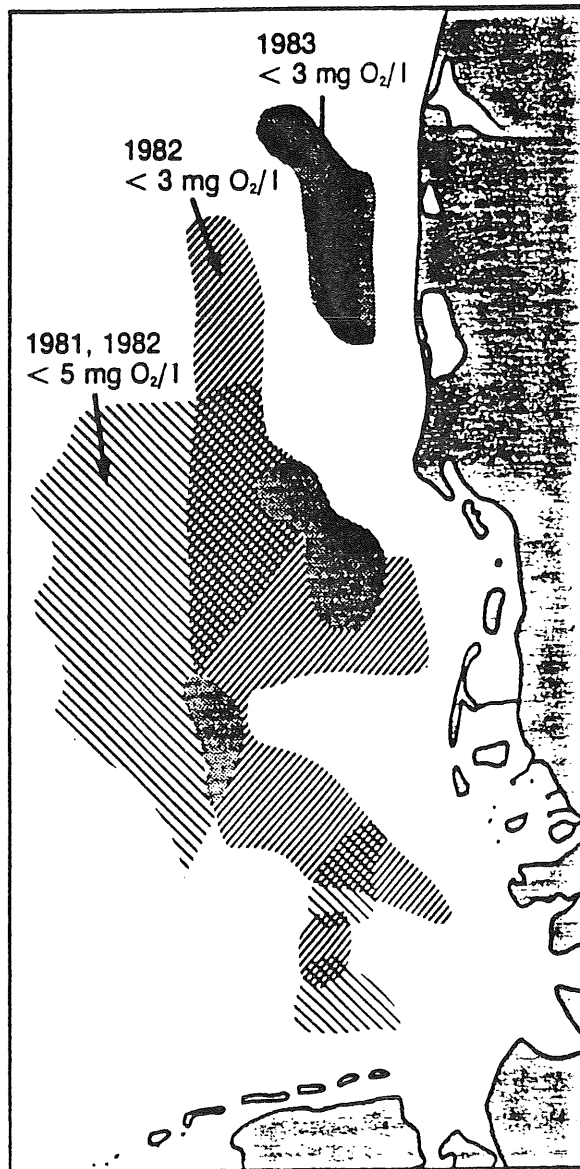


Figure 5.4.1 Areas of low oxygen concentration in near-bottom water on the Danish west-coast and German Bight in August 1981, 1982 and 1983 (after Dethlefsen and Westernhagen, 1983).

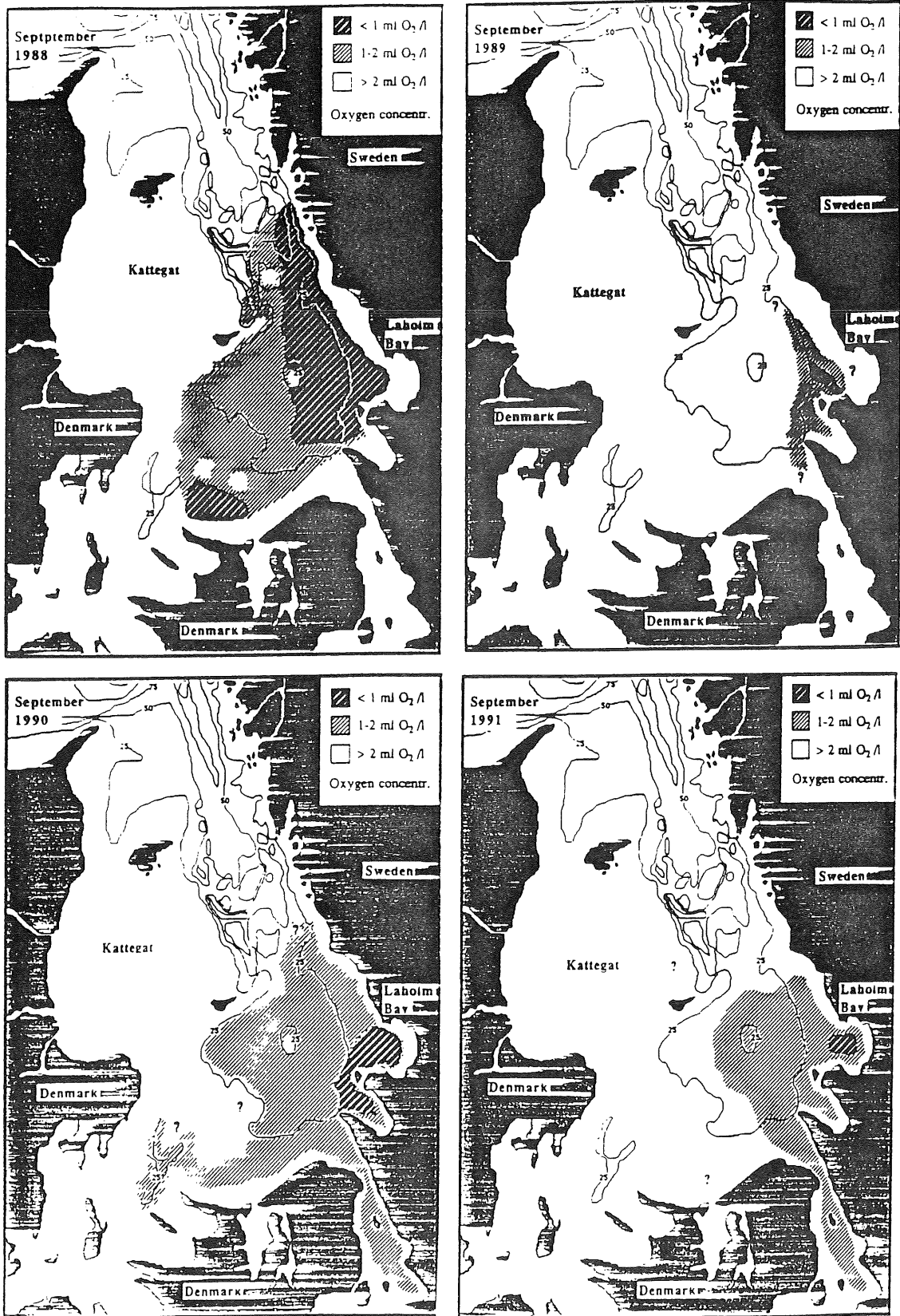


Figure 5.4.2 a-d Oxygen concentrations in the Southern Kattegat in September, between 1988 and 1991.

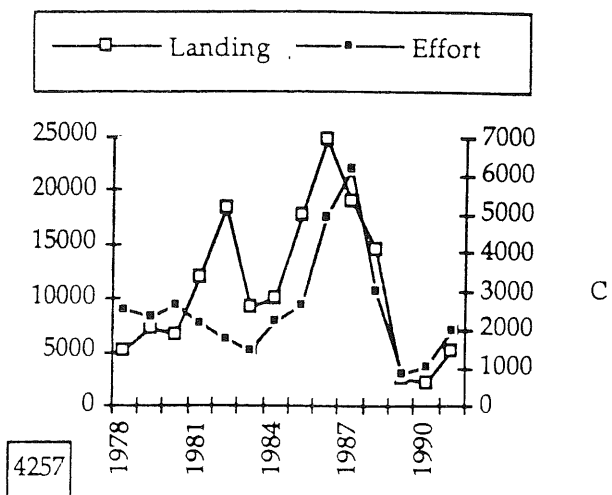
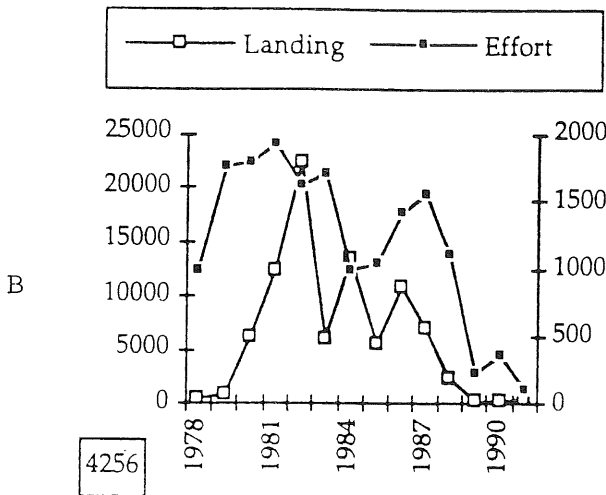
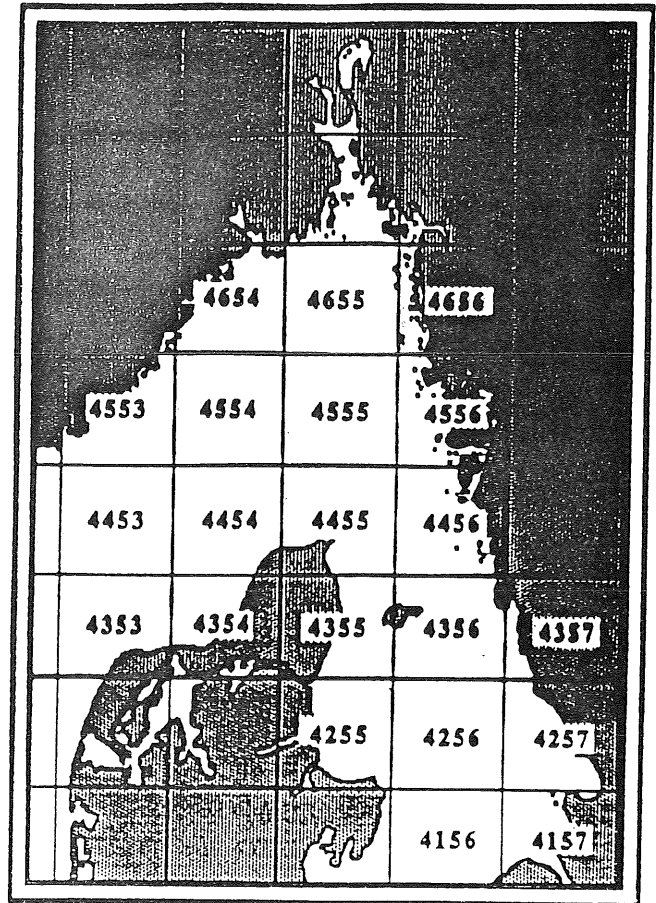
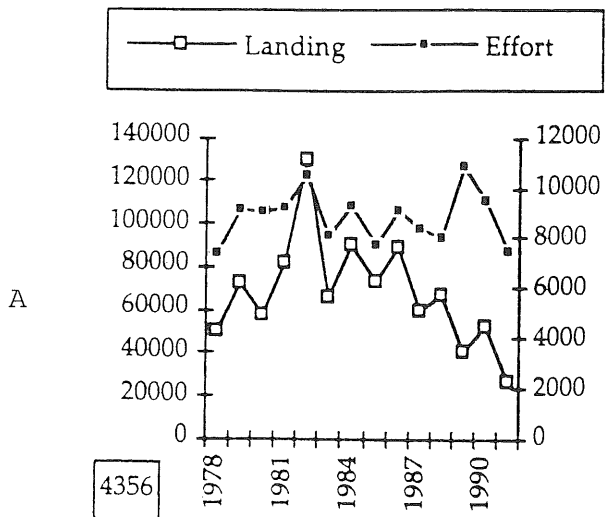
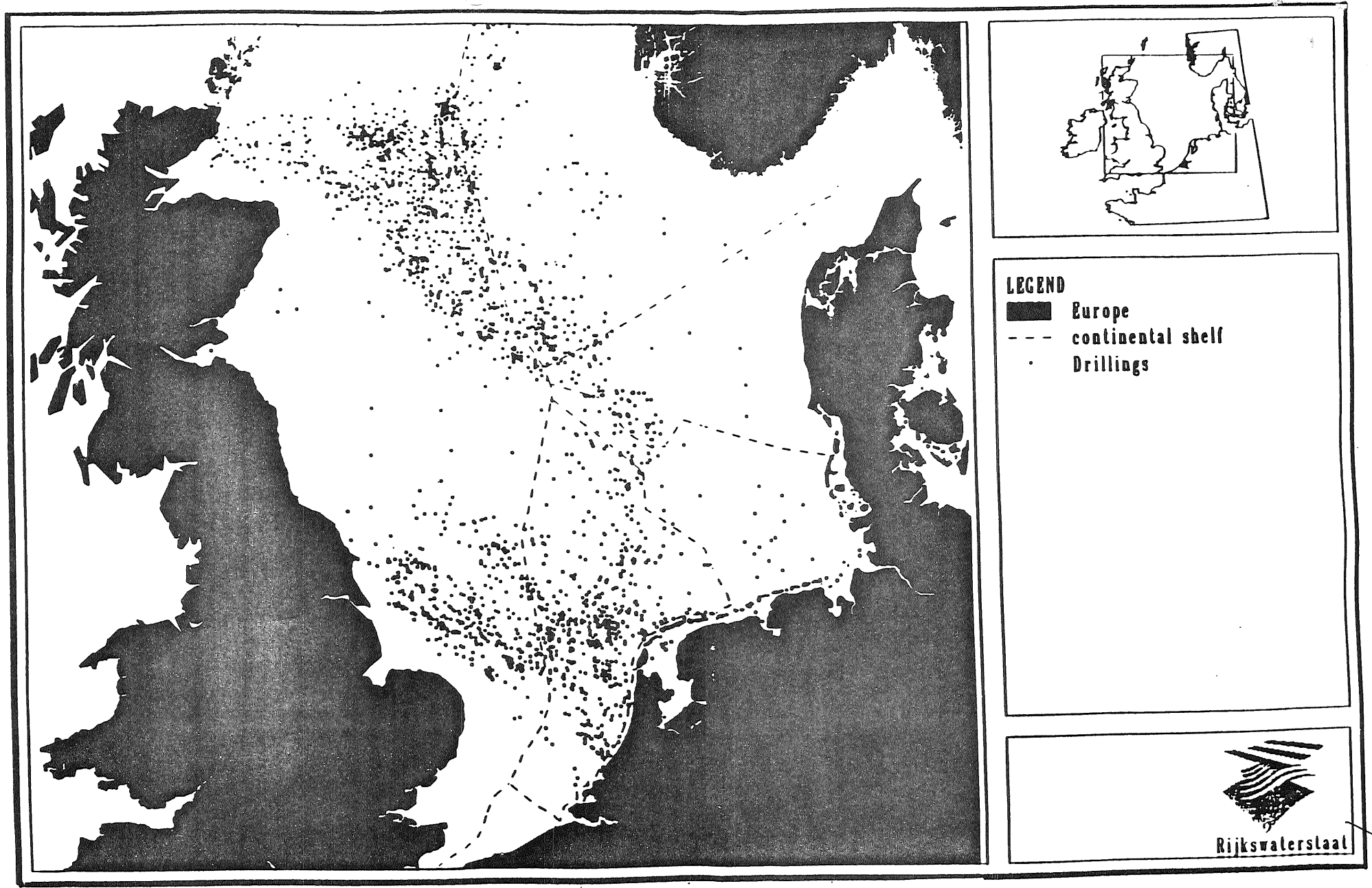


Figure 5.4.3 *Nephrops* landings (kg) and effort (hours) for single trawl in three fishing areas (A = 4356; B = 4256; C = 4257) of the Kattegat (D) from 1978-1991 (from Swedish log books).



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Figure 5.8.1 Total amount of drillings up to 1990 in the North Sea between 50°N and 59°N (after Zevenboom *et al.* (1992)).

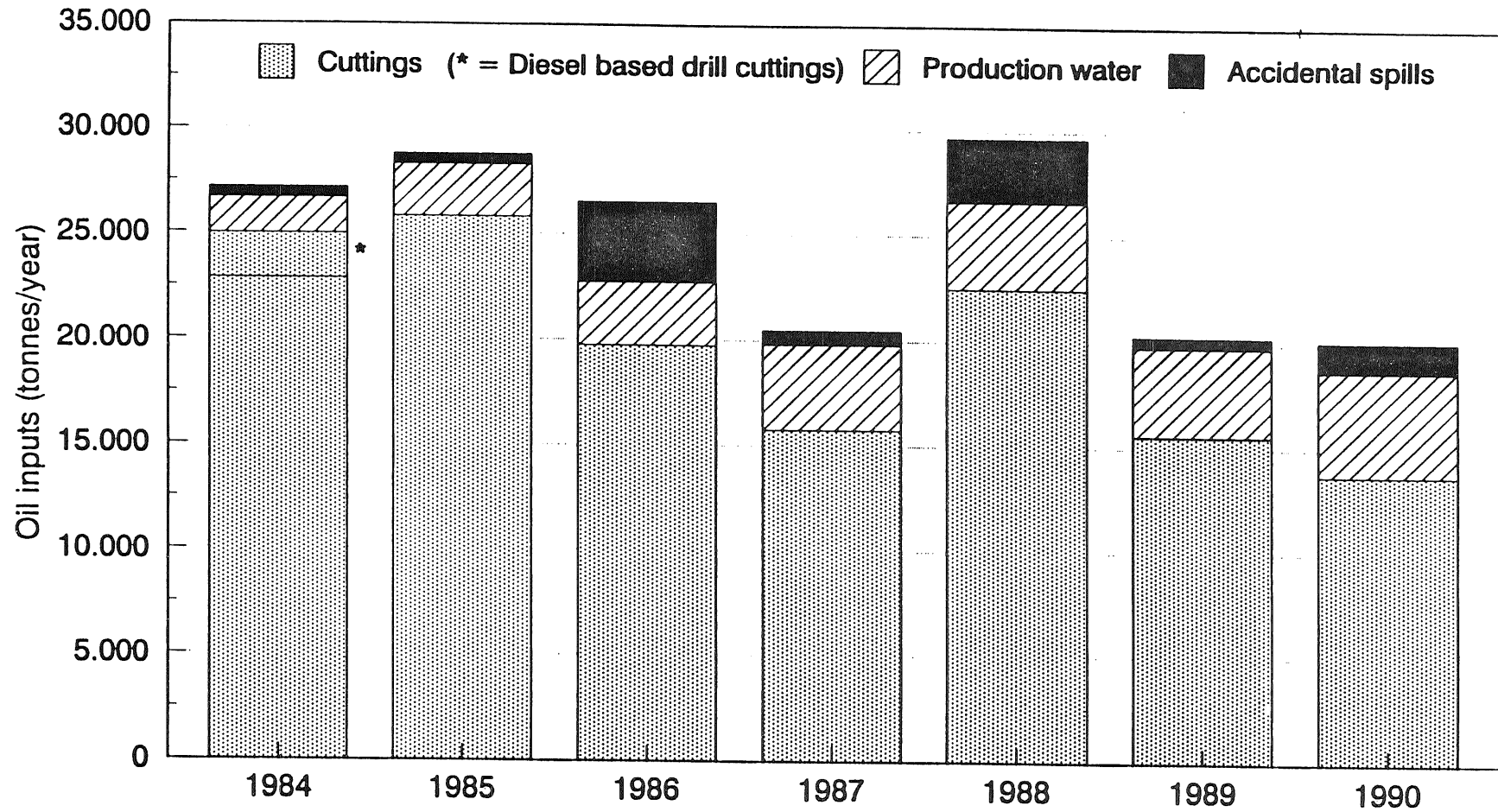


Figure 5.8.2 Total quantities of oil (tonnes per year) for 1984-1990 discharged by offshore mining industry via cuttings, production water or accidental spills into the various North Sea sectors (Denmark + Netherlands + Norway + United Kingdom). Source: Group on Oil Pollution, Paris Commission (PARCOM 13/4/7; GOP 16/2/5; TWG 19/5/6 Rev. 1.) (After Zevenboom *et al.*, 1992.)

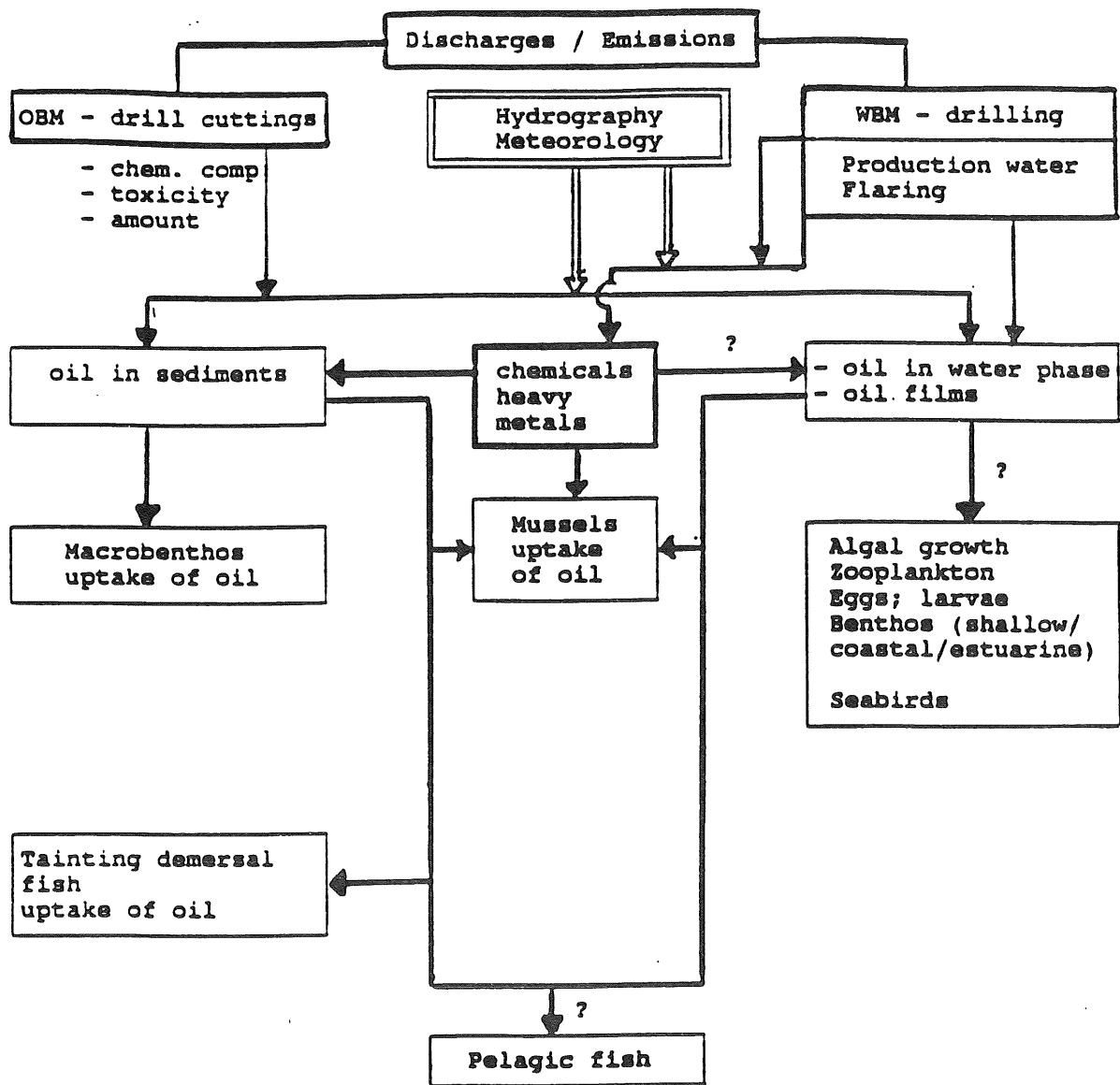


Figure 5.8.3 Effects scheme: adverse effects of offshore mining discharges on the marine ecosystem (after Zevenboom *et al.* (1992)). ? = adverse effects can be expected.

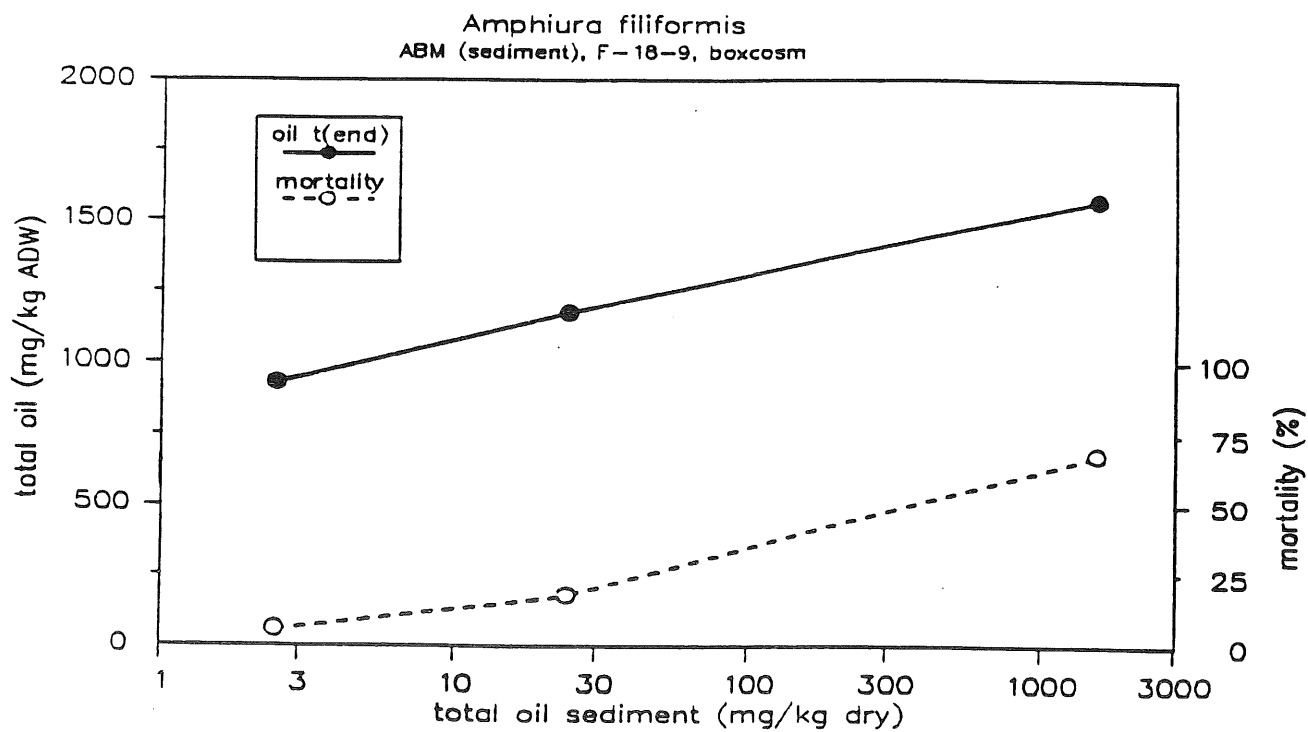
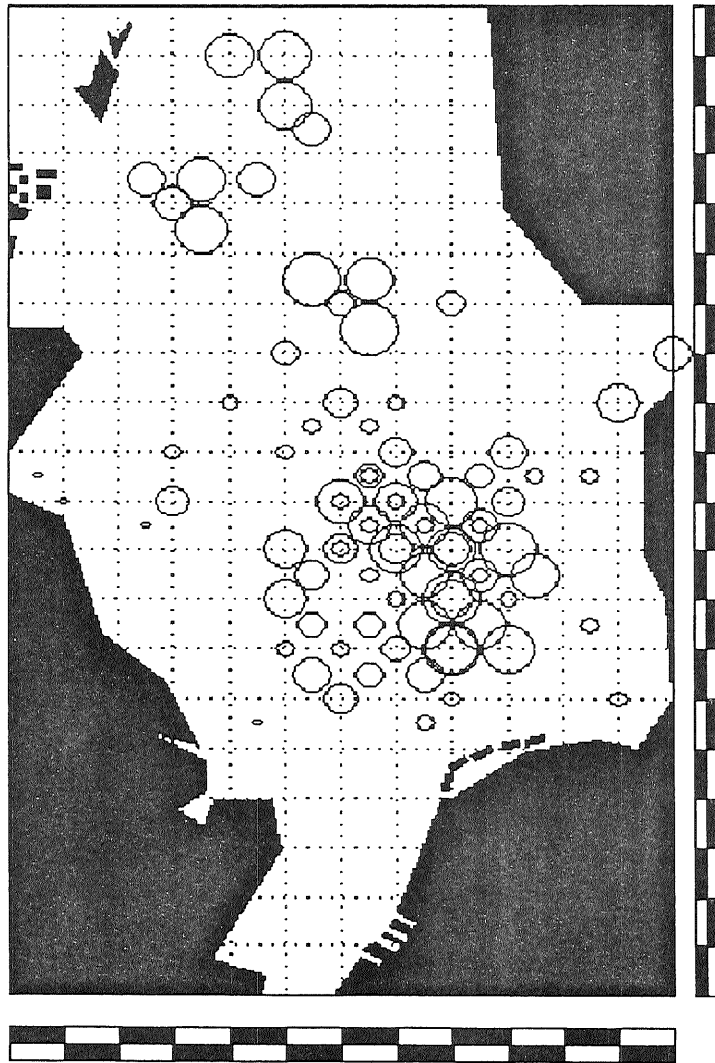
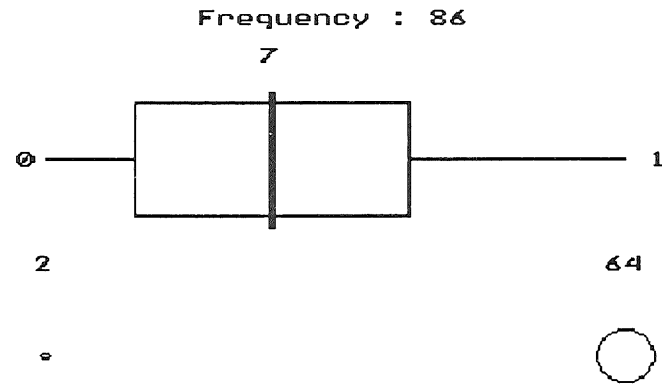


Figure 5.8.4 Oil contents in tissue and relative mortality of *Amphiura filiformis* related to oil content in sediments of drilling site F-18-9 (in boxcosms). (De Jong and Zevenboom, 1991).



**Arctica
islandica**



MACROBENTHOS N. SEA BENTHOS SURVEY
SPECIES DENSITIES

MOLLUSCA
Cyprinidae
Arctica

Figure 7.2.1.1 Abundance of *Arctica islandica* (ind./m²) in 1986. Data from ICES North Sea Benthos Survey.

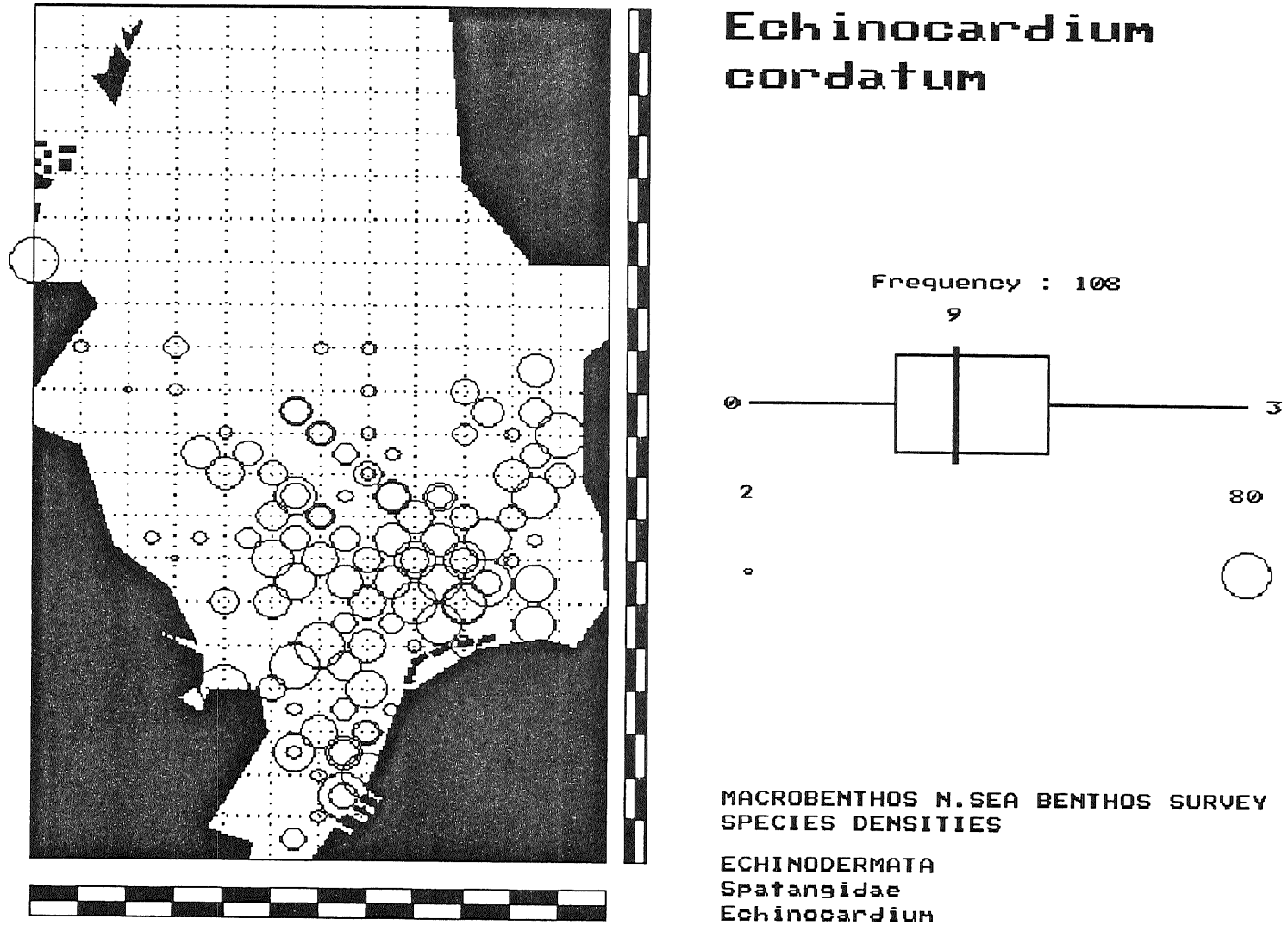


Figure 7.2.1.2 Abundance of *Echinocardium cordatum* (ind./m²) in 1986. Data from ICES North Sea Benthos Survey.

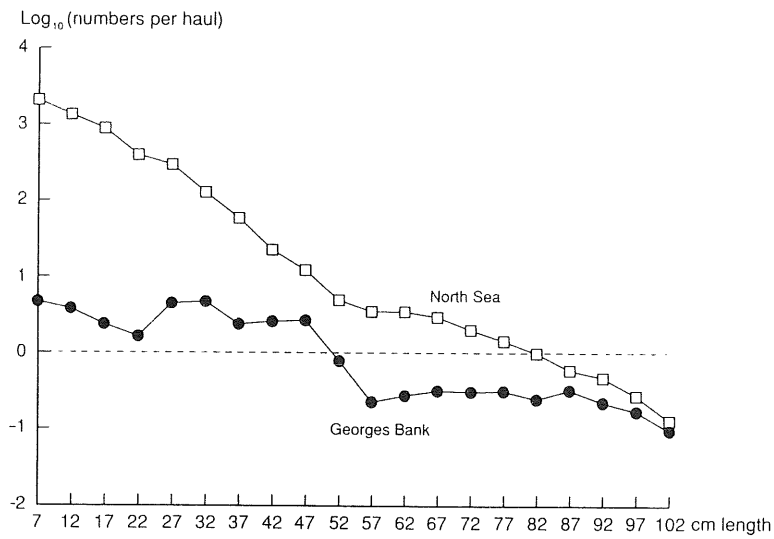
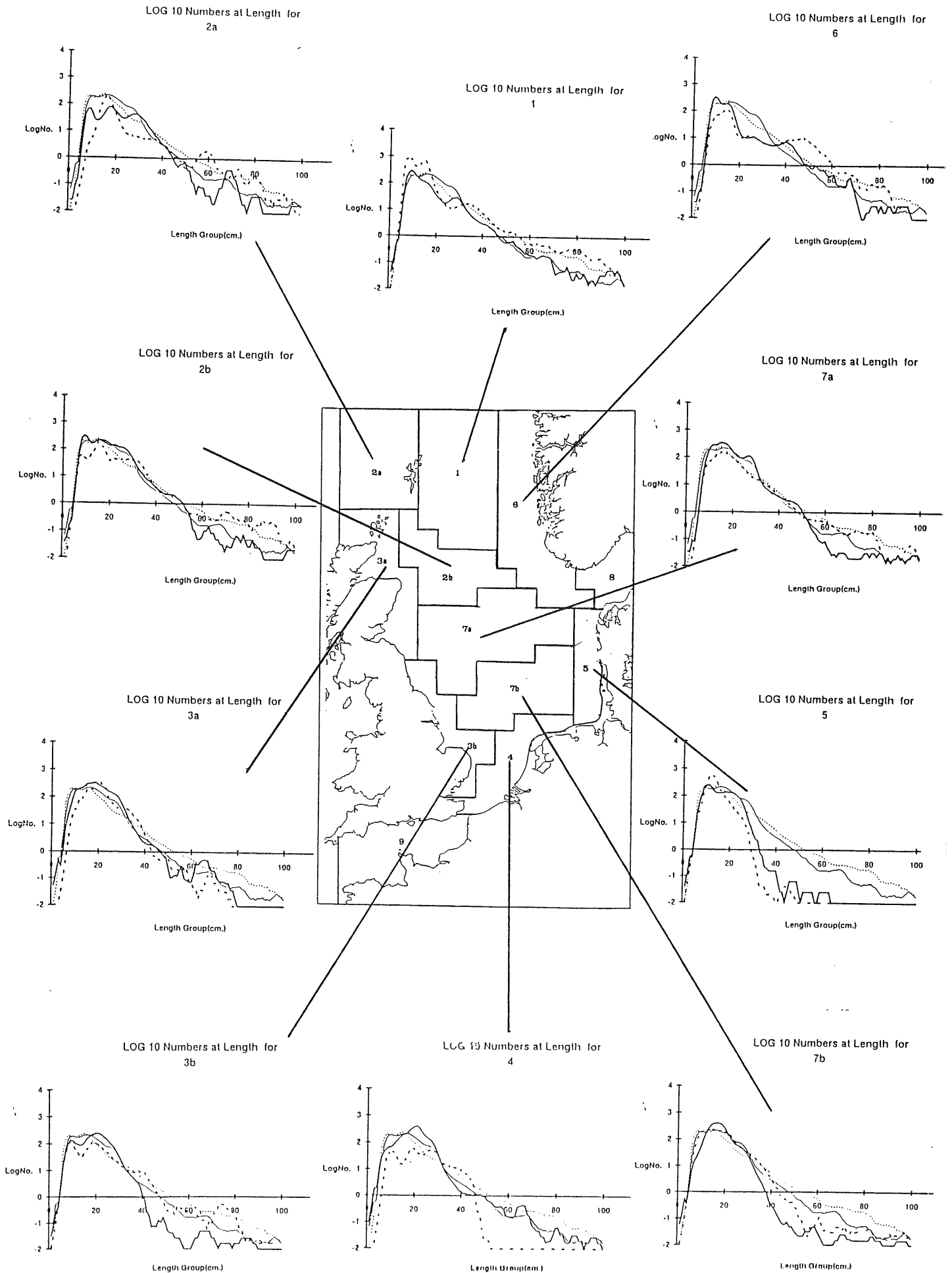


Figure 7.2.2.1 Comparison of the total finfish catch at length per trawl haul on Georges Bank and in the North Sea. (From Pope, 1989.)



118 **Figure 7.2.2.2** Size distribution of fish by NSTF area.

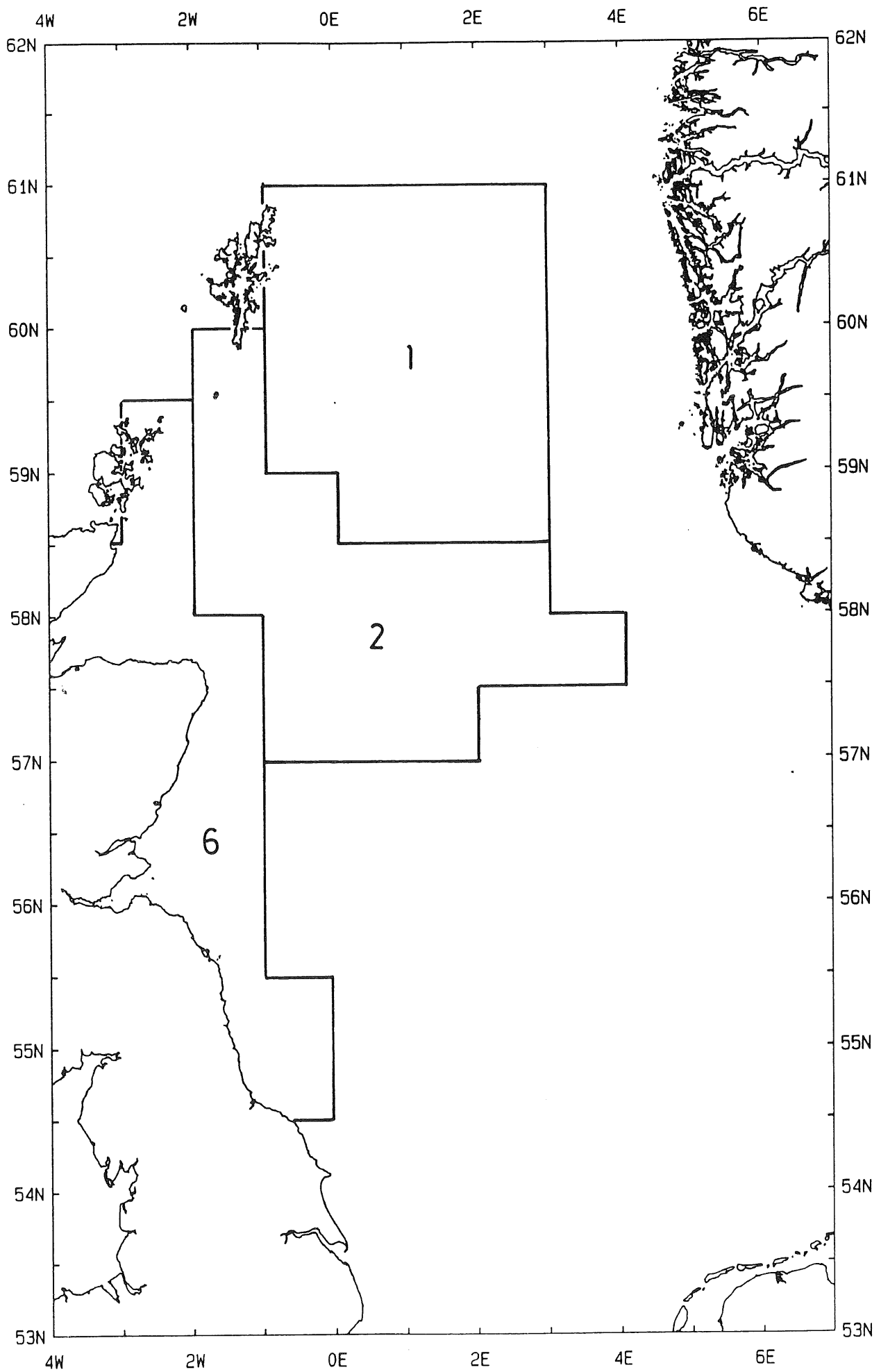


Figure 9.1.1 Extent of the three ERSEM boxes.

Figure 9.1.2a Variation in mean (± 1 S.E.) of three diversity indices between three ERSEM boxes.

Data from International Young Fish Surveys, ERSEM Boxes 6, 2, and 1.

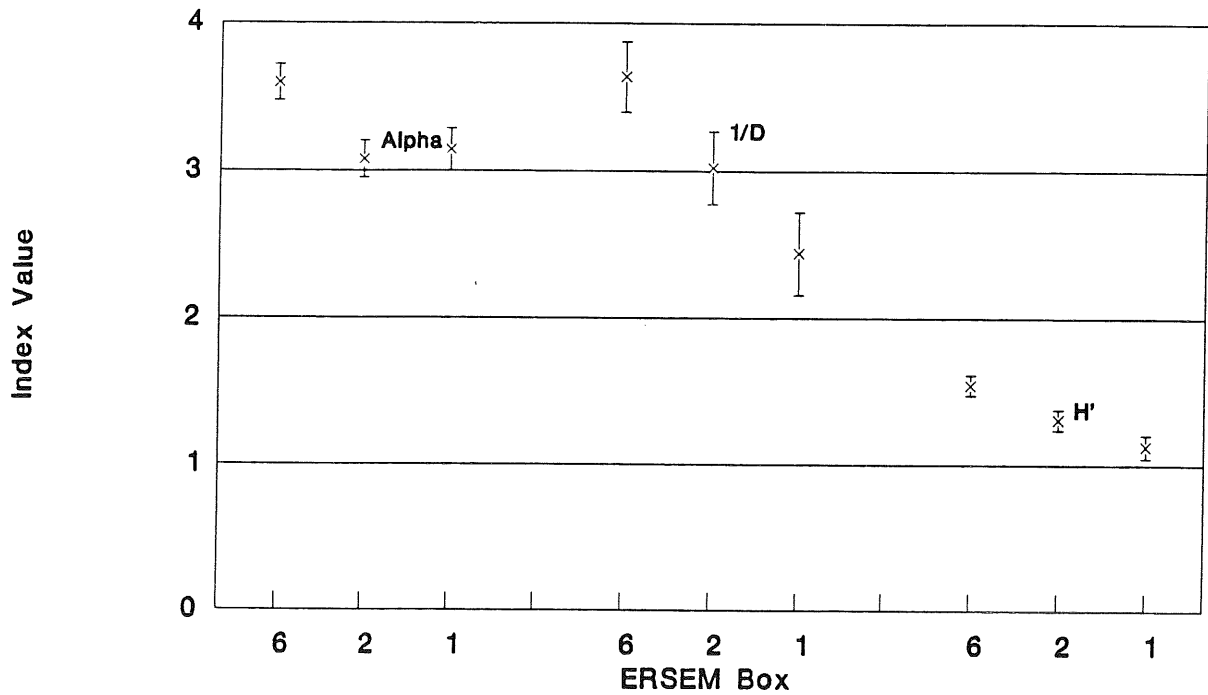


Figure 9.1.2b Annual variation in mean (± 1 S.E.) of Log-series diversity index, alpha; all areas combined.

Data from International Young Fish Surveys, ERSEM Boxes 6, 2, and 1.

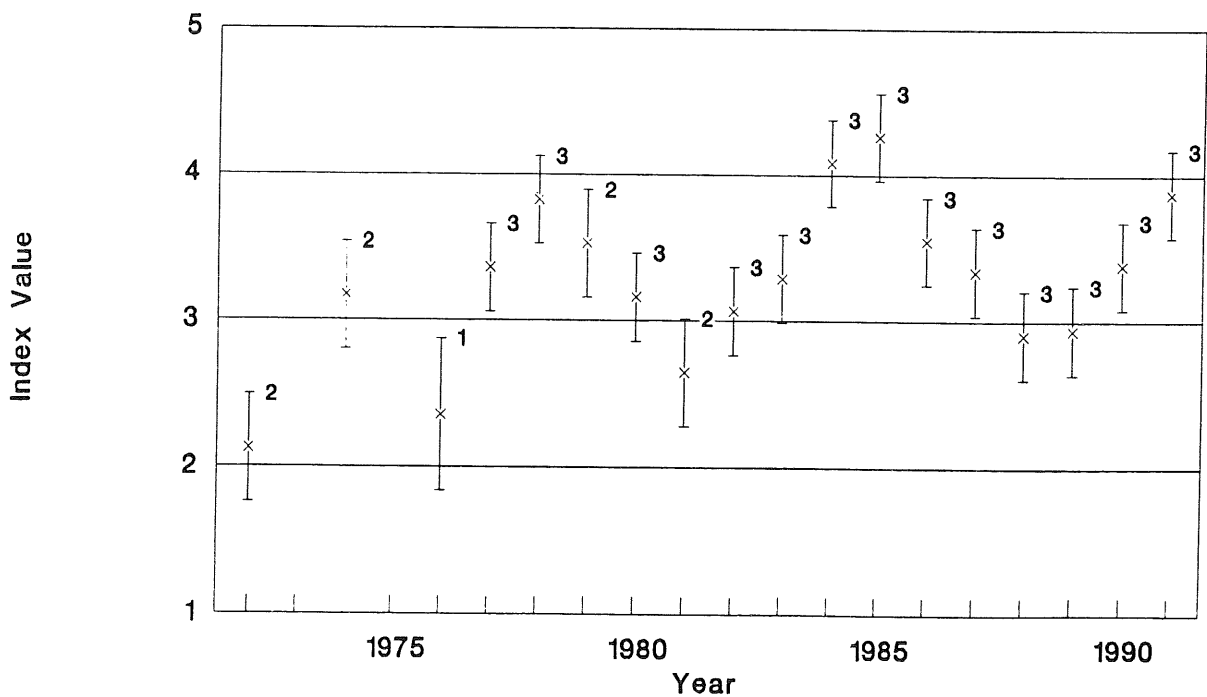


Figure 9.1.3a Log number of fish caught per hour.

IYFS; Box 6; 1972 to 1991.

IYFS; Box 2; 1972 to 1991.

IYFS; Box 1; 1972 to 1991.

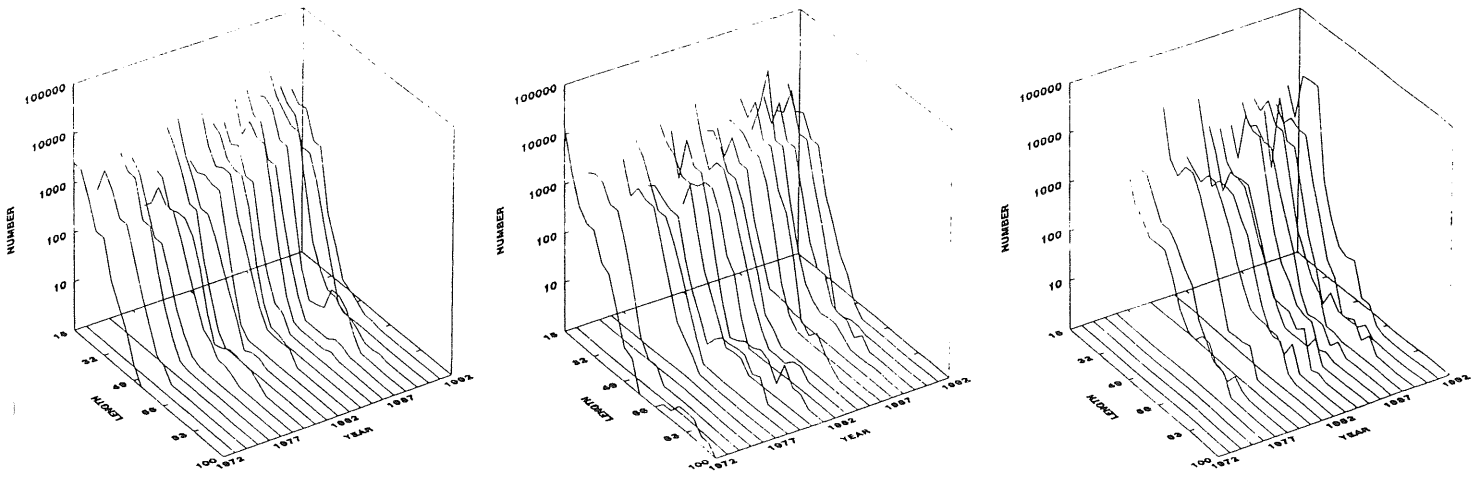
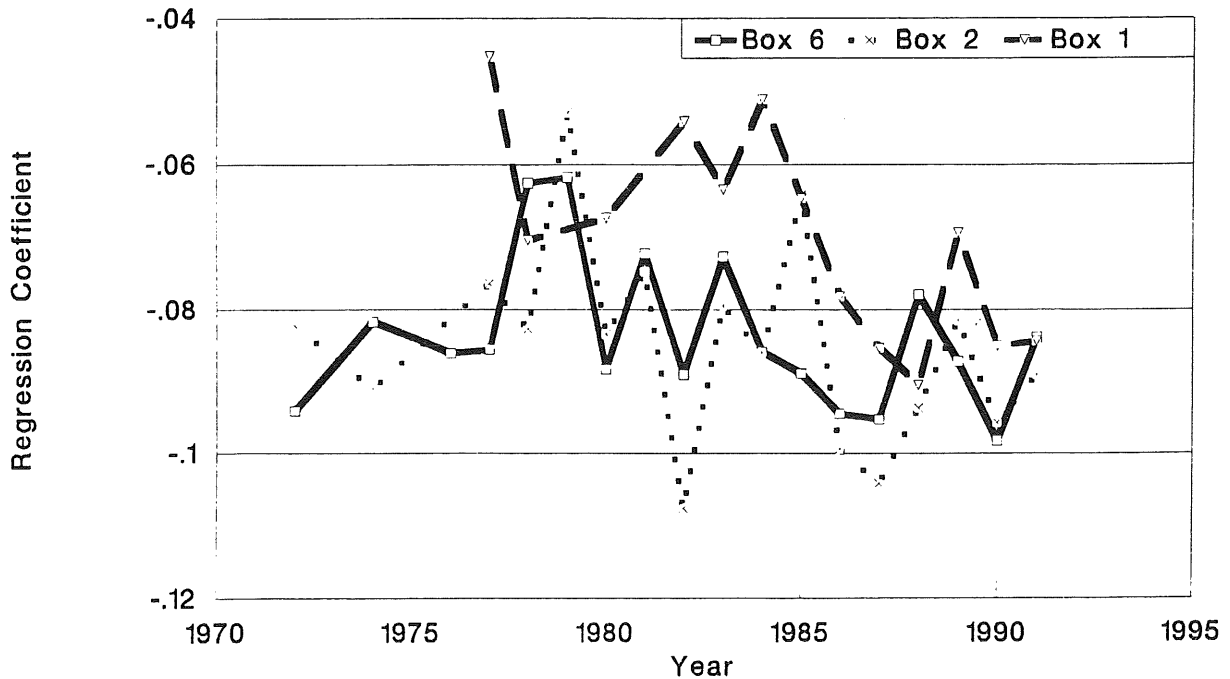


Figure 9.1.3b Annual variation in the regression coefficient of log-number fish caught per hour at length.

Data from International Young Fish Surveys, ERSEM Boxes 6, 2, and 1.



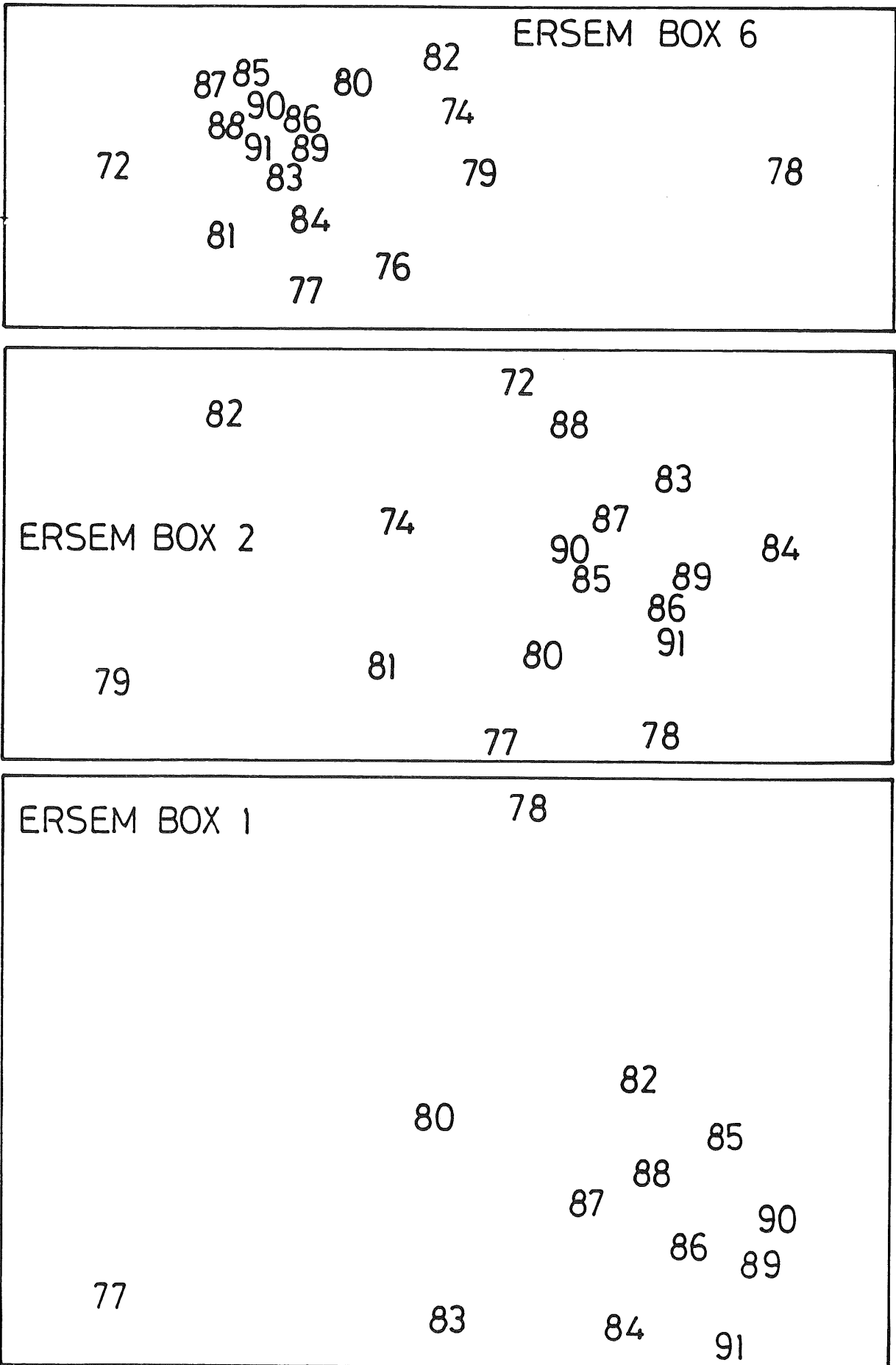


Figure 9.1.4 Multi-dimensional scaling plots showing clustering of years with the most similar species abundance in each ERSEM box.

ANNEX 1

List of Working Documents

1. Dare, P.J. A review of the effects of molluscan dredge fisheries upon benthos and substrates.
2. Greenstreet, S.P.R. Changes in the fish community of the North Sea between 1972 and 1991.
3. de Groot, B. and Vogel, J. D.. A comparison of the energy transferred to the seabed by the passage of a beamtrawl and during a storm.
4. Hall, S.J. Notes on sediment transport, natural large-scale physical disturbance and benthic community dynamics.
5. Hall, S.J. The effects of fishing disturbance in the northern North Sea: Preliminary results of a field investigation.
6. Hall, S.J. Notes on bioturbation in the North Sea.
7. Rees, H.L. and Dare, P.J. Sources of mortality and associated Life-Cycle Traits of selected benthic species: A review.
8. Rice, J. What can we know, measure, and model about feeding interactions?
9. Rowell, T.W. Studies relevant to the ICES Study Group on Ecosystem Effects of Fishing Activities.
10. Sangster, G.I. The survival of fish escaping from fishing gears.
11. Stokes, T.K. Report of a meeting on the evolution of exploited populations held in Julich, Germany, September 1991.
12. Valk, L. van der. Estimated amount of physical disturbance of the seabed in the shallow southern North Sea due to natural causes.

ANNEX 2

Distribution of fishing effort by NSTF areas in 1989

Catch and effort data from 49 fleets composed of types defined by nation/gear combinations were made available from the archives of the Working Group on the Improvement of North Sea Exploitation Patterns established by the EC's Scientific and Technical Committee on Fisheries (STCF). Data were made available by all countries except Belgium who have some problems with data confidentiality. The data consist of catch by fleet, area and quarter in 1989. The geographical distribution of effort has to a large extent been based on logbooks. Vessels of less than 10 GRT are not obliged to deliver logbooks, and the small scale coastal and inshore fisheries have therefore not been covered particularly well.

For all of the 49 fleets there are catch data and for most there are effort data, however, the Norwegian fleets are without effort data as are the English, Scottish and French Other gears. In order to make the data comprehensible the 49 fleets were classified into 9 main gear types. Table A.2.1 shows the various countries gears and how they were classified to gear types 1-9.

An ANOVA was used to standardize the effort for the various fleets included in each of the gear types by fitting the catch rate by fleet, area and quarter to a model containing a fleet factor and an area, quarter interaction term (see Sect. 3.3.4). The ANOVA provides combined estimates of area quarter catch rates, standardized to units of one particular fleet. The area quarter catch of each fleet was divided by the corresponding standardized catch rate in order to produce standardized effort values for each gear type by area and quarter.

The catch by gear type, area and quarter is given in Table A.2.2.a-i.

Three gears; German pelagic freezers and Scottish pair trawls for herring and for mackerel; did not fit into the larger categories. The catches from the three gears are included in Table A.2.2.g where the catches from pelagic trawls are presented, but the catch rates were not included in the estimates of effort levels by area and quarter. As all three gears are pelagic their exclusion will not affect estimates of area swept by gears in contact with the bottom. The Other gears category is a mixture of fixed gears and unclassified gears. The catches show an irregular pattern among quarters and areas, consistent with the heterogeneous nature of this category of gear. The ANOVA model did not reduce the sum of squares, and no standardized effort levels were estimated.

Table A.2.3.a-h shows the output from the ANOVA for the 8 remaining gear types.

Detailed comments on these tables are as follows:

Table A.2.3.a, Beam trawlers did not fish at all in areas 1 and 2a and only intermittently in area 3a. Overall area 4 received the most effort, with substantial effort also in areas 7 and 5. Effort is evenly distributed among the quarters. The ANOVA model shows a reasonable residual variation and a fair reduction in sum of squares, indicating that the standardization of the effort to units of Netherlands Beam trawlers of more than 300 HP was successful.

Table A.2.3.b, Otter Trawlers fished in all areas in all quarters, but areas 1 to 3 received most effort. Areas The ANOVA model shows a reasonable residual variation and the sum of squares is reduced substantially by the model indicating that the standardization to Scottish Otter trawler hours is reasonable.

Table A.2.3.c, Pair trawlers fished in all areas in all quarters. Overall areas 4 and 6 was the most heavily fished with areas 3b, 5 and 7b fished at lower levels than the other areas. Fishing effort was well represented in all quarters. The ANOVA model shows a reasonable residual variation and the sum of squares is reduced substantially by fitting the model with scottish pair trawlers as the standard.

The areal distribution of the industrial effort is influenced by the seasonality of the various species caught in the fishery.

Table A.2.3.d, Industrial Otter Trawlers concentrated their effort in areas 6 and 7, while areas 2a, 3 and 4 received very little effort. Area 3a was very lightly fished. The effort was standardized to Danish Industrial Otter Trawlers larger than 100 GRT days absent. The ANOVA model shows a reasonable residual variation and a substantial reduction in sums of squares.

Table A.2.3.e, Industrial Pair Trawlers mostly fished in area 6 and not at all 2a, 3b and 4. The effort was standardized to Danish Industrial Pair Trawlers larger than 100 GRT days absent. A substantial amount of variation in CPUE remains unaccounted for by the standardization.

Table A.2.3.f, Seiners and Ring Netters fished in all areas in all quarters. The effort was standardized to

Scottish seiner hours. Area 1 was the most heavily fished with substantial fishing in 2b and 7a. Area 4 the least heavily fished. The ANOVA model shows a very low residual variation and the sum of squares is reduced substantially by fitting the model.

Table A.2.3.g, Pelagic Trawlers fished very lightly in 1989. The effort was standardized to French high seas fleet hours fished. Most effort was applied in the second half of the year. Overall area 4 was the most heavily fished, but a relatively large amount of effort was also allocated to areas 3b and 7b. No effort was reported from areas 5 and 6. Since the fishery is very patchily distributed among areas and quarters, the ANOVA was expected to perform poorly. However, the reduction in sums of squares is surprisingly large and suggests a very good fit of the model.

Table A.2.3.h, Purse Seiners fished mostly in northern areas in 1989. Most effort was applied in the second half of the year. The effort was standardized to Danish purse seiners days absent. Overall area 1 was the most heavily fished and areas 3b, 4, 5 and 7b were hardly fished or not fished at all. The ANOVA model shows a poor residual variation and the sum of squares is not reduced by fitting the model, suggesting purse seine effort is heterogeneous among fleets.

Table A.2.3.i, Gill Netters fished in all areas and quarters, but very little in the northern areas and most in areas 4 and 5. Most effort was applied in the first two quarters. The ANOVA model shows a small residual variation and good reduction of sum of squares, indicating a good fit. Danish gill netters were used as the standard.

Table A.2.1 Fleets used in analysis by gear type.

FLEET DESCRIPTION	EFFORT UNIT	INDUSTRIAL PAIR TRAWLERS	
		DEN IND. TRAWL PAIR < 100 GRT	DAYS ABSENT
		DEN IND. TRAWL PAIR > 100 GRT	DAYS ABSENT
BEAM TRAWLERS			
ENG ALL BEAM TRAWLS	HOURS FISHED		
GER SMALL BEAM TRAWL	Only Catch Data Available		
FRA COASTAL FLEET-BEAM	HOURS	SEINES AND RINGNETS	
NET BEAM TRAWL <300HP	DAYS FROM PORT	ENG ALL SEINES AND RING NET	HOURS FISHED
NET BEAM TRAWL >300HP	DAYS FROM PORT	DEN DANISH SEINE	DAYS ABSENT
		SCO SEINE NET	HOURS FISHING
OTTER TRAWLERS			
ENG ALL TYPES OTTER TRAWL	HOURS FISHED	PELAGIC TRAWLERS	
GER OTTER TRAWL (L.D. CUTTER)	Only Catch Data Available	FRA COASTAL FLEET-PELAGIC	HOURS
GER OTTER TRAWL (L.D. TRAWLER)	Only Catch Data Available	FRA FREEZERS + BIG TRAWL PELAGIC	HOURS
GER OTTER TRAWL (S.D. CUTTER)	Only Catch Data Available	FRA HIGH SEA FLEET - PELAGIC	HOURS
DEN CON. TRAWL SIN. < 100 GRT	DAYS ABSENT		
DEN CON. TRAWL SIN. > 100 GRT	DAYS ABSENT	PURSE SEINERS	
SCO LIGHT TRAWL	HOURS FISHING	DEN PURSE SEINE	DAYS ABSENT
SCO NEPHROPS TRAWL	HOURS FISHING	SCO PURSE SEINE (HERRING)	HOURS FISHING
SCO TRAWL	HOURS FISHING	SCO PURSE SEINE MACKEREL	HOURS FISHING
NOR SAI THE TRAWLERS (OTTER)	Only Catch Data Available	NOR PURSE SEINERS	Only Catch Data Available
FRA COASTAL FLEET-BOTTOM	HOURS		
FRA FREEZERS+BIG TRAWL-BOTTOM	HOURS	GILL NETS	
FRA HIGH SEA FLEET-BOTTOM	HOURS	DEN GILL-NET	DAYS ABSENT
NET OTTER TRAWLERS	DAYS FROM PORT	ENG GILL-NET	HOURS FISHED
		SCO GILL-NET	DAYS FISHED
PAIR TRAWLERS		NET GILL-NET	DAYS ABSENT
ENG ALL PAIR TRAWLS	HOURS FISHED	FRA GILL-NET	Only Catch Data Used
DEN CON. TRAWL PAIR < 100 GRT	DAYS ABSENT		
DEN CON. TRAWL PAIR > 100 GRT	DAYS ABSENT	OTHER GEARS(inc. gill nets)	
SCO PAIR TRAWL DEMERSAL	HOURS FISHING	ENG ALL OTHER GEARS	HOURS FISHED
SCO PAIR TRAWL HERRING	HOURS FISHING	DEN GILL-NET	DAYS ABSENT
SCO PAIR TRAWL MACKEREL	HOURS FISHING	DEN OTHER	DAYS ABSENT
		SCO OTHER	HOURS FISHING
INDUSTRIAL OTTER TRAWLERS		NOR DUMMY FLEET	Only Catch Data Available
DEN IND. TRAWL SIN. < 100 GRT	DAYS ABSENT	FRA COASTAL FLEET-VARIOUS	HOURS
DEN IND. TRAWL SIN. > 100 GRT	DAYS ABSENT		
SCO INDUSTRIAL TRAWL	HOURS FISHING		
NOR IND TRAWLERS (SANDEEL)	Only Catch Data Available		
NOR IND TRAWLERS (N. POUT)	Only Catch Data Available		

Table A.2.2 Catch by gear, NSTF area and quarter (tonnes) in 1989 estimated from STCF data base (less Belgium).

Area	Quarter				total
	1	2	3	4	
1	0	0	0	0	0
2a	0	0	0	0	0
2b	9	1967	451	115	2542
3a	0	0	6	21	27
3b	3336	855	2588	2536	9315
4	12586	9603	12196	10809	45194
5	678	1642	2671	4921	9912
6	83	668	620	604	1975
7a	3359	7974	4676	3984	19993
7b	13980	9910	11939	16850	52679
total	34031	32619	35147	39840	141637

Area	Quarter				total
	1	2	3	4	
1	16212	7380	18755	11034	53381
2a	12057	15152	17999	4298	49506
2b	4309	6064	11712	4574	26659
3a	3952	4918	8436	7656	24962
3b	3826	3925	3693	5598	17042
4	3834	2016	1338	4358	11546
5	1979	2761	2364	4613	11717
6	4623	4906	4196	3203	16928
7a	1707	2789	9251	3130	16877
7b	1242	1884	5224	1833	10183
total	53741	51795	82968	50297	238801

Area	Quarter				total
	1	2	3	4	
1	3490	2379	3132	2205	11206
2a	1553	3940	1637	672	7802
2b	745	3960	3210	1076	8991
3a	894	2717	3028	2358	8997
3b	626	731	1302	1047	3706
4	4028	1707	1716	3397	10848
5	584	902	992	534	3012
6	7538	2601	4395	1118	15652
7a	1212	2227	4111	4950	12500
7b	347	818	1071	262	2498
total	21017	21982	24594	17619	85212

AREA	QUARTER				Total
	1	2	3	4	
1	8500	13383	13254	35539	70676
2a	11	3158	52	11	3232
2b	43849	28254	10108	18248	100459
3a	423	0	0	163	586
3b	460	37371	2920	0	40751
4	145	8649	6122	267	15183
5	639	104749	73247	2380	181015
6	31314	131634	46561	90200	299709
7a	36804	312206	35957	9809	394776
7b	9858	237710	64014	7458	319040
Total	132003	877114	252235	164075	1425427

AREA	QUARTER				Total
	1	2	3	4	
1	734	748	1636	1972	5090
2a	0	0	0	0	0
2b	831	831	2830	2842	7334
3a	0	50	50	50	150
3b	0	0	0	0	0
4	0	0	0	0	0
5	195	576	4733	5417	10921
6	8375	17562	74352	85204	185493
7a	963	1698	7124	7609	17394
7b	1239	1269	1574	1748	5830
Total	12337	22734	92299	104842	232212

Table A.2.2 (cont'd)

f) Seines and ringnet

Area	Quarter				total
	1	2	3	4	
1	18279	7410	14689	12302	52680
2a	1055	809	24197	926	26987
2b	4436	6087	3133	3682	17338
3a	943	2942	6934	1437	12256
3b	35	174	1214	21	1444
4	32	199	70	18	319
5	193	2868	2621	829	6511
6	1036	6053	990	1497	9576
7a	1899	10906	3571	2934	19310
7b	1320	2294	4019	1545	9178
total	29228	39742	61438	25191	155599

g) Pelagic trawls

Area	Quarter				total
	1	2	3	4	
1	102	3400	8180	14806	26488
2a	2	996	4067	3896	8961
2b	0	181	7423	623	8227
3a	0	441	2031	0	2472
3b	92	0	6572	434	7098
4	70	0	28	745	843
5	0	0	0	0	0
6	0	0	0	5	5
7a	0	0	1128	572	1700
7b	0	0	0	2043	2043
total	266	5018	29429	23124	57837

h) Purse seines

Area	Quarter				total
	1	2	3	4	
1	36285	44992	71511	54789	207577
2a	0	1500	13707	10853	26060
2b	1260	1190	7623	19287	29360
3a	0	0	6211	515	6726
3b	0	0	1045	0	1045
4	0	0	0	0	0
5	0	0	0	0	0
6	20286	9436	54808	80521	165051
7a	195	24	13003	6342	19564
7b	0	0	93	0	93
total	58026	57142	168001	172307	455476

i) Gill net

Area	Quarter				total
	1	2	3	4	
1	0	16	6	0	22
2a	7	254	10	0	271
2b	11	54	70	11	146
3a	111	37	76	125	349
3b	267	344	257	292	1160
4	2329	1447	409	511	4696
5	1131	1795	791	623	4340
6	261	513	338	116	1228
7a	1199	878	1049	250	3376
7b	1188	1049	891	155	3283
total	6504	6387	3897	2083	18871

j) Other gears

Area	Quarter				total
	1	2	3	4	
1	60	70	209	3350	3689
2a	89	155	418	814	1476
2b	69	270	311	114	764
3a	361	172	1264	1956	3753
3b	1119	743	338	1081	3281
4	305	164	53	227	749
5	291	1718	1872	3023	6904
6	411	5009	2311	5652	13383
7a	333	501	328	220	1382
7b	225	257	176	105	763
total	3263	9059	7280	16542	36144

Table A.2.3.a Beam trawlers. Analysis of CPUE.

Standardized CPUE in units of		NET BEAM TRAWL >300HP			
area	quarter	1	2	3	4
1					
2a					
2b		5,85	5,65	1,84	1,15
3a				2,52	0,64
3b		3,15	1,65	1,71	1,51
4		2,07	1,33	1,55	2,04
5		0,16	0,60	2,44	3,52
6		1,41	5,17	2,34	3,83
7a		5,84	2,60	2,93	2,73
7b		3,97	2,87	2,68	3,60

Beam Trawlers Total International Effort (excluding Belgium)

Total effort in units of		NET BEAM TRAWL >300HP				
area	quarter	1	2	3	4	total
1						
2a						
2b		2	348	246	100	695
3a				2	33	35
3b		1058	519	1513	1680	4770
4		6093	7197	7853	5308	26451
5		4327	2731	1094	1398	9550
6		59	129	265	158	611
7a		575	3069	1593	1457	6694
7b		3524	3449	4448	4686	16107

ANOVA CPUE Beam Trawlers				
Cause	D.F.	S.Sqs	M.Sqs	F
Model	32	331,66	10,36	11,24
Residual	50	46,11	0,92	
Total	82	377,77	4,61	

Beam Trawlers Fleets Included And Their Relative Efficiency

Fleet Name	Fleet #	Gear #	Effort Unit	Relativity
UK ALL BEAM TRAWLS	1	1	HOURS FISHED	1,00
GER SMALL BEAM TRAWL	9	1	No Data Available	?
FRA COASTAL FLEET-BEAM	38	1	HOURS	3,67
NET BEAM TRAWL <300HP	46	1	DAYS FROM PORT	35,52
NET BEAM TRAWL >300HP	47	1	DAYS FROM PORT	87,18

Table A.2.3.b Otter trawlers. Analysis of CPUE.

Standardized CPUE in units of		Scottish Otter Trawler Hours			
area	quarter	1	2	3	4
1		0,82	0,54	1,01	0,58
2a		0,69	0,69	0,84	0,44
2b		0,62	0,33	1,08	0,48
3a		0,37	0,29	0,62	0,29
3b		0,25	0,25	0,45	0,44
4		0,55	0,36	0,41	0,29
5		0,63	0,43	0,93	0,27
6		0,85	0,57	0,44	0,71
7a		0,45	0,41	0,82	0,78
7b		0,49	0,33	0,45	0,42

Otter Trawlers Total International Effort

Total effort in units of		Scottish Otter Trawler Hours				
area	quarter	1	2	3	4	total
1		19868	13675	18620	18996	71158
2a		17553	22055	21511	9742	70860
2b		6971	18614	10831	9601	46016
3a		10583	16812	13629	25958	66982
3b		15571	15608	8143	12798	52120
4		6975	5593	3285	15207	31061
5		3155	6473	2543	17066	29237
6		5409	8722	9505	4527	28162
7a		3820	6823	11235	4001	25879
7b		2516	5746	11670	4335	24266

ANOVA	CPUE	Otter Trawlers		
Cause	D.F.	S.Sqs	M.Sqs	F
Model	48	546,99	11,40	18,59
Residual	183	112,15	0,61	
Total	231	659,14	2,85	

Otter Trawlers Fleets Included And Their Relative Efficiency

Fleet Name	Fleet #	Gear #	Effort Unit	Relativity
ENG OTTER TRAWL		5	2 HOURS FISHED	0,02
GER OTTER TRAWL (L.D. CUTTER)		6	2 No Data Available	?
GER OTTER TRAWL (L.D. TRAWLER)		7	2 No Data Available	?
GER OTTER TRAWL (S.D. CUTTER)		8	2 No Data Available	?
DEN CON. TRAWL SIN. < 100 GRT		12	2 DAYS ABSENT	0,71
DEN CON. TRAWL SIN. > 100 GRT		13	2 DAYS ABSENT	1,64
SCO LIGHT TRAWL		23	2 HOURS FISHING	0,11
SCO NEPHROPS TRAWL		24	2 HOURS FISHING	0,07
SCO TRAWL		29	2 HOURS FISHING	0,48
NOR SAIHE TRAWLERS (OTTER)		37	2 No Data Available	?
FRA COASTAL FLEET-BOTTOM		39	2 HOURS	0,21
FRA FREEZERS+BIG TRAWL-BOTTOM		42	2 HOURS	0,52
FRA HIGH SEA FLEET-BOTTOM		43	2 HOURS	0,69
NET OTTER TRAWLERS		48	2 DAYS FROM PORT	1,00

Table A.2.3.c Pair trawlers. Analysis of CPUE.

Standardized CPUE in units of		SCO PAIR TRAWL DEMERSAL			
area	quarter	1	2	3	4
1		0,32	0,29	0,48	0,46
2a		0,19	0,12	0,30	0,26
2b		0,31	0,21	0,26	0,58
3a		0,38	0,26	0,17	0,30
3b		0,26	0,13	0,14	0,32
4		0,20	0,13	0,13	0,08
5		0,19	0,17	0,27	0,14
6		0,23	0,24	0,35	0,17
7a		0,13	0,14	0,26	0,40
7b		0,11	0,21	0,11	0,14

Pair Trawlers Total International Effort

Total effort in units of		SCO PAIR TRAWL DEMERSAL				
area	quarter	1	2	3	4	total
1		10971	8227	6553	4787	30538
2a		8307	32425	5493	2560	48785
2b		2371	18726	12162	1868	35127
3a		2367	10513	17745	7795	38420
3b		2422	5489	9171	3303	20385
4		20461	13172	13270	43472	90375
5		3030	5432	3720	3928	16110
6		32953	10727	12583	6436	62699
7a		9099	15683	15525	12456	52763
7b		3087	3850	9635	1873	18445

ANOVA	CPUE	Pair Trawlers		
Cause	D.F.	S.Sqs	M.Sqs	F
Model	43	580,01	13,49	17,77
Residual	85	64,50	0,76	
Total	128	644,51	5,04	

Pair Trawlers Fleets Included And Their Relative Efficiency

Fleet Name	Fleet #	Gear #	Effort Unit	Relativity
ENG PAIR TRAWLS		3	3 HOURS FISHED	0,01
DEN PAIR < 100 GRT		10	3 DAYS ABSENT	0,72
DEN PAIR > 100 GRT		11	3 DAYS ABSENT	2,24
SCO PAIR TRAWL DEMERSAL		26	3 HOURS FISHING	0,09
NET PAIR TRAWLERS		49	3 DAYS FROM PORT	1,00

Table A.2.3.d Industrial otter trawl. Analysis of CPUE.

Standardized CPUE in units of		DEN IND. TRAWL > 100 GRT			
area	quarter	1	2	3	4
1		10,64	32,00	25,67	58,54
2a		3,47	18,07	2,74	11,00
2b		26,57	31,34	23,91	17,20
3a		30,21			16,30
3b		41,82	60,55	27,29	
4		26,59	40,61	32,17	20,54
5		14,28	33,56	33,23	16,03
6		17,50	30,79	22,91	15,70
7a		22,90	40,96	33,48	17,65
7b		15,76	30,35	31,45	14,84

Ind. Otter Trawl Total International Effort

Total effort in units of		DEN IND. TRAWL > 100 GRT				
area	quarter	1	2	3	4	total
1		799	418	516	607	2340
2a		3	175	19	1	198
2b		1650	902	423	1061	4036
3a		14			10	24
3b		11	617	107		735
4		5	213	190	13	422
5		45	3121	2204	149	5519
6		1789	4275	2032	5745	13841
7a		1607	7622	1074	556	10859
7b		625	7833	2035	502	10996

ANOVA	CPUE	Ind. Otter Trawl		
Cause	D.F.	S.Sqs	M.Sqs	F
Model	38	41,57	1,09	5,07
Residual	23	4,97	0,22	
Total	61	46,54	0,76	

Ind. Otter Trawl Fleets Included And Their Relative Efficiency

Fleet Name	Fleet #	Gear #	Effort Unit	Relativity
DEN IND. TRAWL < 100 GRT	18	4	DAYS ABSENT	3,02
DEN IND. TRAWL > 100 GRT	19	4	DAYS ABSENT	5,68
SCO INDUSTRIAL TRAWL	22	4	HOURS FISHING	1,00
NOR IND TRAWL (SANDEEL)	34	4	No Data Available	?
NOR IND TRAWL (N. POUT)	35	4	No Data Available	?

Table A.2.3.e Industrial pair trawlers. Analysis of CPUE.

Standardized CPUE in units of		DEN IND. PAIR > 100 GRT			
area	quarter	1	2	3	4
1		30,58	14,00	24,67	24,00
2a					
2b		22,86		40,80	12,00
3a			19,07		
3b					
4					
5		7,25	26,03	28,07	18,85
6		21,50	17,52	28,27	23,23
7a		29,18	24,04	36,94	18,13
7b		22,53	28,60	27,08	10,88

Ind.Pair Trawlers Total International Effort

Total effort in units of		DEN IND. PAIR > 100 GRT				
area	quarter	1	2	3	4	total
1		24	1	36	14	75
2a						
2b		36		49	1	86
3a			3			3
3b						
4						
5		27	15	148	36	226
6		389	524	2009	467	3390
7a		33	31	147	27	237
7b		55	1	11	16	83

ANOVA	CPUE	Ind.Pair Trawlers		
Cause	D.F.	S.Sqs	M.Sqs	F
Model	24	8,88	0,37	1,79
Residual	12	2,48	0,21	
Total	36	11,36	0,32	

Ind.Pair Trawlers Fleets Included And Their Relative Efficiency

Fleet Name	Fleet #	Gear #	Effort Unit	Relativity
DEN IND. PAIR < 100 GRT	16	4	DAYS ABSENT	0,52
DEN IND. PAIR > 100 GRT	17	4	DAYS ABSENT	1,00

Table A.2.3.f Seines and ringnets. Analysis of CPUE.

Standardized CPUE in units of		SCO SEINE NET			
area	quarter	1	2	3	4
1		0,40	0,24	0,25	0,28
2a		0,30	0,17	6,17	0,19
2b		0,97	0,23	0,16	0,27
3a		0,15	0,28	0,39	0,19
3b		0,17	0,32	0,17	0,21
4		0,21	0,45	0,50	0,24
5		0,19	0,43	0,33	0,20
6		0,23	0,28	0,31	0,31
7a		0,25	0,24	0,13	0,29
7b		0,23	0,26	0,29	0,30

Seines and Ringnets Total International Effort

Total effort in units of		SCO SEINE NET				
area	quarter	1	2	3	4	total
1		46085	31073	58262	44202	179622
2a		3521	4773	3922	4817	17033
2b		4570	25955	19980	13686	64191
3a		6475	10564	17600	7719	42358
3b		209	540	7208	101	8059
4		151	443	140	74	808
5		1012	6684	8057	4193	19948
6		4516	21758	3210	4813	34297
7a		7605	45576	27534	10157	90872
7b		5626	8682	13639	5197	33144

ANOVA CPUE Seines and Ringnets				
Cause	D.F.	S.Sqs	M.Sqs	F
Model	41	187,00	4,56	16,49
Residual	41	11,34	0,28	
Total	82	198,34	2,42	

Seines and Ringnets Fleets Included And Their Relative Efficiency

Fleet Name	Fleet #	Gear #	Effort Unit	Relativity
UK ALL SEINES AND RING NET	4	5	HOURS FISHED	0,12
DEN DANISH SEINE	14	5	DAYS ABSENT	3,78
SCO SEINE NET	28	5	HOURS FISHING	1,00

Table A.2.3.g Pelagic trawlers. Analysis of CPUE.

Standardized CPUE in units of		French HSF. Pelagic Trawlers hours			
area	quarter	1	2	3	4
1				55.27500	84.36000
2a				70.24400	74.74000
2b				67.58000	
3a			154.14400	88.62000	
3b		4.24200		118.75000	11.42400
4		2.25200		1.76300	8.62000
5					
6					
7a				67.26700	27.59900
7b					20.00100

Pelagic Trawlers Total International Effort

Total effort in units		French HSF. Pelagic Trawlers hours				
area	quarter	1	2	3	4	total
1		0	0	16.6078	12.6126	29.2204
2a		0	0	15.8591	1.1239	16.983
2b		0	0	5.1198	0	5.1198
3a		0	1.4986	3.7463	0	5.2449
3b		21.6893	0	10.9895	37.9895	70.6683
4		31.0832	0	15.8816	86.4268	133.3916
5		0	0	0	0	0
6		0	0	0	0	0
7a		0	0	0.6244	20.7255	21.3499
7b		0	0	0	77.0963	77.0963

ANOVA	CPUE	Pelagic Trawlers		
Cause	D.F.	S.Sqs	M.Sqs	F
Model	17.00	40.13	2.36	20.12
Residual	1.00	0.12	0.12	
Total	18.00	40.25	2.24	

Pelagic Trawlers Fleets Included And Their Relative Efficiency

Fleet Name	Fleet #	Gear #	Effort Unit	Relativity
FRA COASTAL FLEET-PELAGIC	40	6	HOURS	0.1030
FRA FREEZERS + BIG TRAWL PELAGIC	44	6	HOURS	0.1249
FRA HIGH SEA FLEET - PELAGIC	45	6	HOURS	1.0000

Table A.2.3.h Purse seines. Analysis of CPUE.

Standardized CPUE in units of		Danish PurseSeiner Days Absent			
area	quarter	1	2	3	4
1		57.73300	25.10000	6.46300	54.63200
2a			57.73300	12.13900	33.58100
2b		21.33400	112.50600	50.45300	99.00600
3a				67.48600	73.72300
3b				79.32700	
4					
5					
6		62.03900	17.23100	54.51500	44.79600
7a		97.49600	23.99900	84.84200	57.73300
7b				57.73300	

Purse Seines Total International Effort

Total effort in units		Danish PurseSeiner Days Absent				
area	quarter	1	2	3	4	total
1		628.5021	1792.5419	11064.8027	1002.8804	14488.7271
2a		0	25.9819	1129.168	323.1868	1478.3367
2b		59.0617	10.5772	151.091	194.8062	415.5361
3a		0	0	92.034	6.9856	99.0196
3b		0	0	13.1734	0	13.1734
4		0	0	0	0	0
5		0	0	0	0	0
6		326.9857	547.6308	1005.3774	1797.5044	3677.4983
7a		2.0001	1.0001	153.2606	109.8514	266.1122
7b		0	0	1.6109	0	1.6109

ANOVA	CPUE	Purse Seines		
Cause	D.F.	S.Sqs	M.Sqs	F
Model	21.00	32.17	1.53	0.61
Residual	9.00	22.42	2.49	
Total	30.00	54.59	1.82	

Purse Seines Fleets Included, Effort Units And Their Relative Efficiency

Fleet Name	Fleet #	Gear #	Effort Unit	Relativity
DEN PURSE SEINE	21	7	DAYS ABSENT	2.0206
SCO PURSE SEINE (HERRING)	27	7	HOURS FISHING	0.3973
SCO PURSE SEINE MACKEREL	32	7	HOURS FISHING	1.0000
NOR PURSE SEINERS	36	7	No Data Available	?

Table A.2.3.i Gill net. Analysis of CPUE.

Standardized CPUE in units of		Danish Gill Net Days Absent			
area	quarter	1.0000	2.0000	3.0000	4.0000
1		0.00	2.29	0.50	0.00
2a		1.75	1.90	1.67	0.00
2b		2.75	0.56	1.34	2.13
3a		0.80	0.93	1.26	0.80
3b		1.29	0.97	1.98	1.17
4		1.86	0.93	2.45	1.64
5		1.63	0.99	1.05	0.85
6		1.32	1.25	1.26	0.96
7a		1.43	0.81	1.08	0.99
7b		4.27	1.00	1.81	1.17

Gill Net Total International Effort

Total effort in units		Danish Gill Net Days Absent				total
area	quarter	1	2	3	4	
1	1	0	7	12	0	19
	2a	4	134	6	0	144
	2b	4	97	52	5	158
	3a	139	40	61	156	396
	3b	207	354	129	250	941
	4	1254	1557	167	311	3289
	5	696	1809	750	735	3990
	6	198	409	269	121	997
	7a	841	1081	969	252	3143
	7b	278	1048	492	133	1952

ANOVA		CPUE	Gill Net		
Cause	D.F.	S.Sqs	M.Sqs	F	
Model	38	302.08	7.95	22.26	
Residual	18	6.43	0.36		
Total	56	308.51	5.51		

Gill Net Fleets included, Their effort Units and their Relative Efficiency

Fleet Name	Gear #	Effort Unit	Relativity
Denmark gill net		Days	1.0000
England gill net		Hours	0.0016
Scotland Gill net		Days	0.2114
Netherlands gill net		Days ?	0.6361

ACFM ADVICE

1 THE FORM OF ACFM ADVICE**1.1 The Objectives and Tasks of ACFM**

The terms of reference for ACFM are established by the Commissions and ICES member governments which seek its advice. While the precise questions asked vary with the managers' requirements, the tasks of ACFM can be summarized as follows: For all the major fish, shellfish and in some cases, marine mammal resources in the ICES area:

- a) To assess the historic development in terms of size and structure of the stocks.
- b) To advise on the expected impact of various management measures, and where appropriate to make recommendations on management action required.

Setting objectives of fisheries management is within the province of the managers. The role of ACFM is to provide the biological information and advice necessary for managers to achieve the objectives they choose. In some cases, however, ACFM may be in a position to comment on the implications of setting certain objectives and on the feasibility of achieving them. Some objectives, such as achieving stability of catches and maximising long term yield, or achieving stability of both effort and yield, for example, may be mutually incompatible. In appropriate cases, therefore, ACFM may point out the biological constraints that need to be taken into account by managers in setting management objectives.

While understanding that it is not the role of ACFM to set objectives for fishery managers, ACFM nevertheless has a set of objectives of its own to enable it to formulate its advice according to consistent criteria. The objective of ACFM is thus:

TO PROVIDE THE ADVICE NECESSARY TO MAINTAIN VIABLE FISHERIES WITHIN SUSTAINABLE ECOSYSTEMS¹.

1.2 Stock Assessments and Forecasts**1.2.1 Sources of information**

To assess the state of stocks and to make forecasts, ICES is dependent on the provision of reliable data both from the fisheries and from research establishments. To carry out its work, ICES has established a number of working groups and the advice given by ACFM represents a distillation of the assessments provided by these groups.

1.2.2 The aims of stock assessment

Stock assessment aims at understanding the dynamics of exploited resources and involves the estimation of a variety of population parameters, in particular mortality rates due to fishing and other causes, numbers at age (including recruitment) and spawning stock biomass.

Stock assessments in which these parameters can be estimated are described as "analytical assessments".

While analytical assessments are attempted on many stocks, this is not always possible for a number of reasons. In some stocks the reliability of the catch data is inadequate. In others further research is needed on the biology

¹An ecosystem is defined as sustainable if management actions do not result in irretrievable loss of any component of the system.

of the species concerned before reliable assessments can be made. In many of these cases, however, it may be possible to make general statements about the state of exploitation of the stock. In more extreme cases, the only information available may be the reported annual national landings.

In all cases ACFM tries to give some indication of the reliability of the assessments and forecasts. Where possible ACFM also takes uncertainty into account by providing indications of the biological risk associated with particular management options.

ACFM cannot stress too forcibly that the reliability of its assessments and advice depends on the quality of the data provided. In particular, the reliability of the reported national catch statistics is a matter of great concern because the quantitative stock parameters are a direct function of the number of fish that are estimated to have been caught.

1.2.3 The state of stocks

Estimates of the present size of each stock and the level of fishing mortality on it are meaningful only in the context of historical trends and assessments of the long term potential of the stock. To provide indicators of the state of each stock in relation to its long term potential and of the level of exploitation, ACFM has established a series of "Biological Reference Points".

1.3 Biological Reference Points (BRP)

Biological reference points represent a set of signposts to enable scientists and managers to judge the state of exploitation of the stocks. They are of two types: BRPs defined by values of fishing mortality rate on the yield-per-recruit curve; BRPs based on the sustainability of recruitment at different levels of fishing mortality.

1.3.1 BRPs defined by values of fishing mortality rate (F) on the yield-per-recruit curve

Regardless of recruitment fluctuations, the total yield that can be obtained from each year class throughout its existence in the fishery is dependent on the fishing mortality rate and on the exploitation pattern (the pattern of relative values of fishing mortality on each age group). Yield per recruit is also constrained by the growth and natural mortality rates of the stock.

The fishing mortality rate provides a measure of the proportion of the stock that is killed by fishing each year. It is controlled directly by the amount of fishing, i.e., the fishing effort. Fishing effort is a measure derived from the appropriate combination of the number of vessels, the time spent fishing and the size and type of gear.

Stocks that are exploited at levels of fishing mortality in excess of those giving the maximum yield on the yield-per-recruit curve, or with exploitation patterns that give lower than maximum yield-per-recruit, are said to be subject to "growth overfishing".

- i) F_{max} is the level of fishing mortality at which the maximum long term average annual catch can be taken.
- ii) $F_{0.1}$ is an arbitrary level of fishing mortality at which the slope of the yield-per-recruit curve is one tenth of its slope at the origin. $F_{0.1}$ is always less than F_{max} . At $F_{0.1}$ the yield-per-recruit is only slightly less than at F_{max} whereas the implied difference in fishing mortality rate (and hence fishing effort) is much greater. The implication of this is that catch per unit effort is higher at $F_{0.1}$ than at F_{max} with consequent economic benefits, but without foregoing any significant part of the potential yield.

The actual shape of the yield-per-recruit curve and the values of F_{max} and $F_{0.1}$ depend on the exploitation pattern, and any changes in the latter due to mesh size changes, for example, result in new values of F_{max} and $F_{0.1}$.

The level of fishing mortality and the exploitation pattern determine the proportion of recruits that survive to spawn as well as the yield-per-recruit.

1.3.2 BRPs based on the sustainability of recruitment at different levels of fishing mortality

Year-class strength, i.e., the level of recruitment each year, is dependent on the environmental and ecological conditions experienced by young fish between the time they are spawned and when they become available to the fishery. Since these conditions are variable, recruitment is also variable, and more so in some stocks than others. In many stocks, recruitment does not appear to be simply proportional to the size of the parent stock from which it was spawned. There must, nevertheless, be a level of spawning stock below which recruitment will be affected as a result of low egg production. The likelihood that the stock will fall below this level depends on both environmental and ecological factors that affect the survival of the young stages and on the fishing mortality rate both on adults and on juveniles. Stocks that have been reduced below this level by fishing are said to be suffering from "recruitment overfishing".

The BRPs given below are the levels of fishing mortality at which it is almost certain (F_{low}), probable (F_{med}) and doubtful (F_{high}), based on historic evidence, that recruitment will in the long term be sufficient to compensate the losses due to mortality, while the stock remains within its previous limits. They are calculated from the historic series of estimates of spawning stock and recruitment and from estimates of the proportion of recruits that survive to contribute to the spawning stock at different levels of fishing mortality. For this reason, they are likely to be meaningful only for those stocks for which there is a reasonably long series of stock and recruitment estimates covering a range of spawning stock sizes.

- i) F_{high} corresponds to a level of fishing mortality at which survival is so low that recruitment (per unit spawning stock biomass) has in the past been insufficient to compensate the losses due to mortality in about nine years in ten. It is thus a level that is almost certainly not sustainable if the stock is to be maintained within its previous limits.
- ii) F_{med} is the level of fishing mortality at which recruitment (per unit spawning stock biomass) has been sufficient to compensate the losses due to mortality in half the years observed. This suggests that it may be a level that can be sustained assuming that the underlying environmental and ecological factors to which the data relate do not change.
- iii) F_{low} is the level of fishing mortality at which recruitment (per unit spawning stock biomass) has been sufficient to compensate the losses due to mortality in about nine years in ten. It is thus a level that is likely to be sustainable and may facilitate a growth in stock size where stocks are at a low level.

THESE BIOLOGICAL REFERENCE POINTS ARE NOT THE SAME AS MANAGEMENT OBJECTIVES. They simply serve as a guide to aid managers in choosing from the range of options open to them.

1.4 Stock Categories

ACFM has in the past defined a series of stock categories based on the state of the stocks in question. These have provided a basis for deciding the type of advice appropriate for each stock. In the light of the above, ACFM now considers that it is appropriate to redefine these categories.

As explained in Section 1.1.3.2 above, there is likely to be a level of spawning stock size below which the probability of poor recruitment increases as spawning stock size decreases. Because of the inherent variability of recruitment at any given spawning stock size, this level is not known for certain in many stocks. Nevertheless, some idea of the bounds within which it may lie can be obtained by examining the historic variation in recruitment at different levels of spawning stock. For present purposes, this level is named the "minimum biologically acceptable level" (MBAL).

This level may be useful in providing managers with an indication of a lower level of spawning stock above which the stock should, in so far as possible, be maintained.

In stocks for which there is adequate information on historic stock and recruitment, the MBAL can be defined by the level of spawning stock below which the data indicate that the probability of poor recruitment increases as spawning stock size decreases. Any action that was expected to reduce spawning stock size below this level would, therefore, be outside safe biological limits.

It should be noted that stocks below the MBAL are not necessarily in danger of imminent collapse, but simply that recruitment is expected to be lower than at higher levels of spawning stock size. Because of the fact that there must be a direct proportionality between recruitment and spawning stock at low levels of spawning stock, however, any further decrease in spawning stock size may result in an accelerating progression towards collapse.

In stocks for which information is limited, it is safest to assume that the MBAL is equal to the lowest level of spawning stock so far recorded. Even though there may be no evidence that recruitment is depressed at this level of spawning stock, it must be presumed that the MBAL of spawning stock defined above may lie at any level below that so far observed. Stocks below this level are in "unknown territory", and, therefore, any action resulting in such a condition would be outside safe biological limits, even though the level of spawning stock biomass at which the probability of poor recruitment increases as spawning stock decreases is not known.

On the basis of the above considerations, ACFM recognizes two categories of stocks: those that are below the MBAL (or expected to become so in the near future at present levels of exploitation) and those that are not in immediate danger of falling below this level. In addition, there are stocks for which the data are inadequate to define the appropriate category.

Stocks which are below the MBAL or expected to become so in the near future at present levels of exploitation

In identifying stocks that are in this category, it has to be accepted that there will usually be a considerable degree of uncertainty. This can arise for one of three reasons:

- a) Imprecision in the historic estimates of stock size and recruitment due to imprecise data.
- b) Variability in recruitment due to natural causes can obscure any relationship between recruitment and parent stock; thus, low stock size, which is itself often the result of poor recruitment, can be caused by environmental and ecological factors as well as by overfishing.
- c) The time series of stock and recruitment data may not be long enough to determine the likely range of variation in the size of future year classes at different levels of spawning stock size.

In stocks in this category, ACFM will in so far as possible give advice on what measures are needed to rectify the situation. The severity of this advice and the extent to which management options are possible, will normally depend on the degree of depletion of the stock and on what information is available on the historic series of stock and recruitment.

Stocks that are not in imminent danger of falling below the MBAL

This category includes stocks that are currently being exploited over a wide range of fishing mortalities, either above or below the biological reference points on the yield-per-recruit curve, but which are not thought to be in imminent danger of falling below the MBAL. In these stocks, ACFM normally provides a range of options together with impact statements for each option. Where possible, ACFM also provides an indication of the risks of fishing at different levels of fishing mortality in terms of the probability of certain events, e.g., a decrease in stock below a certain level. In addition, in stocks that have historically been at higher levels, ACFM may indicate measures that are expected to allow further recovery with the objective of creating a buffer stock as a hedge against future recruitment fluctuations.

Where fishing mortality is above F_{max} and $F_{0.1}$, the question of whether it should be reduced may be primarily an economic one. ACFM will, if asked or where appropriate, give indications on how longer-term benefits can be

obtained in these cases, e.g., by effort reductions. However, fishing mortalities eventually expected to reduce the spawning stock below or close to the MBAL will not normally be included as management options.

Where the fishing mortality rate is below or close to F_{\max} or $F_{0.1}$ and the stock is above the level at which recruitment is likely to be affected, managers may have a wide range of options to consider. In these cases, ACFM would simply point out that nothing is to be gained in terms of yield-per-recruit by allowing fishing mortality to increase above the F_{\max} or $F_{0.1}$ levels.

ACFM also considers that growth overfishing represents a sub-optimal use of resources, but accepts that this is partly an economic judgement outside its present area of competence. Similarly, ACFM considers that it has a responsibility to indicate measures that prevent a waste of resources (e.g., through discarding) or that minimize any harmful side effects of fishing.

Stocks whose state of exploitation cannot be precisely assessed

This category contains two different types of stock:

- a) Those for which no information exists other than annual landings.
- b) Those for which the state of exploitation is known at least in qualitative terms, but where no stock size or recruitment estimates are available to make forecasts.

In the first of these sub-categories ACFM provides information on the recent trends in landings and, where requested, advises on appropriate catch levels designed to prevent an increase in fishing mortality rate. The purpose of catch limits in these stocks may be purely a management one to remove opportunities to misreport catches made in other areas. In general, a consistent baseline period will be used in order to prevent a drift to higher levels of exploitation. As a result, no annual update will be made for these stocks, unless additional information becomes available.

In the second sub-category, ACFM may be able to advise on the likely losses and gains that will result from changes in fishing effort and/or exploitation pattern. In cases where recruitment indices are available, it may also be able to provide catch options corresponding to *status quo* fishing mortality based on simplified forecasting methods.

1.5 The Multispecies Dimension

In providing advice to managers, ACFM wherever possible considers the interactions between the fisheries, and between the stocks of fish, shellfish and other marine organisms. There are two ways in which interactions can take place: technical interactions and ecological interactions.

1.5.1 Technical interactions

The majority of fisheries take a mixture of stocks of different species living on the same grounds. Fishing targeted at one stock thus has an impact on other stocks in the same area. These technical interactions present a particular problem in unaimed fisheries, i.e., those exploiting a mixture of stocks. Where possible ACFM takes these interactions into account when pointing out the constraints to managers. Progress in this developing process is expected to be enhanced by the setting up of area-based, as distinct from stock-based assessment working groups. The provision to ICES of new area-based information is also likely to improve the possibilities of giving integrated advice.

1.5.2 Ecological interactions

Fish and shellfish stocks are interacting components of inherently complex ecosystems. While considerable progress has been made in recent years in collecting information on the nature of the interactions between fish stocks (competition and predation), and other components of the ecosystem, the state of understanding is as yet inadequate for operational use. Wherever reasonably certain conclusions can be drawn, however, ACFM takes ecological interactions into account in its advice.

THE APPLICATION OF ACMP ADVICE IN A MANAGEMENT CONTEXT

In accordance with the intention expressed by ACMP in its 1988 report, this article concludes the current phase of ACMP discussions on managerial issues.

Summary

It is the opinion of ACMP that a consensus view on a conceptual framework for environmental protection is necessary for its own work, and should also be of value to the recipients of ACMP advice. The article which follows gives a detailed explanation of the key elements of this framework. It follows that the ACMP would strongly encourage open dialogue on this topic with a view to stimulating the gradual introduction of control strategies, applicable to all anthropogenic influences on the marine environment, that would facilitate the application of its advice within the management context intended; that is, management of the marine environment in a manner which will afford protection to the environment as a whole.

It is most improbable that strategies for assessing and controlling human impact on the environment will be effective if they permit the development of an unrestricted array of new practices, products, and waste materials. There are already indications that statutory regulatory mechanisms are as vulnerable to "overloading" as the environment itself. Furthermore, it is obvious that environmental quality which will allow sustained development can not be achieved through case-by-case assessments alone, or through control strategies that are subject to constant change. The need for re-appraisal of existing strategies, and agreement on the future role of science in the environmental protection process, is urgent.

For the above reasons, the ACMP has outlined a management framework within which justification of a practice becomes the primary requirement and which clearly identifies the contribution of science to subsequent elements of the framework.

A justified practice will be one for which the combined benefits to the whole of society are considered to outweigh the combined deficits or detriment, environmental effects being only part of the latter. The second element of the framework would place scientifically derived limits on the environmental changes that are permitted to occur; in the case of contaminants emitted by a practice, these limits would constitute upper bounds to increased levels of exposure for Man and critical components of the environment. The third element of the framework, in certain respects the most critical, demands actions that will minimize environmental changes resulting from the practice as far below the limits as practical, taking into account technological capabilities as well as economic, social, and political considerations. This is the on-going process of optimization of human activities to ensure their compatibility with sustained use of the environment as advocated by the World Commission on Environment and Development (1987) and others.

Taken together, the elements of the framework comprise an integrated approach to environmental management that provides for equate control of a practice throughout its lifespan. An inherent feature of the framework is that it allows the application of a variety of regulatory tools, such as, for example, emission controls an use-related environmental quality objectives, provided they are developed and applied in accordance with the basic principles of the framework.

A simplistic representation of the strategy underlying the framework is shown in Figure 9. It indicates that the "acceptability" of a practice will depend not only on its yielding a "net benefit" but also on the possibilities for optimization of the practice. The practical application of the framework is illustrated in Figure 10. This shows the interrelationships between the elements of the framework as well as the relevant considerations of the associated control process.

Finally, in presenting these proposals, the ACMP wishes to stress its belief that, in parallel with scientific activities aimed at improved understanding of existing environmental problems, there must be greater efforts to develop new and improved environmental management systems that will reduce the potential for future problems. This article is ACMPs contribution to this process.

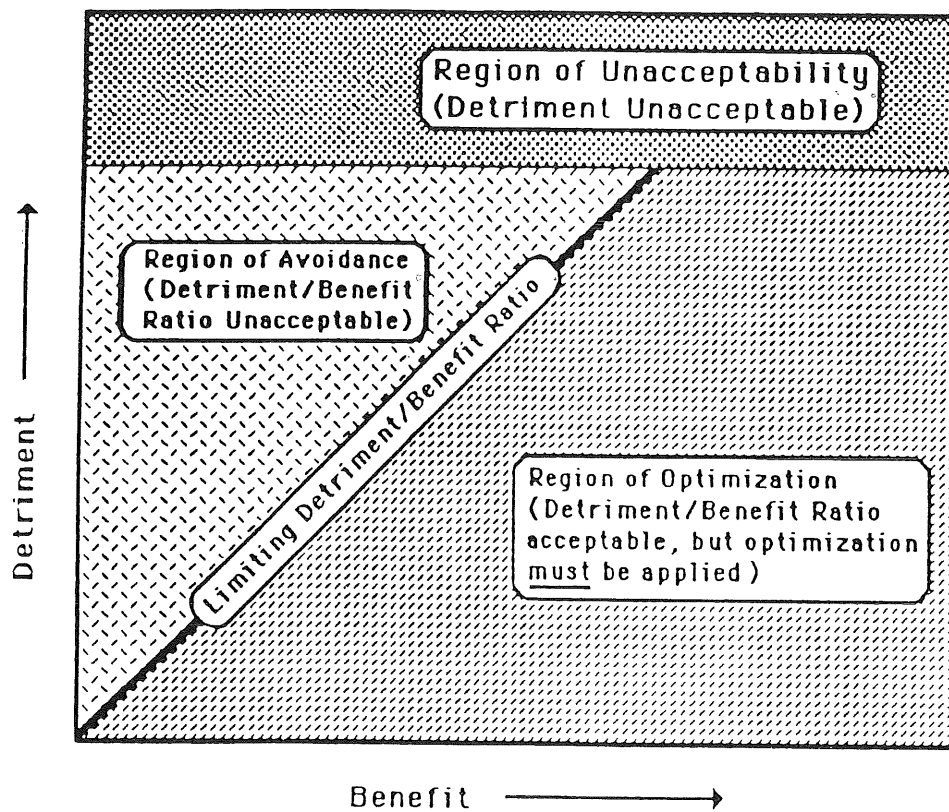


Figure 9 Simplistic Representation of the Environmental Protection Strategy Advocated

