Tøl. 41 Assess/Env

REPORT OF THE
WORKING GROUP ON ECOSYSTEM EFFECTS OF FISHING ACTIVITIES

ICES Headquarters
13–21 March 1996

TABLE OF CONTENTS

1 OPENING OF THE MEETING

The meeting of the Working Group on Ecosystem Effects of Fishing Activities (WGECO) was opened at 10.00 hrs on 13 March 1996 by the Chairman, Dr S. Hall. Dr R.S. Bailey, the Fishery Secretary, welcomed the participants on behalf of the ICES Secretariat in Copenhagen, Denmark.

2 ADOPTION OF THE AGENDA

The terms of reference (C.Res.1995/2:14) for the meeting are listed below.

The Working Group on Ecosystem Effects of Fishing Activities (Chairman: Dr S. Hall, UK) will meet at ICES Headquarters from 13–21 March 1996 to:

a) analyse survey data in terms of appropriate summary parameters for species assemblages, with a view to continuing the study of biodiversity and changes in community structure;

b) examine trends in abundance of non-target and non-assessed fish and shellfish species, taking into account other ICES reports on non-assessed species;

c) refine and implement approaches for quantifying the susceptibility of species to fishing effects;

d) analyse available data on system responses in areas where fishing has been reduced;

e) summarise any new work undertaken on the estimation of by-catches and discards and their exploitation by scavengers;

f) prepare work as required for the Intermediate Ministerial Meeting on Fisheries and the Environment (1997) and ultimately for the Fifth International Conference on the Protection of the North Sea, and address the implications of applying the precautionary principle to fisheries activity;

g) review relevant activities within the entire ICES area with a view to extending the activities of the Working Group to ecosystems outside the North Sea.

The terms of reference set for this meeting were wide ranging and raised issues which required considerable deliberation. While every effort was made to give a full treatment to all terms of reference, the workload given to the group made this impossible. In particular, Sections 8 and 9 contain relatively superficial treatments of the issues the terms of reference raise (items f and g above). Despite these shortcomings, the Working Group feels that, taken as a whole, the work carried out continues to address key issues regarding the effects of fisheries and, most importantly, highlights areas where existing data and theory are inadequate.

One general issue on which there was considerable discussion, in view of the heavy workload, was that of data sources. The material used to arrive at the conclusions reached ranged from published papers in the refereed literature, through the various shades of grey literature, to reports of work currently in progress. In no case was it possible to explicitly and comprehensively review all of the relevent material, and deciding on the weight that should be given to the various sources remains problematic. However, in the final analysis the report represents the collective judgement of the WGECO and it is hoped that, in reaching a consensus, the group distilled the available information appropriately. In judging the report, however, it should be borne in mind that much more detailed treatments of all the topics covered here are possible.

The draft agenda was accepted without amendment, and is appended as Annex 1. The list of participants is attached as Annex 2.

The Chairman emphasized the importance of finalizing the report by the end of the meeting. This was necessary to meet deadlines imposed by the North Sea Secretariat.

3 COMMUNITIES AND ASSEMBLAGES

Analyse survey data in terms of appropriate summary parameters for species assemblages, with a view to continuing the study of biodiversity and changes in community structure.

3.1 Introduction

This term of reference calls for analyses to address at least two aspects of the effects of fishing on exploited fish communities and assemblages. The first aspect is how the structure and function of a community changes as fisheries alter the absolute and relative abundance of fish species through harvest (either directed or bycatch). The second aspect is how the structure and function of the exploited community may change further, as predation and competition among the species respond to the altered levels of abundance of the harvested species. These species interactions might compensate for or amplify the initial changes in abundance directly caused by fishing. The effects arising from species interactions may be present (and even largest) at sizes not directly affected by fishing removals.

In the context of evaluating ecosystem effects of fishing, it is not necessary to subscribe to any specific theoretical framework of community dynamics. Rather, a rigorous pairing of scientific hypothesis testing and pattern extraction from good data sets can help to untangle how the assemblage and its members are affected by and respond to fishing. However, both parts of the task require metrics which reflect informative properties of the community. Again, community ecologists have

proposed many such metrics (Peet, 1974; Pielou, 1975), and there is widespread disagreement about what each does and does not capture about community structure and function (Magurran, 1988). In past meetings this Working Group has explored the application of some of these metrics to fish survey data. It found in particular that the distribution of numbers and diversity across sizes held some promise as a useful metric for reflecting effects of fishing on the statistical patterns in the survey data, and by inference on the community. At this meeting the group reviewed theoretical work on community metrics and on how the biomass and diversity size spectra should be affected by fishing, and conducted further analyses. The analyses were planned to firm up weaknesses in analyses at past meetings, to apply the size-dependent metrics to more data sets, and to explore what other integrating metrics might reflect about community structure and the effects of fishing on that structure.

Notes. The use of terms like index, parameter, statistic, measure, etc., is not consistent in the larger scientific community. For this section we adopt the word "metric" to be an inclusive noun, applying to all these concepts, for use in the context of describing single variables which capture information about the community. We also use the terms "community" and "assemblage" interchangeably within this section, and each in a very generic sense. However, that "generic sense" is limited in an important way. All the analyses in this Section are of data from fish surveys. We use no data on, and make no inferences about, any part of the North Sea marine ecosystem except the fish sampled by the IBTS, EGFS or Beam Trawl surveys. In many places throughout this section we will be talking about "small", "intermediate", or "large" fish or length classes. Generally the adjectives will be used in a conversational sense, and the specific boundaries will not be important to the points being made. Where the specific sizes matter, we try to specify exactly the length intervals referred to.

3.2 Theory of Community and Assemblage Dynamics

3.2.1 Review of literature in community ecology and community metrics

Diversity metrics

Describing the structure of communities and defining the processes upon which they are organised has been a focus of much ecological work. Species diversity has long been considered to be an important property of community structure (Huston, 1994; Rosenzweig, 1995). Numerous indices of species diversity have been introduced, but with little agreement so far as to which is best (for reviews, see May, 1975; Pielou, 1975; Washington, 1984; Magurran, 1991). Part of the problem is that most diversity indices encompass two distinct facets of the relative abundance of species: the number of species in the community (species richness) and the

distribution of individuals among these species (species evenness). Many of the most frequently used indices are mathematically related, differing mainly in the extent to which they are influenced by one or other of these two facets (Hill, 1973). Thus the Shannon-Weiner index (Shannon and Weaver, 1963) is strongly influenced by the additional inclusion of rare species, whereas Simpson's index (Simpson, 1949) is really a measure of dominance, and so is principally affected by variation in species evenness.

The whole concept of species diversity has been questioned (Hurlbert, 1971). Because of the dual nature of diversity indices, different indices may rank the species diversity of different communities in different order (e.g., Chadwick and Canton, 1984; Platt *et al.*, 1984). Perhaps more important is the fact that similar species diversity patterns have been attributed to a variety of different and often contradictory processes. High species diversity has been considered to be a feature of both stable and unstable communities (MacArthur, 1955; May, 1972; 1976); to occur in areas of both stable and unstable or productive and unproductive environments (May and MacArthur, 1972; Rosenzweig, 1971; Brown and Davidson, 1977; Davidson, 1977; Sousa, 1979a; 1979b); to be associated with both high and low predation pressure (Paine, 1966; Connell, 1975; Creed, 1994) or large areas of heterogeneous habitat (MacArthur and MacArthur, 1961; Simberloff, 1976; Eadie and Keast, 1984).

Similarity metrics

These conflicting interpretations of diversity indices will present some difficulties if they are used to investigate ecosystem effects of fishing. Moreover, alone they take no account of species identity. Species fulfilling particular ecological roles may be replaced by others with similar ecology (e.g., Cody, 1968; Fager, 1968; Heatwole and Levins, 1972). Such changes may have little effect on the distribution of individuals among species and consequently may not alter species diversity. Such changes may be of enormous consequence to managers of natural resources. Species replacements in exploited systems may be common (Pimm and Hyman, 1987; Sherman, 1991), may persist for some years (Beddington, 1984), and may make it difficult to return the system to its original state (Brown *et al.*, 1976; Beddington, 1984; Sherman, 1991). Similarity indices are sensitive to variations in the abundance of particular species (for reviews, see Southwood, 1978; Washington, 1984; Magurran, 1991). Similarity index matrices are amenable to multivariate analyses, such as cluster and ordination analysis. These analyses can identify assemblages with similar species composition; identify the particular species either "typifying" or "discriminating" among them (Clarke, 1993); and relate differences among clusters to different environmental or disturbance regimes (e.g., Iglesias, 1981; Mirza and Gray, 1981; Guhl, 1987). Uncertainty and error are often handled quite primitively (if at all) in the analyses of

similarity indices. Moreover, there have been no attempts to link the properties reflected by variation in similarity indices directly to the ecosystem effects of fishing, although there is no conceptual barrier to such linkages.

Size Spectra

The biomass size spectrum is another feature of assemblages which has been proposed to reflect information regarding underlying ecological processes (Sheldon *et al.*, 1972; Platt, 1985; Borgmann, 1987; Beyer, 1989; Boudreau *et al.*, 1991; Boudreau and Dickie, 1992; Thiebaux and Dickie, 1992; 1993). In exploited fish assemblages larger fish generally suffer higher fishing mortality than smaller individuals and the size distribution becomes skewed towards the smaller end of the spectrum (Pope and Knights, 1982; Pope *et al.*, 1988; Murawski and Idoine, 1989). However, this change may have been due at least partly to natural causes, because similar shifts have been noted in plankton where variation in system productivity rather than fishing was the cause (Sprules and Munawar, 1986; Sprules and Goyke, 1995). Changes in the size distribution of fish assemblages may have consequences for diverse life-history traits of species (e.g., lower growth rates and earlier age of maturation) (Law and Grey, 1989; Stokes *et al.*, 1993) and trophodynamics of the assemblages (Daan, 1973; Robb and Hislop, 1980; Lyle, 1983; Basimi and Grove, 1985; Hislop *et al.*, 1991). The trophodynamic effects of decreasing net piscivory as the relative abundance of large predators decreases, may provide a weak negative feedback, possibly mitigating some of the impact of fishing: when fishing mortality increases, natural mortality declines.

The variety of ways in which multispecies assemblages might be affected by fishing (and environmental forcing) prompted previous meetings of WGECO to advocate that a suite of community descriptor measures should be applied to multi-species data sets (ICES, 1994a). The Working Group felt that use of only a few related metrics would provide an incomplete and possibly misleading picture of the community patterns, and make correct interpretation of any observed changes in community structure patterns more difficult.

3.2.2 Theory of the effect of fishing on size and diversity spectra

Fisheries biologists have used a number of models to predict the consequences of fishing in terms of changes in biomass and catch of commercially important fish stocks. With the increasing interest in the broader effects of fisheries on the marine environment, it has become relevant to study whether these models could also produce comprehensible predictions of fisheries impacts on community level attributes such as overall diversity and abundance at size.

A working document by Gislason and Rice demonstrated how length-based single species fisheries models as

summarised in, e.g., Sparre and Venema (1992) and MSFOR predictions (Gislason and Sparre, 1987; ICES, 1994a) could be used to predict how the slopes and intercepts of size and diversity spectra of commercially important North Sea fish would change in response to changes in overall fishing effort. Such prediction were made for eleven of the commercially exploited fish species. Both the single species length-based predictions and the multispecies predictions were made with constant recruitment.

Size spectra were constructed by plotting the natural logarithm of the average numbers by 10 cm length group (size range 10–100 cm) *versus* the natural logarithm of the mid-length of the size class. Diversity was expressed as the Shannon-Weiner index and plotted *versus* the mid-length of the corresponding size class. Multispecies predictions of numbers at age were assigned to length groups by assuming that the length was equal to 100 times the weight raised to a power of 1/3.

The model without species interactions produced the size and diversity spectra shown in Figure 3.2.2.1. The curves in the figure represent three levels of overall fishing intensity. These levels correspond to a halving, to the present, and to a doubling of the present level of fishing mortality for all species. The resulting size spectra can be reasonably well described by straight lines with a slope which is more negative for higher levels of fishing mortality. The diversity spectra, however, exhibit a peak in the length range from 30 to 40 cm. A similar curvilinearity was found in diversity spectra estimated from survey data from the North Sea (Figure 3.2.2.1). Because of the peak it was found most sensible only to fit a straight line to the diversity spectrum in the size range from 30–100 cm. For the size spectrum the whole size range from 10 to 100 cm was used.

Figure 3.2.2.2 shows how the slopes and intercepts of the diversity and size spectra will change as a function of fishing intensity. As fishing intensity increases the slope of the size spectrum steepens and the intercept increases. Note that the slope and intercept of the size spectrum are both linear functions of fishing intensity ($r^2 = 0.9997$ and $r^2 = 0.9967$). Contrary to the size spectrum the slope and the intercept of the diversity spectrum both changed very little with fishing effort. Here the points also fall on straight lines, but the slopes of the lines are close to zero. The results from the MSFOR predictions were close to the predictions from the non-interactive model. As in the non-interactive case the slope and the intercept of the number spectrum were linearly related to fishing effort and again the diversity spectrum shows little change with changes in fishing effort.

This preliminary analysis has revealed some noteworthy patterns in the predicted response of size and diversity spectra to changes in fishing mortality. In particular the linear change in the size spectrum with changes in fishing mortality deserves considerable attention. The linear change seems to be a common feature of both the non-

interactive and the multispecies predictions. If slopes and intercepts of size spectra are indeed linear functions of fishing mortality, it should be possible to estimate the relative development in fishing intensity from survey data on the overall change in the size composition of the fish fauna. In particular in tropical countries, where most fisheries are multispecific, species are short lived, and good and reliable time series of catch and effort data are lacking, the analysis of size spectra from survey data could provide an easy and cost-effective way of measuring the development of the fishery. Provided that the characteristics of the fishery (gear type, species and size selectivity) remained reasonably constant over the time period considered, the results presented suggested that the changes in the slope and intercept of the survey size spectrum were directly proportional to overall fishing mortality.

For species diversity the analysis suggests that the Shannon-Weiner index is less likely to be useful for monitoring changes in fishing intensity. The changes in the slopes and intercepts of the diversity spectrum, for fish larger than 30 cm were small compared to the changes which took place in the size spectrum and the response differed on either side of the peak in the spectrum. Diversity at length would therefore seem to be a less easily interpretable measure of fisheries impacts at the community level.

3.3 Intersessional Analyses of Species Diversity and Ordinations

3.3.1 Hypotheses and analysis framework

Following the strategy for examining fish assemblage structure recommended in earlier reports of WGECO, Greenstreet and Hall (in press) used a variety of different community measures to examine variation in the structure and composition of the groundfish assemblage (excluding shoaling pelagic species not adequately sampled by the groundfish gears) within a 64-year time period in three areas of the northwestern North Sea (Figure 3.3.1.1). Variation in species diversity was followed using two diversity indices, one sensitive to variation in species richness, the other to variation in species evenness. These were reinforced by analysis of k-dominance curves. Changes in species composition were examined using non-parametric multidimensional scaling ordination analysis, backed up by cluster analysis. The statistical significance of any clustering was determined using analysis of similarity randomisation tests. The key species responsible for clustering were identified using similarity of percentages analysis. Variations in length frequency distributions were also examined.

3.3.2 Surveys analysed

Variation in three areas was examined for two reasons. First, the patterns of fishing effort in each of the areas chosen were quite different over recent time (1960 to 1993), allowing the possibility of relating changes in

assemblage structure to variation in fishing pressure. Secondly, reducing the habitat heterogeneity by restricting the areas was also expected to reduce the β (between habitat) diversity in the metric, and make the investigation of fishing effects more powerful. In addition, Greenstreet and Hall (in press) applied their suite of analyses to both the whole groundfish assemblage, and to a sub-set of species which they considered were not the specific target of fishing. In this way they attempted to assess the indirect effects of fishing as well as the more direct effects of harvesting.

3.3.3 Results

1) Long-term differences in species diversity were subtle and most apparent when k-dominance curves were examined (Figure 3.3.3.1). Species diversity in the whole groundfish assemblage was marginally greater in the period 1929 to 1953. No difference was apparent within the non-target species assemblage in the two offshore areas. However, in the inshore area - the area where fishing effort, although on the decline, had been highest since the 1960s - clear long-term differences in the non-target assemblage were apparent. Spatial variation in species diversity was clearly defined within the whole groundfish assemblage; diversity was highest in the inshore region and least in the offshore area. No such obvious spatial gradient was detected for the non-target species assemblage.

2) Multivariate analysis indicated long-term changes and between-area differences in the species composition for both the whole groundfish assemblage and the non-target species sub-set (Figure 3.3.3.2). More detailed examination of the data revealed that much of the long-term variation resulted from relatively small and subtle differences in the relative abundance of rarer species, such as grey gurnard and spurdog. In contrast, changes in the relative abundance of the more common species, such as whiting and Norway pout, explained much of the between-area variation. An increase in Norway pout abundance was also implicated in the long-term changes (Figure 3.3.3.3).

3) Examination of species-aggregated length frequency distributions suggested that by the 1980s there had been a shift towards assemblages in which smaller fish were more highly represented. This was only apparent, however, in the whole groundfish assemblage. Specifically, the relative abundance of fish in the size category 11 to 20 cm was markedly higher between 1980 and 1993 than during the 1929 to 1953 period. This increase occurred in all three areas, although the difference was greatest in the offshore area and least in the inshore area. No trend was apparent in fish smaller than this, while the relative abundance of fish in all size catagories greater than 20 cm was higher during the early time period. The length frequency distributions of the non-

target species were almost identical in the two time periods.

4) Overall, the results suggested that, although differences in the structure of the whole fish assemblage can be detected, the non-target groundfish assemblage appears to have remained relatively unchanged, despite a century of intensive fishing activity.

3.4 Data Analyses: North Sea as a Whole

3.4.1 Data sets

IBTS

Data from the International Bottom Trawl Survey (IBTS) series in the first quarter from 1973 to 1993 were analysed. The core records were mean number per hour of towing per 1 cm length class for every fish species measured on each survey. Means were taken for all sets within each statistical area and for all sets within each of the North Sea Task Force (NSTF) areas. For analyses on the scale of the North Sea, data from statistical areas rather than North Sea Task Force areas were used.

EGFS

Data from the English Groundfish Survey (EGFS) in the third quarter from 1977 to 1993 were analysed. The core records were extracted as number per hour of towing per species, by 10 mm size interval, on the same scales as the IBTS data. Although benthos were recorded in many of the surveys, they were excluded from the community analyses reported here.

Both data sets were aggregated to numbers per 5 cm size class per haul per year, which were the basic intervals for analyses of properties of the size spectra. For each data series all species were included in community metrics. No partitions were made on the basis of trophic level, demersal or pelagic distribution, or geographic range (aside from the analyses investigating differences among NSTF areas).

3.4.2 Variables and data preparation steps

Total numbers per size class (TNUM): The sum of the number of individuals of all species within a size interval, for each year.

Diversity index (DIV): The Shannon-Weiner index of diversity.

$$\sum_{i=1}^{s} (p_i) (\log p_i)$$

In calculating proportions, the mean abundance per species within the size class was first calculated for the entire geographic unit (either for the entire North Sea or the specific North Sea Task Force area), and then means were summed across all fish in the size class.

Number of species per abundance class (NUMCLS): Ten classes of abundance were determined, following the octave scale used in biogeography (Preston, 1948). The categories were exclusive, with upper boundaries of None, < 1.0, < 2.0, < 4.0, < 8.0, ..., > 256 individuals per haul. The largest category had no upper bound. For each year the mean abundance across all statistical areas was calculated for each fish species and the species was assigned to the appropriate abundance category. The basic datum for analysis was the number of species in an abundance category each year.

The number of species recorded at least once was tallied for each survey time series, and the number of species in the zero category each year was that total number of species ever seen, less the sum of the number of species in the other categories that year. This means that the results may be affected by differences in sampling effort across years (the rarefaction effect) or differences in attention to recording rare species during surveys (the malefaction effect). A number of runs were made leaving out the zero category and exploring different lower bounds for excluding species as being too rare to be considered part of the community. The dominant patterns were similar among all the runs, so analyses contained in this report are for the full suite of species. The issues of sampling effort and recording dedication require further investigation in intersessional work, however.

Number of species per biomass class (BIOCLS): ten classes of biomass (kg per tow) were established, again on an octave scale. Because of differences in catch sizes of the IBTS and EGFS surveys, different boundaries were necessary between the second and third lowest classes (after class 1 – none). For the IBTS survey, the classes were: None, < 0.125, < 0.25, < 0.5, < 1.0, 2.0, 4.0, etc. For the EGFS, the categories were: None, < 0.25, < 0.5, etc. Assignment to categories followed the same procedures as were used for number per abundance class.

3.4.3 Statistical analyses

Size and diversity spectra. These were modelled with proc GLM in SAS on the ICES workstation. For all data sets, models were simple linear models fit separately for each year. From the annual analyses slopes, intercepts, their standard errors, and goodness of fit (as reflected by r^2) were extracted for investigation of trends over time. The model for the number spectra was ln (numbers) as a function of ln (size class), whereas for the diversity spectra the model was diversity as a function of size class. For the EGFS data, an additional series of models were fit, with terms for NSTF area and slopes nested under areas.

The range of size classes included in the spectrum analyses will affect the estimates of model parameters. The survey gear does not sample the smallest size classes with an efficiency comparable to that for larger sizes, providing a possible downward bias to numbers in the

small size classes. Very large fish are also rare, particularly in recent years, so the numbers in very large size classes are likely to be poorly determined. The diversity spectrum will be affected by both of these sources of error in the tails of the number spectrum. Moreover, work at past meetings and intersessionally has shown that the diversity spectrum is dome-shaped, so the relevant parameters are the slope and intercept on only the descending portion of the curve. The GLM models were fit to size intervals of 10–80 cm, 20–80 cm, and 40–95 cm. Results are reported only briefly in this report, but the conclusion was that the 20–80 cm interval was appropriate for estimation of parameters of the number size spectrum, and the 40–95 cm interval was needed for the parameters of the diversity size spectrum.

Meta-analyses were conducted on the annual results, regressing the parameters on year. Graphical displays of trends in parameters over time as well as in numbers and diversity over both size and years are presented. Because the diversity spectrum does not capture trends in diversity of small size classes, separate regressions were conducted on the change in diversity in the 15–35 cm size classes over years.

Analyses of frequencies of species by abundance and by biomass class. Because number of species is an ordinal variable, the tables of species by abundance and biomass classes (NUMCLS and BIOCLS) were fit by log-linear models using SAS Proc CATMOD, using the model = _Response_ mode of analysis. Each year was a separate row, but the abundance and biomass classes were aggregated into five categories, in order to allow full-rank estimation of the log-linear parameters. For abundance the categories were: None, < 1, < 16, < 128, and > 128 individuals per haul. For biomass, the categories for the IBTS surveys were: None, < 0.125, < 1.0, < 8.0, > 8.0 kg per haul, and for the EGFS the categories were: None, < 0.25, < 1.0, < 16.0, > 16.0 kg per haul. The key results used for biological inference were the Maximum-likelihood Analysis of Variance table, apportioning variance in the counts to factors and interactions, and parameter estimates for the individual year/category combinations. From the parameter estimates it is possible to investigate if there is a tendency for communities to become composed of perhaps more rare or fewer common species when intense fishing occurs.

For the analyses of the species by biomass categories, and the total species by abundance categories, the log-linear model had terms for year, category, and the year by category interaction. Year and category are design features. Years should not be significant; the same number of species was categorized in each year, although there will be some variance due to this factor because different numbers of species each year were in the size class which was aliased in the estimation step. Category will be significant but this merely reflects that, on average, the number of species in the various categories was unequal. The interaction between year and category

is the feature of interest. Its size relative to the main effects reflects the tendency for the community to consist of greater or lesser proportions of common and rare species over time. Note that this does not track the effect of fishing on individual species. Rather it would reflect the possibility that fishing causes a community to contain more rare species through widespread removals or to contain more common species perhaps as small species increase when released from predation as their predators are harvested.

For the analyses when species were assigned to abundance categories separately for individuals greater and less than 40 cm three factors were considered: size, year, and abundance category. The size factor was another design feature expected to be significant—more species can be small than large. The size by category interaction is of modest interest. The important features are the year by category and year by size interactions. The additional information in a significant year by size interaction is that over time there may be a tendency for fewer species to reach the 40 cm size class (a possible consequence of intensive exploitation) or for more species to grow to 40 cm or more (a possible, although unlikely, consequence of significant release from predation mortality).

3.4.4 Annual size spectra of the North Sea numbers and diversity

For both the IBTS and the EGFS data sets the regressions of ln (numbers) on ln (length class) were highly significant for each year (Tables 3.4.4.1 and 3.4.4.2). Except for two years (1973, 1980) in the IBTS data set, all r^2 values were > 0.9. These fits were for the standard range of length classes from 20–80 cm. For both data sets the goodness of fits were significantly poorer (binomial tests, H_0 that P (r^2) 20–80 > P (r^2) alternative size interval = 0.5) for intervals of similar width (i.e., with similar numbers of observations on the independent variable), but focused on ranges of length classes which were either smaller (10–70 cm) or larger (40–95 cm) (Table 3.4.4.3). All subsequent results regarding the number size spectra are with the 20–80 cm size interval.

For both the IBTS and EGFS data sets, all estimates of slopes and intercepts were significantly different from 0.0. Model parameters were generally well determined, with standard errors generally about 5% of the parameter value (Tables 3.4.4.1 and 3.4.4.2). For the IBTS data sets, the 1973 and 1980 parameter estimates have somewhat larger errors, but the parameters are still highly significantly different from 0.0. The 1980 estimates of slope and intercept are both outliers relative to estimates from surveys in other years.

The parameter estimates of annual slopes and intercepts are in the same general range for both the IBTS and EGFS data sets (Tables 3.4.4.1 and 3.4.4.2). Slopes range from −4.43 to −6.07 for the EGFS data, and −3.38 to −6.19 for the IBTS surveys (excluding the 1980 IBTS

estimate narrows the range to −4.50 to −6.19). The intercepts range from 17.23 to 20.26 for the EGFS and from 13.36 (or 15.28 without 1980) to 19.13 for the IBTS. The higher intercepts for the EGFS survey indicate that this survey catches more fish per haul in the North Sea than does the IBTS (especially because these are in ln numbers). However, the very similar range of slopes suggests that the two surveys are seeing very similar structures in how the numbers are distributed among size classes. More detailed examination of similarities between the parameters of the two surveys will be presented in Section 3.4.6. Note that the higher intercepts in the EGFS surveys are consistent with an interpretation that more fish (new recruits?) are present in the third quarter than in the first quarter, when the IBTS survey occurs. However, without detailed calibration of the size-specific capture efficiency of the gears, such an interpretation would be premature.

For both the IBTS and EGFS data sets, the regressions of diversity on size class were significant for the majority of years across the 40–95 cm size interval (9 of 14 years for EGFS, 14 of 21 years for IBTS; Tables 3.4.4.4 and 3.4.4.5). With size intervals across lower ranges, fewer regressions had significant slopes: 3 years with 10–70 cm for both data sets, 5 of 14 and 6 of 21 with length classes extending from 20–80 cm (Table 3.4.4.6). Correspondingly, the models for the range including the largest length classes accounted for a greater amount of variance than with ranges from 20–80 cm (IBTS, binomial $P = 0.022$; EGFS, $P = 0.031$), or 10–70 cm ($P < 0.01$, both data sets). In the early years of the IBTS series, the fits were better including length classes in the 20–80 cm range than across the 40–95 cm range. However, by 1979 the fits were usually better with the range starting at 40 cm; often by a large amount.

The intercepts were generally well estimated. Standard errors were generally around 20% of the estimate for the EGFS data sets, and for all but the early years of the IBTS data. In the early 1970s the standard errors were around 35% or more of the intercepts. The slopes were consistently negative, but for both data sets in some cases the estimates of slopes were close to 0.0 and with high uncertainty (Tables 3.4.4.4 and 3.4.4.5).

Again, the ranges of parameter estimates were similar for the two data sets. Slopes for the EGFS survey ranged between −0.0028 and −0.112, and slopes for the IBTS survey ranged between +0.0059 and −0.1637. Although the EGFS survey may be catching more fish than the IBTS survey, the fish sampled then are distributed in similar ways among species. The slopes of the diversity size spectra range from −1.53 to −3.16 for the EGFS survey and from −1.01 to −3.91 for the IBTS surveys. As with numbers, the size dependency of diversity is similar in both surveys.

From these results, the following was concluded:

1) The number size spectrum in both data sets can be described well by a simple linear model with ln (length class) as the independent variable.

2) The best range of length classes for estimating slope parameters of the number size spectrum is 20–80 cm. Including smaller sizes reduces the fit by including sizes in which some fish are probably not fully sampled.

3) The diversity size spectrum is described less well by a linear model with length class.

4) The best range of size classes for estimating slope parameters of the diversity size spectrum is 40–95 cm. Including smaller sizes includes sizes where diversity tends to increase with size rather than decrease.

5) The "dome" shape found in the previous meeting, and in the analyses reported in the intersessional studies (Section 3.2) is apparent, with a maximum at length classes as high as 35–40 cm in some years.

6) The maximum of the dome may have been lower in the early to mid-1970s than after about 1979.

7) It is possible that the number and diversity size spectra would remain linear above 95 cm. However, there are too few data to warrant fitting models to numbers and diversity in the very large size groups.

8) From the results of these analyses, the two surveys provide consistent descriptions of the North Sea fish assemblages.

3.4.5 Area effects on the annual size and diversity spectra

To begin to investigate the role of spatial heterogeneity on the number and diversity size spectra, the models fit to the ln (numbers) at ln (length class) and diversity at length had terms for North Sea Task Force area added. This term should capture overall differences in abundance or in the shape of the spectra among the areas. It is also possible that the dependency of numbers or diversity on size varied among areas. This possible relationship was investigated by including an additional term to estimate separate slopes for each NSTF area.

These analyses presented a plethora of output which could only be examined superficially in the time available at the meeting. Generally, however, results were similar to the results obtained at the previous meeting. The Type I sum-of-squares tables are presented for each year, for the number and diversity spectra in the IBTS and EGFS data (Tables 3.4.5.1a and 3.4.5.1b). The Type I sum-of squares are likely to overestimate the contribution of later terms (the area and the slopes nested under area) to explaining the variability in the numbers and diversity with size. Hence, the results discussed here

are probably an upper bound on the role of spatial heterogeneity in affecting these relationships, at the spatial scale of NSTF areas, and within the limitations of the linear models to fit the dome-shaped diversity size spectra.

For the number spectra in the EGFS data in all years the ln (length) was the model term capturing by far the greatest amount of variance in ln (numbers). These annual regressions have already been discussed (Section 3.4.4), and the analyses here do not alter the conclusions of those analyses. Further discussion will address only the additional contribution of the area and nested slope terms. Both terms were statistically significant in every year. However, the amount of variance explained by the area terms is small compared to the overall size-dependent trend (the main slope of ln (length)). In only two of the years did the area term explain as much as a third of the variance captured by the overall size effect, and in nearly half of the years the area term (with 10 levels) explained less than 20% of the variance explained by the single linear trend with size. The nested slopes never increased the variance explained by the model by as much as 10%, and in only 3 years by as much as 5%.

For the IBTS series, similar patterns were present. The ln (length) term dominated the models, with an even smaller contribution from the area terms. In about half the years prior to 1982, the area term accounted for as much as 15% of the variance explained by the overall size effect. After 1982, the area term never accounted for as much as 5% of the variance explained by the size term alone. In 1980 the nested slopes accounted for 15% as much variance as the ln (length) term, and in 1980 the nested slopes accounted for 10% as much variance. In all other years the nested slopes (again with 10 levels) never added as much as 5% to the variance explained by the model, although the term was always significant at the 0.02 level or more.

For both data sets, models fit to the diversity spectra captured relatively little variance, although generally most or all terms were statistically significant at the 0.05 level. A few patterns were apparent, but none of the relationships were strong enough to justify strong statements about the effect of area on diversity. For the IBTS data sets, in more than half the years the main effect for area accounted for more variance than did the overall size effect. This is of some interest, but it is not news that areas of the North Sea differ in fish species diversity. On the other hand, in 17 of the 21 years terms for the separate slopes for each area captured the least variance in diversity. The size trends apparently differ much less among NSTF areas. In the EGFS data, the size effect was always the largest term in the model, with the main area effect and nested slopes of lesser but usually similar importance.

The extent of output from these analyses, and the limited time at the meeting, precluded detailed exploration of these results, conduct of planned contrasts among specific areas, or looking at models fit to more restricted portions of the diversity size spectrum. Such investigations might be informative, but both these results and the analyses at the previous meeting suggest that the overall size effect is more important than the differences among NSTF areas. Possibly much finer spatial analyses might reveal much stronger local patterns in diversity or number spectra, although analyses at very fine spatial scales present many additional problems in data collection, analysis, interpretation, and presentation of results.

Conclusions

1) For the number size spectrum, the overall length effect is much larger than any area effects. The area effects generally increase the explanatory power of the models by just a few percent at best.

2) The variability among areas in slopes of the number spectra is usually even smaller than the overall area differences in numbers of fish.

3) For the diversity spectra, the Area and nested slopes explained amounts of variance more comparable to the variance explained by the overall size spectrum. This was predominantly because the size effect accounted for much less variation in diversity than in numbers, although areas did differ in diversity.

4) More complete analyses of the role of areas in the diversity spectrum should be undertaken before drawing strong conclusions about the importance or lack of importance of spatial effects.

3.4.6   Trends over years in the size spectra of numbers and diversity

The slopes and intercepts of the number size spectra from both surveys changed in a consistent linear manner over the duration of the surveys (Figures 3.4.6.1 and 3.4.6.2). As was reported at the last meeting, the number size spectra are becoming steeper, with large fish becoming less and less abundant from the 1970s to the 1990s. Correspondingly, the numbers of small fish, reflected by the intercepts, seem to be increasing over the period. The changes in slopes over time are similar for the two surveys: an annual change of −0.087 in the EGFS survey and −0.077 in the IBTS survey. The intercepts also change at similar annual rates: 0.142 for EGFS and 0.177 for the IBTS survey.

All these trends are brought together in Figures 3.4.6.3 and 3.4.6.4. For the EGFS, the increasing scarcity of large fish over time is apparent, as is the increase in smaller fish. At the top (large fish) the contour lines slope down and to the right, at progressively greater rates over time. In the lower part of the figure the contour lines slope up to the right, indicating more fish at smaller sizes. (The very lowest contour lines appear parallel, in sizes where the sampling gear is giving us little

information about abundance.) The contours for the IBTS survey are less smooth, but similar patterns are present. The increase in abundance of smaller fish, particularly in the 15 to 25 cm size classes, since 1985 is especially clear, as are the anomalous catches in 1980 (which provided anomalous parameter estimates and outliers in many of the analyses in Sections 3.4.3 to 3.4.5).

The slopes and intercepts of the diversity size spectra changed much less than for the numbers (Figures 3.4.6.5 and 3.4.6.6). However strong patterns in the residuals, particularly for the EGFS, and the large standard errors in the annual estimates, which were not accounted for in the regression model applied at the meeting, mean that the statistics of fit are not reliable indicators of the interannual trends. There is a general upward trend in intercepts starting in the early 1980s in both data sets, suggesting that overall diversity of the North Sea fish assemblage has been increasing. Both data sets also show lower slopes in diversity size spectra after the mid-1980s and a slight upward pulse in annual slope in the early 1980s.

Again, the summary Figures 3.4.6.7 and 3.4.6.8 display the overall patterns well. In the EGFS, a portion of the assemblage with low diversity is apparent at around 20 cm in 1977. This trough in diversity spreads to larger sizes, and a broader range of sizes over time, moving up, to the right, and widening out in Figure 3.4.6.7. As the low diversity reaches the largest sizes in the mid-1980s, there is an increase in diversity in the 30–50 cm, and larger 30–70 cm size classes. The trough is not apparent in the IBTS (Figure 3.4.6.8), but contour lines do slope up and to the right for intermediate to large sizes through the 1980s. The increase in diversity in the 30–70 cm size classes since the mid-1980s is also clear, as is the brief drop in diversity in the 25–45 cm size range in the early 1980s.

There was insufficient time at the meeting to explore in depth the causes of these patterns in numbers and diversity over size and time. However, the patterns and results are easily reconciled with a few processes. Fishing is altering the number size spectrum, removing biomass and numbers from the large size classes over time, and having effects on intermediate (40+ cm) length groups in recent years. Fishing could also be responsible for the increase in diversity of the intermediate length groups from the mid-1980s onward. If fisheries target on the most abundant species, by removing large numbers of individuals (the increasing slope of the number size spectrum), the fisheries increase the evenness of fish in those size ranges. This would increase diversity in these size classes, and decrease the slope of the annual size dependency of diversity.

The drop in slope of the diversity spectra in the early 1980s may have a different cause. The change in slope was due to a lowering of diversity in the intermediate size classes. Recruitment of a very strong year class of an abundant species, say herring or whiting, moving from the 20 to 50/60 cm range through the 1980s, could have such an effect, by increasing dominance (lowering evenness) in progressively larger size intervals.

This explanation of the patterns has fishing and recruitment variation affecting different parts of the number and diversity size spectra in different places and in different but consistent ways. It is attractive, because it has the potential to partition the effects of environmental forcing and fish harvests on community attributes. The results are also consistent with general ecological theory (Section 3.2.1) and developing theory of how fishing affects number and diversity size spectra (Section 3.2.2). This is encouraging, but much more rigorous investigation of the data, independent tests of the ideas, and further development of the theoretical basis for these relationships are all needed.

Conclusions

1) The slopes of the annual number size spectra have become steeper over the past 20 years, due to a decrease in numbers of fish in larger length classes and an increase in numbers of fish in the small length classes. This has probably been caused by fishing.

2) The intercepts of the annual number spectra have become larger over the past 20 years, due at least in part to larger numbers of fish in small size classes. This could have several causes, including increased productivity of exploited populations or reduced predation mortality (indirect effects of fishing), or environmental factors being somewhat more favourable for recruitment of at least some species (not a fishing effect).

3) Different kinds of analyses will be needed to resolve the alternatives in 2), and to see if biomass and numbers are conserved in the North Sea (does the increase in intercept of the number size spectrum compensate fully for the increase in slope over the same period?).

4) The rates of change in the number size spectra are similar in both the IBTS and EGFS data sets, and are major features of the size-specific patterns.

5) The diversity spectrum does change over time, but the variation in both the annual slopes and intercepts is described poorly by a linear model. The slopes, in particular, are affected in some years by dynamics in intermediate size classes, and in other years by dynamics in larger sizes.

6) In both data sets, diversity in size groups above 40 cm has increased since the mid-1980s. This is probably due to fishing, which has increased evenness of species in these sizes.

7) There is a trough in diversity that moves from small sizes in the early 1980s to large sizes by the late 1980s in the EGFS data. This is unlikely to be a fishing effect, but may be an environmental signal. Further analyses are necessary before drawing conclusions on this possibility.

8) The patterns in the number size spectra are consistent with the theory developed for fishing effects on communities.

9) Some patterns in the diversity size spectra are consistent with theory, and some patterns seem not to be. It is unclear whether the theory is wrong, is correct but incomplete, or the diversity size spectra are a weak feature of the system.

10) The major effects of fishing on diversity appear to be on intermediate sizes, increasing evenness of species, and therefore increasing diversity, rather than reducing diversity through making species rare.

3.4.7 Analysis of frequencies of species by abundance and biomass class

In the light of some of the above analyses, the Working Group was prompted to undertake some preliminary analyses of species abundance frequency distributions. Ecologists have viewed ecosystem structure in this perspective for several decades (Preston, 1948). Because fish may respond to exploitation in many ways, including changes in growth and recruitment levels, the Working Group investigated species occurrences by categories of biomass, as well as by categories of abundance.

The counts of numbers of species in each abundance group each year are presented in Tables 3.4.7.1 and 3.4.7.2 for the EGFS and IBTS. The tables are stratified into small (< 40 cm) and intermediate to large (> 40 cm) fish. The combined counts are presented in Figure 3.4.7.1, with lowess smoothers fit to each panel. Trends over time are clear for most of the abundance classes. In the IBTS data fewer and fewer species are completely absent from the surveys over years; for the EGFS surveys the number of species not seen increases for a few years, but has decreased markedly since the early 1980s. It would be rash to conclude that the North Sea ecosystem is becoming species richer, however. Increasing care in recording rare species on surveys would produce the same pattern.

In the panels reflecting categories of abundance, the two surveys show quite different patterns. In the IBTS surveys, the number of rare species (panel 2) increased for the first half of the time period, but has decreased since the mid-1980s. All other categories show increasing numbers of species, except the three largest categories. There, number of species declined for the early part of the series, and increased since the mid-1980s. In the EGFS surveys the numbers of rare species (panels 2 and 3) have increased more or less steadily over the period,

whereas the species of intermediate abundance have generally declined. The number of species averaging between 32 and 128 individuals per tow was stable or declined until the mid-1980s, and increased thereafter. The numbers of species averaging more than 128 individuals per tow showed an inverse pattern.

The patterns in the individual panels are not independent, of course: species move from category to category as their abundance changes with time. Overall the IBTS data suggest that more species are becoming numerous with time. Species not recorded in the early years do not just get recorded rarely or occasionally (which would place all the change in the first few panels). Almost all the abundance classes are receiving more species over time, which is consistent with the increasing intercepts in the regression number size spectra over time. The EGFS surveys suggest that there are higher numbers of rare species in the North Sea, fewer species of on average 3–16 individuals per tow (although this average could occur from a few large catches or many small ones within a year), and fewer very common species.

These patterns are statistically real. The log-linear models fit to both data sets have significant Year by Abundance Class interactions (Table 3.4.7.3). In both surveys the variance attributable to the interaction term is significant, and comprises about 35–40% of the total variance. This percentage is of limited information on its own, because changing abundance category boundaries could change its value substantially, although not the significance of the interaction. It is useful for contrast among data sets, suggesting that the surveys see the same magnitude of system change over time, in the sense of net numbers of species experiencing changes in abundance.

The parameter estimates of this interaction term evaluate whether the individual observations are larger or smaller than expected given the row [year] and column [abundance class] totals. Their patterns show where the major dynamics are centered. The full matrices of parameter estimates are difficult to assimilate, but by splitting the time series into two (1977–1983; 1984–1990 for EGFS) or three (1973–1979; 1980–1985; 1986–1993 for IBTS) parts, consistent trends over time in some parameters are apparent. For the EGFS data, in the early period estimates generally show a surplus of species in the abundant (> 64 fish per tow) category, and a deficit of species in the rare (< 1 fish per tow) category. These deficits and surpluses switch in the second half of the series. The EGFS is seeing significantly fewer common species and more rare species over time. The IBTS survey is showing something quite different: compared to the earliest period fewer species are absent, fewer species are of intermediate abundance, and more species are very numerous.

None of the conclusions are altered when the species counts by abundance categories are stratified by size. The Year by Abundance Class interaction remains significant for both surveys, and still accounts for about a third as

much variance as the main effects in the model (Table 3.4.7.4). Parameter estimates of the year by abundance class interaction level show nearly identical patterns between early and late periods for both surveys.

With a few exceptions, classifying species by biomass category rather than abundance category produces similar but less marked patterns (Figure 3.4.7.2; Tables 3.4.7.5a and 3.4.7.5b). The main difference in the EGFS surveys is a possible recent increase in the number of species with intermediate biomass in catches. This rise is strongly influenced by a single year's survey, however. The numbers of species in the two highest categories of biomass show marked declines over most of the time series. In the IBTS data, the number of species reaching high biomasses is not increasing, although the number of species reaching high abundance does increase.

The log-linear models support the interpretation above (Table 3.4.7.6). For the EGFS surveys, however, the interaction between Year and Biomass Categories is not significant overall, suggesting a large scatter from year to year, with no consistent trends in numbers of species reaching various biomass levels. This suggests that the intuition that the response of fish populations to exploitation is manifest more directly by biomass than numbers—or that the fisheries community conserves biomass much better than it conserves numbers—has some merit. With the interaction term insignificant, it is not legitimate to use the individual parameter estimates (but those who cheat would see that the patterns are similar to patterns seen in the analysis of abundance: fewer species reach high biomass in recent years compared to earlier, and more species have low biomasses).

In the IBTS data, the interaction term is significant, but substantially smaller than the term in the analysis of classification by abundance. Patterns in the parameter estimates are marked. The IBTS indicates that there are more species in the intermediate and high biomass categories, and many fewer species missing altogether. Although analysis of the number size spectra from the IBTS surveys showed that the size composition of the North Sea fish community was changing (at least partly because of fishing), these analyses suggest that more species have been numerous and of high total biomass recently than 15–20 years ago.

Conclusions

1) The numbers of species which are rare and which are common, or which are of low or high total biomass have changed significantly over the two decades.

2) Both surveys indicate that the patterns of change are strongest in the largest and most numerous size classes, but show different trends.

3) The number of species completely missed by surveys in a year has decreased over the survey period for both surveys, but this may be a recording artifact.

4) The EGFS provides preliminary evidence that the distribution of species by level of biomass may be a more consistent property of the North Sea than the distribution of species by abundance level.

5) Until the possible artifacts due to recording of rare species are dealt with effectively, analyses of species occurrences by abundance or by biomass should be interpreted with great caution.

6) If the problem in 5) can be resolved, species-abundance relationships in fisheries surveys should be analysed more thoroughly, possibly to bring another body of community ecological theory to bear on questions about the effects of fishing on fish assemblages.

7) Along with 6), species-biomass categorizations should also be investigated. Such relationships may provide an avenue to link fish population dynamics models to some types of community ecological theory.

3.5   Contrasts Among Selected Areas in the Northwestern North Sea

This analysis used the same data sets as in Greenstreet and Hall (in press), and 10 cm length categories. The natural log of the number of fish in each category was regressed on the natural log of the category mid-point over the size category range 25 cm to 85 cm. Species diversity, calculated using the Shannon-Weiner index, at each length category was regressed on category mid-point size over the category range 15 cm to 75 cm. The slope and intercept parameters thus obtained were plotted against time for each of the three areas in the northwestern North Sea (Figure 3.5.1).

Data in the early time period were more variable than in the later data set. Because of this, long-term temporal differences, particularly for the numbers at length spectrum, are not compelling. Even so, numbers at length spectrum slopes tended to be more negative and the intercepts higher in the later time period, suggesting a shift in the size distribution towards smaller fish. Greenstreet and Hall (in press) present the actual size frequency distribution data. These data showed that between 1980 and 1993 the proportion of fish in the 11 cm to 20 cm size category increased markedly whereas the proportion of fish in all the larger categories declined. Their data suggested that this was mostly due to an increase in the relative abundance of Norway pout, particularly in the two offshore areas. Data for the diversity at length spectrum are more convincing and indicate a marked between-area effect in the later period. During the period 1929 to 1953 diversity spectrum slopes

were mostly negative and their intercepts greater than one in all three areas. The same is true for the Scottish east coast area in the period 1980–1993, but in the two offshore areas the diversity spectrum slopes tended to be positive and their intercepts less than one. These data suggest that in the two offshore areas species diversity increased in the larger size categories and declined in the smaller size classes over the intervening period.

A problem with these analyses is their assumption that the data fit a linear model. Despite this, they do appear to be sensitive to variation in some characteristics of the groundfish species assemblage structure. To overcome the problem of fitting a linear model, the actual data were plotted and curves fitted by lowess smoothing (Figure 3.5.2). Clear shifts in the numbers at length and diversity at length spectra are obvious. The changes are similar in the two offshore areas where numbers of fish in length categories up to 60 cm have increased. Species diversity in the fish under 40 cm in length has declined, whereas in fish of greater length species diversity has increased.

There has been some debate as to the value of species diversity indices (see above) and some authors have suggested that species richness, the actual number of species recorded in a sample, is a more valuable statistic (see, for example, Rosenzweig, 1995). Hall (1996) used a graphical approach to examine time trends in abundance and species richness at length in the same data sets. Use of a graphical method avoided the problem of fitting some form of model to the data. No overall trend in groundfish species richness was evident between the two time periods, but interesting trends in species richness and log abundance at length were revealed. For the latter analyses, data were only presented for the east Shetland area, although the general patterns were similar in all three areas. Within the recent time span (1980–1993) species richness in the smaller 5 cm size classes increased; on average species richness was greatest among fish of between 25 cm and 30 cm in length. The log abundance of fish greater than 65 cm in length declined over the period while an increase was noted among the smallest fish. Data for the earlier time period (1929–1953) were sparser and more variable; however, trends in species richness were indicated among fish less than 15 cm in length, but in this instance they were negative. Hall (1996) made no direct comparison between time periods. However, he presents the data which make this possible, at least for the east Shetland region, and the results are consistent with those observed when a Shannon-Weiner index of diversity was used rather than the simple count of species (Figure 3.5.3). The interesting point to note from this is that variation in species evenness does not alone account for the changes apparent in Figure 3.5.2; the actual number of species also changes.

One notable result which emerges from these analyses is the between-area differences that are clearly evident in the later time period. The difference is most obvious when the species assemblage in the Scottish east coast area is compared with those in the two more offshore areas. The Scottish east coast region was the only one where long-term changes in the non-target species assemblage were identified. Greenstreet and Hall (in press) present data for variation in fishing effort in each of the three areas over the period 1960 to 1993. Effort declined by 60% in the Scottish east coast area, from 500,000 hours in 1960 to 200,000 hours in 1993, while in the other two areas effort has doubled from around 125,000 hours to approximately 250,000 hours. These clear differences in the patterns of fishing effort in the three areas appear to be reflected in the diversity at length metrics (Figures 3.5.1 and 3.5.2). This raises the intriguing prospect that diversity spectrum analyses may reflect the differences in the history of fishing effort, in circumstances of both increasing and declining fishing activity.

3.6 International Beam Trawl Surveys 1990–1995

3.6.1 Introduction

Commercial beam trawl surveys in the third quarter have been undertaken in the North Sea and western waters by the UK, the Netherlands, Belgium and Germany since the mid-1980s, and summary results have been presented in reports of the Study Group on Beam Trawl Surveys (ICES, 1991; 1993; 1994b; 1995). The main purpose of the surveys is to provide abundance indices of sole and plaice for use in assessment; a detailed account of the sampling procedures used and data collected has been presented elsewhere. In summary, data are collected by beam trawls of commercial design, ranging from 4–8 m wide and operating in depths of 6–55 m. All fish species caught are measured, and subsamples of some species are aged.

The following section describes a preliminary analysis of spatial changes in community structure by ICES rectangle for the North Sea to 57° N and for the west coast of the UK to 9° W.

3.6.2 Sampling and analysis

The data used here are combined by haul since 1990 for the UK and Dutch surveys, since 1993 for Belgian surveys and since 1995 for the German surveys. Records for each haul include the total catch number of each species caught (numbers/hour/8 m beam trawl), and the total catch number of each species by 5 cm length classes. Only demersal non-target finfish species and demersal finfish species of commercial importance have been included in this analysis.

3.6.3 Results

Species richness decreased since 1990 from southwest to northeast and from coastal waters to offshore areas (Figure 3.6.3.1). Species richness was highest in coastal waters of the Irish Sea and southwest England (Divisions VIIa, VIIf, VIIg, mean = 33) and in the Channel

(Divisions VIId, VIIe, mean = 35), and was lowest for the rectangles sampled in the North Sea (Divisions IVb, IVc, mean = 22). There appears to be a sharp decrease in the number of species as one moves from the eastern Channel to the southern North Sea. This reflects the change between the more numerous Lusitanian fauna to the southwest and the boreal fauna to the northeast. As expected, higher sampling effort within a rectangle generally provided larger numbers of fish species (Figure 3.6.3.1).

The Shannon-Weiner index of diversity showed a similar pattern to the number of species. Diversity decreased from southwest to northeast and declined from inshore waters to offshore waters, except perhaps in the German Bight of the North Sea (rectangles 38F7, 39F7, and 40F7) where the diversity was low in inshore waters due to the predominance of small dab and plaice in the catch (Figure 3.6.3.2). In the North Sea, the diversity tended to be somewhat higher on the English coast than along the continental coast. Highest diversities were observed in the coastal rectangles of the Irish Sea.

In all areas the majority of demersal species caught were less than 40 cm in length and peaked at 10–25 cm (Figures 3.6.3.3a and 3.6.3.3b). In the western areas (VIIa, VIIf) a small peak occurred in the length range of 45–65 cm which included lesser spotted dogfish, thornback ray, turbot, and some brill (Figure 3.6.3.3b). The catch of fish between 10–25 cm was dominated by flatfish species throughout the area, but the absolute abundances were higher in the North Sea than in the western areas, except on the English east coast.

Diversity was calculated by 5 cm size class and the diversity spectra for each area are shown in Figures 3.6.3.4a and 3.6.3.4b. In general, the diversity spectra were dome-shaped and decreased with increasing fish size. In the western areas, diversity remained above a level of 1.5 for size classes up to 60 cm, whereas in the North Sea diversity fell below this level at size classes of above 40 cm.

The present analysis is a preliminary overview of the main features of the demersal fish community. Further analysis will investigate the evenness of the distribution of species within the community, analyse the relationships with habitat characteristics (sediment type, depth, hydrography) and a comparison between present-day structure and historic records.

3.7 Overall Conclusions

3.7.1 Regarding metrics of community structure

1) The number size spectrum does seem to be an appropriate summary parameter for analysis of changes in species assemblages. There is now both some theoretical foundation and some empirical evidence that the slope (and possibly intercept) of

the number spectrum does reflect some biologically meaningful properties of communities, and should respond in consistent ways to fishing. The number size spectrum may be a valuable tool in fisheries investigations when reliable catch-age and/or effort data are not available.

2) The diversity size spectrum is another appropriate summary parameter. However, the slope and intercept of the linear portion of the spectrum has limited information. The theoretical analyses indicate that the diversity spectrum should contain very little information about the effects of fishing on the community. Instead, the different portions of the spectrum–the diversity in the small, intermediate, and large sizes of fish taken in surveys–should be examined individually.

3) The analysis of trends in species richness by size class may be at least as informative as analysis of trends in diversity by size class.

4) For the IBTS and EGFS, 20–80 cm is a good size interval for analyses of properties of the number spectrum. When properties of the diversity spectrum are analysed, 40–95 cm may be a more suitable size range. (See the cautions in 2, however.) For the Beam Trawl survey, a range of 15–40 cm is appropriate.

5) Analyses of the frequency of occurrence of species by abundance category suggested that this approach also has some value as a summary parameter for community analyses. However, much more exploratory work is needed on this topic. It is also necessary to examine the ecological theory on species abundance relationships, with the goal of developing linkages with fish population dynamics modelling.

6) Analysis of the frequency of occurrence of species by biomass category is a new twist on the more commonly encountered species-abundance relationships. Because the biomass responses of fish populations to exploitation may differ from their numerical responses, both analyses should be pursued together.

7) Although the effect of area on number and diversity of fish seems to be substantially smaller than the effect of length, this meeting only superficially explored the potential of spatial information to improve the ability of the summary parameters to reflect community dynamics or ecosystem effects of fishing. The Beam Trawl data on a much finer scale of spatial resolution, yet with wide coverage, do show strong spatial effects on community properties.

8) Many of the analyses conducted at this meeting are characterized better as pattern analyses than as

scientific hypothesis testing. The formulation of clear, testable hypotheses, capable of supporting the approach of strong inference (Steele, 1973), should be a core tool for research and analyses on ecosystem effects of fishing (as it should be elsewhere in fisheries and marine science).

9) We are ready to move from a strictly exploratory approach to the application of these metrics to fish (and other) communities, and begin to deal more rigorously with statistical considerations such as autocorrelated annual values for many of our summary parameters.

3.7.2 Regarding the dynamics of the North Sea fish community (as reflected by surveys) and the effects of fishing on that community

1) Many, but not ALL, features present in the EGFS data set were also present, and of similar magnitude, in the IBTS data set. The availability of two survey series was valuable in interpreting the results of these analyses. The new Beam Trawl survey database will be a very valuable contribution to these types of studies. The initial analyses conducted at this meeting suggest that this data set displays some patterns observed in the other two data sets.

2) The slope of the number size spectrum has increased over the past 20 years, and some (possibly much) of this change is probably due to fishing. Numbers of large fish have decreased, whereas numbers of fish in the smallest sizes sampled by the survey gears have increased.

3) Diversity has increased in the intermediate size groups, particularly over the past 10 years. This change is at least partly due to fisheries reducing the abundance of dominant species, thereby increasing community evenness.

4) Although the number of species showing high numbers and high biomass may have increased over the past decade or more, this should not be taken as an indication that the North Sea fisheries ecosystem is necessarily thriving. Rather, the combined results indicate that the North Sea fish community is becoming more and more composed of large numbers of relatively small individuals, even for taxa which once contained substantial numbers of large individuals.

5) The diversity of fish in the 30–70 cm size classes has increased in the past decade, to the highest levels seen by either survey. Again, however, this should not be taken as evidence that the North Sea fish community is healthier than ever. Rather, fishing pressure is reducing the abundance and ecological dominance of many or all species

common in this size range, increasing evenness, and therefore diversity.

6) Changes in overall richness of the North Sea fish community cannot be examined until possible artifacts due to differential effort or care in recording rare species over time are resolved.

7) Our investigation of these summary parameters has reached a point where we feel that combinations of them can be used (albeit with caution) to explore hypotheses regarding mechanisms causing the patterns in these and other data, or the mechanisms underlying the indirect effects of fishing on fish communities. Some illustrations are contained in Section 10.

3.8 References

Basimi, R.A., and Grove, D.J. 1985. Studies on feeding, growth and production of a recruited inshore population of *Pleuronectes platessa* (L.) at East Anglesey, North Wales. Journal of Fish Biology, 27: 765–783.

Beddington, J.R. 1984. The responses of multispecies systems to perturbations. *In* Exploitation of Marine Communities, pp. 209–225. Ed. by R.M. May. Life Sciences Research Report 32. Springer Verlag, New York.

Beyer, J. 1989. Recruitment stability and survival— simple size-specific theory with examples from the early life dynamics of marine fish. Dana, 7: 45–147.

Borgmann, U. 1987. Models on the slope of, and biomass flow up, the biomass size spectrum. Canadian Journal of Fisheries and Aquatic Sciences, 44 (Suppl. 2): 136–140.

Boudreau, P.R., and Dickie, L.E. 1992. Biomass spectra of aquatic ecosystems in relation to fisheries yield. Canadian Journal of Fisheries and Aquatic Sciences, 49: 1528–1538.

Boudreau, P.R., Dickie, L.E., and Kerr, S.R. 1991. Body size spectra of production and biomass as system-level indicators of ecological dynamics. Journal of Theoretical Biology, 152: 329–340.

Brown, J.H., and Davidson, D.W. 1977. Competition between seed-eating rodents and ants in desert ecosystems. Science, 196: 880–882.

Brown, B.E., Brennan, J.A., Grosslein, M.D., Heyerdahl, E.G., and Hennemuth, R.C. 1976. The effect of fishing on the marine finfish biomass in the Northwest Atlantic from the Gulf of Maine to Cape Hatteras. International Commission for the Northwest Atlantic Fisheries Research Bulletin, 12: 49–68.

Chadwick, J.W., and Canton, S.P. 1984. Inadequacy of diversity indices in discerning metal mine drainage effects on a stream invertebrate community. Water, Air and Soil Pollution, 22: 217–223.

Clarke, K.R. 1993. Non-parametric multivariate analyses of changes in community structure. Australian Journal of Ecology, 18: 117–143.

Cody, M. 1968. On the methods of resource division in grassland bird communities. American Naturalist, 102: 107–148.

Connell, J.H. 1975. Some mechanisms producing structure in natural communities. In Ecology and Evolution of Communities, pp. 460–490. Ed. by M.L. Cody and J.M. Diamond. Belknap Harvard University Press, Cambridge, Massachusetts.

Creed, R.P. 1994. Direct and indirect effects of crayfish grazing in a stream community. Ecology, 75: 2091–2103.

Daan, N. 1973. A quantitative analysis of the food intake of North Sea cod, Gadus morhua. Netherlands Journal of Sea Research, 6: 479–517.

Davidson, D.W. 1977. Species diversity and community organization in desert seed-eating ants. Ecology, 58: 711–724.

Eadie, J.McA., and Keast, A. 1984. Resource heterogeneity and fish species diversity in lakes. Canadian Journal of Zoology, 62: 1689–1695.

Fager, E.W. 1968. The community of invertebrates in decaying oak wood. Journal of Animal Ecology, 37: 121–142.

Gislason, H., and Sparre, P. 1987. Some theoretical aspects of the implementation of Multispecies Virtual Population Analysis in ICES. ICES CM 1987/G:51.

Greenstreet, S.P.R., and Hall, S.J. In press. Fishing and groundfish assemblages structure in the northwestern North Sea: on analysis of long-term and spatial trends. Journal of Animal Ecology.

Guhl, W. 1987. Aquatic ecosystem characterizations by biotic indices. Internationale Revue der gesamten Hydrobiologie, 72: 431–455.

Hall, S.J. 1996. Trends in body size and species richness in marine fish assemblages. WGECO 1996/4/1.

Heatwole, H., and Levins, R. 1972. Trophic structure stability and faunal change during recolonization. Ecology, 53: 531–534.

Hill, M.O. 1973. Diversity and evenness: a unifying notation and its consequences. Ecology, 54: 427–432.

Hislop, J.R.G., Robb, A.P., Bell, M.A., and Armstrong, D.W. 1991. The diet and food consumption of whiting (Merlangius merlangus) in the North Sea. ICES Jornal of Marine Science, 48: 139–156.

Hurlbert, S.H. 1971. The non-concept of species diversity: a critique and alternative parameters. Ecology, 52: 577–586.

Huston, A.H. 1994. Biological diversity: the coexistence of species on changing landscapes. Cambridge University Press, Cambridge. 681 pp.

ICES. 1991. Report of the Study Group on Beam Trawl Surveys in 1990. ICES CM 1991/G:81.

ICES. 1993. Report of the Study Group on Beam Trawl Surveys. ICES CM 1993/G:5.

ICES. 1994a. Report of the Multispecies Assessment Working Group. ICES CM 1994/Assess:9.

ICES. 1994b. Report of the Study Group on Beam Trawl Surveys in 1993. ICES CM 1994/G:5.

ICES. 1995. Report of the Study Group on Beam Trawl Surveys in 1994. ICES CM 1995/G:5.

Iglesias, J. 1981. Spatial and temporal changes in the demersal fish community of the Arosa River (NW Spain). Marine Biology, 65: 199–208.

Law, R., and Grey, D.R. 1989. Life-history evolution and sustainable yields from populations with age-specific cropping. Evolutionary Ecology, 3: 343–359.

Lyle, J.M. 1983. Food and feeding habits of the lesser spotted dogfish in Isle of Man waters. Journal of Fish Biology, 23: 725–737.

MacArthur, R.H. 1955. Fluctuations of animal populations, and a measure of community stability. Ecology, 36: 533–536.

MacArthur, R.H., and MacArthur, J.W. 1961. On bird species diversity. Ecology, 42: 594–598.

Magurran, A.E. 1988. Ecological diversity and its measurement. Chapman and Hall, London.

Magurran, A.E. 1991. Ecological diversity and its measurement. Chapman and Hall, London.

May, R.M. 1972. Will a large complex system be stable? Nature, 238: 413–414.

May, R.M. 1975. Patterns of species abundance and diversity. In Ecology and Evolution of Communities, pp. 81–120. Ed. by M.L. Cody and J.M. Diamond. Belknap Press, Harvard.

May, R.M. 1976. Patterns in multi-species communities. *In* Theoretical Ecology: Principles and Applications, pp 142–162. Ed. by R.M. May. Blackwell Scientific, Oxford.

May, R.M., and MacArthur, R.H. 1972. Niche overlap as a function of environmental variability. Proceedings of the National Academy of Science of the United States, 69: 1109–1113.

Mirza, F.B., and Gray, J.S. 1981. The fauna of benthic sediments from the organically enriched Oslofjord, Norway. Journal of Experimental Marine Biology and Ecology, 54: 181–207.

Murawski, S.A., and Idoine, J.S. 1989. Multispecies size composition: a conservative property of exploited fishery systems. Northwest Atlantic Fisheries Organization Document 89/76. 12 pp.

Paine, R.T. 1966. Foodweb complexity and species diversity. American Naturalist, 100: 65–75.

Peet, R.K. 1974. The measurement of species diversity. Annual Review of Ecology and Systematics, 5: 285–307.

Pielou, E.C. 1975. Ecological diversity. Wiley, New York.

Pimm, S.L., and Hyman, J.B. 1987. Ecological stability in the context of multispecies fisheries. Canadian Journal of Fisheries and Aquatic Science, 44 (suppl. 2): 84–94.

Platt, H.M., Shaw, K.M., and Lambshead, P.J.D. 1984. Nematode species abundance and their use in the detection of environmental perturbations. Hydrobiologia, 118: 59–66.

Platt, T., 1985. Structure of marine ecosystems: its allometric basis. Canadian Bulletin of Fisheries and Aquatic Sciences, 213: 55–64.

Pope, J.G., and Knights, B.J. 1982. Comparison of length distributions of combined catches of all demersal fishes in surveys in the North Sea and at Faroe Bank. *In* Multispecies Approaches to Fisheries Management Advice. Ed. by M.C. Mercer. Canadian Special Publication in Fisheries and Aquatic Science, 59: 116–118.

Pope, J.G., Stokes, T.K., Murawski, S.A., and Idoine, S.I. 1988. A comparison of fish size-composition in the North Sea and on Georges Bank. *In* Ecodynamics; Contributions to Theoretical Ecology, pp. 146–152. Ed. by W. Wolff, C.-J. Soeder, and F.R. Drepper. Springer-Verlag, Berlin.

Preston, F.W. 1948. The commonness and rarity of species. Ecology, 29: 254–283.

Robb, A.P., and Hislop, J.R.G. 1980. The food of five gadoid species during the pelagic 0-group phase in the North Sea. Journal of Fish Biology, 16: 199–217.

Rosenzweig, M.L. 1971. Paradox of enrichment: destabilization of exploitation ecosystems in ecological time. Science, 171: 385–387.

Rosenzweig, M.L. 1995. Species diversity in time and space. Cambridge University Press, Cambridge. 436 pp.

Shannon, C.E., and Weaver, W. 1963. The mathematical theory of communication. University of Illinois Press, Urbana. 117 pp.

Sheldon, R.W., Prakash, A., and Sutcliffe, W.H., Jr. 1972. The size distribution of particles in the ocean. Limnology and Oceanography, 17: 327–340.

Sherman, K. 1991. The large marine ecosystem concept: research and management strategy for living marine resources. Ecological Applications, 1: 349–360.

Simberloff, D.S. 1976. Experimental zoogeography of islands: effects of island size. Ecology, 57: 629–648.

Simpson, E.H. 1949. Measurement of diversity. Nature, 163: 688.

Sousa, W.P. 1979a. Experimental investigation of disturbance and ecological succession in a rocky intertidal algal community. Ecological Monographs, 49: 227–254.

Sousa, W.P. 1979b. Disturbance in marine intertidal boulder fields: the non-equilibrium maintenance of species diversity. Ecology, 60: 1225–1239.

Southwood, T.R.E. 1978. Ecological methods with particular reference to the study of insect populations. 2nd Edition. Chapman and Hall, London.

Sparre, P., and Venema, S.C. 1992. Introduction to tropical fish stock assessment. Part 1. Manual. FAO Fisheries Technical Paper No. 306.1, Rev. 1, Rome, FAO. 376 pp.

Sprules, W.G., and Goyke, A.P. 1994. Size-based structure and production in the pelagia of Lakes Ontario and Michigan. Canadian Journal of Fisheries and Aquatic Sciences, 51: 2603–2611.

Sprules, W.G., and Munuwar, M. 1986. Plankton size spectra in relation to ecosystem productivity, size and perturbation. Canadian Journal of Fisheries and Aquatic Science, 43: 1789–1794.

Steele, J.H. 1973. Marine food chains. Oliver and Boyd, Edinburgh.

Stokes, T.K.A., Law, R., and McGlade, J. (Eds.) 1993. The exploitation of evolving populations. Springer Verlag, New York. 264 pp.

Thiebaux, M.L., and Dickie, L.M. 1992. Models of aquatic biomass size spectra and the common structure of their solutions. Journal of Theoretical Biology, 159: 147–161.

Thiebaux, M.L., and Dickie, L.M. 1993. Structure of the body-size spectrum of the biomass in aquatic ecosystems: a consequence of allometry in predator-prey interactions. Canadian Journal of Fisheries and Aquatic Sciences, 50: 1308–1317.

Washington, H.G. 1984. Diversity, biotic and similarity indices: a review with special relevance to aquatic ecosystems. Water Research, 18: 653–694.

4 NON-TARGET FISH POPULATIONS

Examine trends in abundance of non-target and non-assessed fish and shellfish species, taking into account other ICES reports on non-assessed species.

4.1 Introduction

Considerable attention has recently been paid to long-term changes in ecosystems and the role that anthropogenic factors might play in causing such changes (see, for example, ICES Århus Symposium, *Long Term Changes in the North Sea Ecosystem and their Causes*, July 1995). These studies have relied to a large extent on the analysis of time-series data. Some of the most extensive time series in the North Atlantic relate to zooplankton and in particular the Continuous Plankton Recorder (CPR) survey. Such data have been used to look for long-term changes in the zooplankton and to examine possible causal factors. For example, it has been shown that inter-annual variations in zooplankton abundance and copepod numbers is correlated with hydrographic/meteorological variation, including the proportion of northerly winds, and the position of the north wall of the Gulf Stream (Taylor, 1995; Frid and Huliselan, in press). Similar relationships have been shown in freshwater zooplankton and tree rings (see George and Taylor, 1995, for a review). The generality of these relationships implies that some form of climatic controls are important in the long-term regulation of some biological systems.

In some areas such changes seem to be occurring simultaneously in a number of components of the marine ecosystem (Aebischer et al., 1990). Long-term data on phytoplankton, zooplankton, macrofauna, and birds collected at Texel (NIOZ), Plymouth (CPR), and Helgoland (BAH) indicate a sudden change in the North Sea and Wadden Sea ecosystems in the late 1970s (Lindeboom et al., 1995). Different possible causes of the change and of the observed trends have been identified, e.g., variations in climate, quantity and quality of inflowing ocean water, river inputs, extreme winters and anthropogenic influences. The present explanation for the observed phenomena is that several causes together led to a major shift in the ecosystems in the late 1970s.

Compared with the 1940s and 1950s, the structure and functioning of the ecosystem of the Gulf of Riga (southeastern Baltic Sea) has been transformed. The observed changes include changes in phytoplankton species composition, increased primary productivity, invasion and population outbreak by the cladoceran *Cercopagis pengoi*, impoverishment of some bottom biocoenoses and the fish fauna. This appears to be due to a variety of causes including changes in environmental conditions as a result of long-term climatic variations, increased anthropogenic pollution, and fishing (Ojaveer, 1995; Ojaveer and Lumberg, 1995).

This section reviews existing time series on non-target species of fish and shellfish to explore whether there is any evidence of long-term changes and to what extent these may be the result of fishing activity. It then explores whether other temporal and spatial data can be used to assess fishing-induced changes. The focus is on species which have been independently identified as having characteristics that would lead them to respond, positively or negatively, to fishing effects and, as a sort of control group, a number of species not expected to show such changes.

4.2 Trends in Fish Populations

4.2.1 Landings data

In the *Atlas of North Sea Fishes* (Knijn et al., 1993) data series on landings of commercial species are abundant, but of the non-target species only data on total North Sea landings of rays (all species combined) and common dab *Limanda limanda* are given. The landings of rays decreased between 1960 and 1975 from 12,000 tonnes to around 5,000 tonnes. Between 1975 and 1988 they fluctuated around 5,000 tonnes per year. For common dab there was no clear trend over time, although during the 1970s and early 1980s the landings were high compared with the 1960s and the later 1980s. Inferences from such data should be made with caution, however, since landings are probably of limited value in determining trends in abundance of non-target species. They are also influenced by market forces, so that trends in landings may not reflect trends in abundance.

ICES (1994a) contains some data on the abundance of by-catch species in the fisheries for brown shrimp *Crangon* in the southern North Sea and the Wadden Sea from 1954 to 1988. Gobies and sea snail *Liparis liparis* declined in abundance over the first ten years of the time series. Hooknose *Agonus cataphractus* increased in the first half of the 1970s but declined again to a level comparable to that found before the increase. Sea

scorpion showed the opposite trend, with a decrease in abundance in the first half of the 1970s.

Cephalopods are likely to have an important role in the marine food web of the North Atlantic (ICES, 1994b). Even so, there are no data on trends in abundance available and the few landings data that are available contain major shortfalls. Under-reporting or total failure to report the landings are common, and when landings are reported, the species composition is frequently uncertain.

Very little information is currently available on the exploitation of deep-water fish species in the Northeast Atlantic (ICES, 1995a; 1996). These species are generally believed to be vulnerable to exploitation. For some of the species, fecundity is low and growth is generally believed to be slow. In addition some of the species aggregate in shoals where they can be caught in great numbers. Exploitation is currently increasing but due to a lack of information, it is difficult to estimate to what extent the species are over-exploited at present. ICES (1996) presents a summary of the available information on the state of the stocks for some of the more important species. For five out of the nine species for which data were available, the stocks were found to be declining and probably over-exploited.

The Study Group on Elasmobranch Fishes found that CPUE data from Scottish seine netters and trawlers suggested that the population of spurdog increased in abundance in the North Sea between 1967 and 1977 presumably due to a migration into the area, after which abundance declined to levels comparable to that observed in the early 1960s (ICES, 1995b). Spurdog is known to be highly migratory and the Working Group therefore concluded that the changes observed in the North Sea are unlikely to reflect the trends in the overall stock.

In a report on rays and skates on the Dutch continental shelf, Walker (1995) indicated that several species of rays and skates were common in Dutch territorial waters in the beginning of this century and there was a directed fishery on these species. During the 1930s a decline in landings was seen and shortly after the Second World War there were few rays left in Dutch waters. At the present time only two species of ray are caught in Dutch waters (*Raja radiata* and the stingray *Dasyatis pastinaca*), neither being commercially important.

In a further analysis, Walker and Heessen (1996) reported on long-term changes in ray populations in the North Sea. They analysed three data sets: landing statistics (1903–1993); International Bottom Trawl Surveys (1979–1993); and survey data collected in Dutch coastal waters between 1951 and 1994. They concluded that landings of skates and rays have decreased by more than half compared to the post-war level. In the southern North Sea, landings declined around 1948 and again in 1963, whereas in the northern and central North Sea the major decline started around 1965. Since the early 1970s,

total landings have remained more or less constant (see also Heessen and Daan, 1996). The starry ray *Raja radiata* increased in abundance in the central North Sea between 1970 and 1993. The thornback ray *Raja clavata* virtually disappeared from the Dutch coastal area in 1958. Between 1970 and 1993 no significant trend was observed for the other ray species. A decline in the abundance of *Raja batis* has also been observed in the Irish Sea (Brander, 1981).

A decreasing trend can also be observed in the FAO data on total landings of skates and rays between 1970 and 1992 in the Northeast Atlantic area (Nielsen, 1994). For the spurdog no trend was obvious between 1980 and 1992.

Simon and Frank (1995) assessed the skate fishery in the Northwest Atlantic. They also concluded that in that area the skate fisheries in the late 1960s and 1970s followed the general pattern found in most elasmobranch fisheries of high initial exploitation followed by a rapid collapse.

At the Netherlands Institute for Sea Research, a record has been kept of rare fish species and invertebrates caught by the Den Helder and Texel fishing fleet between 52° and 55°N and 3° and 7°E from 1930 to 1990 (Lindeboom, 1995). From a PCA analysis of the data on 16 fish species and 15 invertebrates, it was concluded that there were two distinct peaks in the number of rare species recorded. One occurred between 1949 and 1953 and the other in the second half of the 1960s. A recent analysis, in which the trends were compared with trawl effort, has indicated that the first peak coincides with the increasing otter trawl activity after the Second World War, while the second peak follows the introduction of beam trawls. After 1970 most of the species show a continuous declining trend. Several possible causes for the observed trends have been indicated: (a) the behaviour of the reporting fishermen, (b) pollution (e.g., effect of TBT on whelks), (c) eutrophication, (d) fisheries, and (e) effect of climate or incoming ocean water (Lindeboom *et al.*, 1995). Although all these causes may play a role, it was concluded from the close relationship between observed trends and fishing intensity that the occurrence of the following fish species was probably negatively influenced by fisheries: skate *Raja batis*, stingray *Dasyatis pastinaca*, dogfish *Scyliorhinus canicula*, smooth hound *Mustelus mustelus*, angler *Lophius piscatorius*, and greater weever *Trachinus draco*. [Note: For invertebrates the picture seems to be more complex. There is a close match between the observed trends in occurrence of sea mouse *Aphrodite aculeata*, red whelk *Neptunea antiqua*, lesser octopus *Eledone cirrosa* and helmet crabs *Corystes cassivelannas* and development of the trawl fishery.]

4.2.2   Survey data

Of more value than landings data are those from fishery-independent surveys, and for the larger-bodied non-assessed and non-target fish species in the North Sea a

number of survey data sets exist. In such surveys the abundance of all species caught in the trawl is usually recorded, and in the past few years efforts to explore these data sets to examine trends for species that are not subject to major fisheries have increased.

Greenstreet and Hall (in press) examined long-term changes in the structure and composition of the groundfish species assemblages in three regions of the northwestern North Sea. Scottish fisheries research vessel data collected over the period 1929 to 1953 were compared with groundfish survey data covering the period 1980 to 1993. Long-term differences in species assemblages were subtle, and were most apparent in the dominance structure. Multivariate analyses indicated long-term changes and between-area differences in the species composition for both the whole groundfish assemblage and the non-target species subset. These changes resulted from relatively small and subtle differences in the relative abundance of less common species, such as grey gurnard *Eutrigla gurnardus* and spurdog *Squalus acanthius*. In contrast, changes in the relative abundance of the more common species, such as Norway pout *Trisopterus esmarkii* and whiting *Merlangius merlangus* explained much of the between-area variation.

Further analysis of this data set shows that while the target species did not change in abundance between the two periods (e.g., cod, Figure 4.2.2.1) there was a reduction in the variability between sampling periods. In addition, there were marked decreases in the occurrence of several non-target species (e.g., grey gurnard, skate and spurdog; Figures 4.2.2.2, 4.2.2.3 and 4.2.2.4). Both these changes are in line with the predicted effects of fishing on target and non-target populations, respectively.

Analysis of the IBTS database suggests that, when density in the northwestern North Sea is considered, there was a marked increase in the population size of spurdog between 1970 and 1978, but that abundance has been at low levels both before and since (Heessen and Daan, 1996). These authors also note that grey gurnard has increased in abundance, albeit with considerable inter-annual variability, over the period 1970 to 1994. This suggests that the trends noted by Greenstreet and Hall (in press) for these two species in the northwestern North Sea were not just a local phenomenon. Heessen and Daan (1996) present data for seven other non-target species in addition to the spurdog, starry ray and grey gurnard already discussed. All show a high, although variable, degree of between-year variation; however trends are apparent for most of them. The populations of bullrout, poor cod, and lemon sole have gradually increased since the 1970s, while the abundance of four-bearded rockling, long rough dab and common dab has gradually risen since around 1980. No trend in the population size of bib is apparent.

On a more local scale, Rogers and Millner (1996) sampled the epibenthic fish assemblage off the east coast

of England between 1973 and 1995. Once again between-year variation was high and long-term trends were difficult to discern for some species (e.g., pogge *Agonus cataphractus*, lesser weever, and solenette). However, declines in abundance of butterfish and spotted ray were apparent over the whole time period, while eelpout seem to have declined since the mid-1980s. Some of this variation could be attributed to environmental fluctuations, particularly in water temperature and salinity.

Long-term changes in the composition of the fish community and the abundance of certain fish species are being followed in the Gulf of Riga (see Ojaveer, in press; Ojaveer and Gaumiga, 1995). While this is work in progress, some preliminary conclusions can be drawn:

1) Based on experimental bottom trawling (mesh size at the codend 4 mm) covering the depth range 5–55 m, it can be concluded that species richness of the total fish community has decreased during the period 1974–1995, varying between 3 to 10 species per haul in 1994–1995, and being higher in shallower areas in the coastal zone.

2) There have been declines in the abundance of glacial relicts, cold-water preferring demersal fish species and certain other fish species—fourhorned sculpin *Triglopsis quadricornis*, seasnail *Liparis liparis*, lumpsucker *Cyclopterus lumpus*, eelpout *Zoarces viviparus*, sea scorpion *Myxocephalus scorpius*, and smelt *Osmerus eperlanus*.

3) Two species of sticklebacks: three-spined stickleback *Gasterosteus aculeatus* and nine-spined stickleback *Pungitius pungitius,* are widely distributed over the Gulf of Riga and have maintained relatively high abundances or even increased in the fish community in shallower areas.

4) The abundance of other less commonly occurring, non-commercial fishes (sandeel, gobies) has greatly diminished and they are now recorded in experimental bottom-trawl catches only occasionally.

These changes appear to have taken place due to a complex of causes: fishing activity, other anthropogenic impacts, especially eutrophication and pollution of spawning and feeding grounds and long-term changes in climatic conditions.

As with all fish species, variation in the abundance of non-assessed species can arise through variation in recruitment. Fyke net sampling in the western Wadden Sea has been used to examine trends in recruit abundance of a number of species over the period 1972 to 1994 (Philippart *et al.*, in press). Pollack and turbot recruit abundance appears to have declined over this period, while flounder, twaite shad and lumpsucker have increased. Grey mullet abundance declined between 1972 and 1984 then subsequently increased from 1990

onwards. No trend in scad recruit abundance was apparent. In all cases inter-annual variance in recruit abundance was high and between-species covariance was low. Recruit abundance was influenced by a combination of density-dependent processes in nursery areas and year-to-year variation in environmental factors.

4.3 Trends in Shellfish Populations

No time series data for non-assessed or non-target shellfish were available to the Working Group. However, as shellfish live in much more intimate contact with the benthic environment than fish populations do, it may be legitimate to examine trends in benthic communities in areas where shellfish are exploited or where other demersal fisheries operate. Even then, however, there are only limited time series data available on the benthos of the North Sea with which to assess the long-term effects of fishing (Section 3.1, ICES, 1995c).

4.3.1 Northumberland (Northwest North Sea)

There are two extensive continuous time series on North Sea benthos: off Northumberland and in the Kattegat between Sweden and Denmark (see Austen *et al.*, 1991). The Northumberland monitoring programme, initiated by Buchanan (see Buchanan and Moore, 1986; Buchanan, 1993) involves biannual sampling at two near-shore stations. The inshore of these two stations lies within the Northumberland *Nephrops* (otter) trawl fishing grounds. This fishery has shown considerable development since the 1970s, when the dominant fishery was for herring. With the demise of the herring, effort switched to this inshore trawl fishery. There is, therefore, a time series of benthic data over a period when fishing effort has been known to increase from moderate to high levels. This provides scope for analysing the benthic time series against this change in fishing intensity (see Figure 4.3.1.1). During the first ten years of monitoring species abundances, there was a clear biennial cycle at this station with post-winter numbers which fell within a narrow range. Buchanan and Moore (1986) reported this as being the result of density dependent regulation of winter survival. From 1981 this biennial cycle broke down and macrofaunal abundances became much more variable. Following the density dependent hypothesis, the availability of food as indexed by the phytoplankton standing stock in the area—a crude index of organic flux to the benthos—was correlated with the number of macrofauna surviving the wintering. A significant correlation, explaining over 55% of the inter-annual variation, was found between phytoplankton abundance two years previously and macrobenthos numbers. For individual species abundances the relationship with phytoplankton two years previously often explained > 80% of the inter-annual variation.

In attempting to examine the Northumberland time series for any fisheries effects, one is faced with two difficulties. Firstly, as the relationship with primary production is so strong, one is endeavouring to use changes in fishing effort to explain part of the residual variation in benthos numbers. Secondly, the level of spatial resolution on the fishing effort data is low, which means one is relating the benthos at a single station to changes in effort over an entire ICES statistical rectangle. In spite of these problems, this work is proceeding. Initial analyses indicate that changes in fishing effort (hours) in ICES rectangle 39E8 are unable to explain a significant amount of the residual variation in the benthos data after the fitting of the phytoplankton index. This suggests that in this system, the increase in otter trawling has not led to a marked shift in total macrobenthos abundance over the last 20 years. However, this result must be treated with caution due to the above difficulties and the possibility that the levels of fishing occurring prior to 1970 had already induced changes in the benthos of this area.

4.3.2 Central North Sea

Kröncke (1990) compared data on the Dogger Bank from the early 1950s and the late 1980s. In the 1950s there was fishing effort in this area, but during the 1980s there was only very limited fishing in the area sampled. Changes in the macrofauna over this period were dominated by the increasing abundances of small, opportunistic, polychaetes, i.e., *Spiophanes bombyx*, *Scoloplos armiger* and *Magelona* spp. Total biomass declined by about 30%, mainly due to the disappearance of *Spisula subtruncata*. The distribution of the sea-urchin *Echinocardium cordatum* throughout the Dogger Bank remained unchanged. The increasing abundance of the opportunistic species during a period of reduced fishing pressure contrasts with the predictions originating from experimental studies on the short-term impacts of fishing on benthic communities (ICES, 1995d).

4.3.3 Baltic Sea

In the Gulf of Riga, long-term changes in abundance and biomass of mysids were studied between 1974–1991 (Kotta, 1995). The greatest *Mysis mixta* and *M. relicta* abundance and biomass values have generally been found after severe winters: 1976–1980 and 1985–1987. In contrast, the numbers of *Neomysis integer* were directly related to the summer temperature. Generally, the abundance and biomass of Mysidacea were highest in 1979–1980 and 1985–1991. Long-term dynamics (1950s–1990) of the macrozoobenthos community in the Gulf of Riga indicate a considerable increase in productivity and the dominance of species tolerant to pollution (e.g., *Macoma balthica*, *Mytilus edulis*, *Saduria entomon*), with a simultaneous decline in the abundance of less tolerant species. The decrease in the abundance of some numerically dominant species and structural changes in the macrofaunal community can be explained as the impact of eutrophication (Gaumiga and Lagzdinsh, 1995).

4.3.4 Southeast North Sea

Craeymeersch *et al.* (1995) investigated long-term changes in fauna by analysing by-catch data from the RIVO-DLO beam trawl survey (Tridens Sole Net Survey) for the period 1974–1991. It was concluded that some of the density changes found might be related to changes in fishing effort. On the transects near Helgoland and the Klaverbank there was a decrease in numbers of the quahog *Arctica islandica*, a species known to be affected by fisheries (Klein and Witbaard, 1993). In the coastal area off Scheveningen, Netherlands, more starfish *Asterias rubens* were found from 1985 onwards, corresponding with an increased number of fishing hours in that area.

Van der Veer *et al.* (1995) compared three data sets of trends in seven invertebrate species in the North Sea: the NIOZ archive, mentioned above; the RIVO-DLO by-catches from beam trawl surveys over the period 1970–1991; and the NIOZ fyke catches for a western Wadden Sea station, over the period 1960–1991. A few consistent trends were apparent across the data sets, for whelks and sea urchins in particular. For crabs, different patterns were apparent in the different data sets. No inferences about the causes of the patterns were drawn.

Using the records of finds recorded by local fishermen and reported to the NIOZ laboratory, De Vooys *et al.* (1993) examined spatial and temporal abundances of 15 macrobenthic species in the Dutch sector of the North Sea over the period 1930–1990. Of the 15 species examined, 13 were shellfish, and although some are target species in other parts of the North Sea, they are not the target species of the fishery in the area. Data were aggregated into 10-year groups and for five of the shellfish species (slender spindle shell *Colus gracilis*; common cuttlefish *Sepia officinalis*; common European squid *Loligo vulgaris*; masked (helmet) crab *Corystes cassivelaunus*; and the velvet swimming crab *Liocarcinus puber*) there was no clear temporal trend. For the European lobster *Homarus gammarus*, there was a steady decline over the entire period. The remaining seven species (whelks *Buccinum undatum*; red whelks *Neptunea antiqua*; edible crabs *Cancer pagurus*; Norway lobster *Nephrops norvegicus*; lesser octopus *Eledone cirrosa*; green sea urchin *Psammechinus miliaris*; purple heart urchin *Spatangus purpureus*) showed variable densities, but were consistently recorded in the earlier part of the period, and showed marked declines in the 1970s or 1980s. This pattern was also shown by the two non-shellfish species (Dahlia anemone *Urticina felina* and sea mouse *Aphrodite aculeata*) considered. These changes are attributed to a combination of climate change, pollution and fisheries impacts. In particular, the marked declines in the 1970s coincided with major increases in beam trawling effort in the area.

4.3.5 Extending the time series

All of the above studies involve comparisons of data covering periods of less than 50 years and therefore the earlier data are unlikely to be from 'unfished' communities. Fortunately there are data on the community composition of the benthos as a result of one-off sampling occasions from the latter part of the nineteenth century which may provide useful information for contrasts.

The quality of the earliest (pre-1910) data tends to be very variable. Firstly, there are studies undertaken before the adoption of quantitative sampling devices (grabs and cores). These are often published as species lists, with no numerical data, or with categorical estimates of numbers of individuals (e.g., "a few" or "abundant"). Secondly, there may have been problems with the taxonomy which was still inconsistent. Since many of the people recording the data were hobbyists, records tend to concentrate on certain sections of the benthos, with the collection of molluscs being particularly popular. However, these factors do not invalidate these earlier data. Comparisons of presence/absence data on species identified as being especially sensitive to impact by mobile gears are potentially valuable.

With the adoption of the Petersen and van Veen grabs in the early part of this century, quantitative data became available. Large-scale surveys were carried out in the North Sea by Denmark in 1883–1915 (Petersen, 1915) and the UK in 1923 (Davis, 1923) and data are also available for the English Channel (e.g., Ford, 1923).

Presently, one of the tasks within a project funded by the EC, on *Long term changes in the benthos of North Sea fishing grounds*, is the analysis of historic data from the North Sea. This work started in August 1995 and includes:

- Carrying out library searches for appropriate literature.

- Compiling a database of information available, and sampling positions.

- Identifying sites that have been sampled on more than one occasion.

- Carrying out "quality control" to aid comparability and account for changes in nomenclature.

So far 2,500 samples have been entered and three North Sea areas considered as being impacted by fishing have been chosen. As more data become available, further areas will be added. Data from the Channel are also being obtained and entered onto a separate database. As estimates of sensitivity of species are obtained, they will be entered, along with other details.

4.4 Conclusions

Compared to the landings data available for target species, there is a paucity of data on non-target and non-assessed fish species. The few time series of survey data reaching back to the 1930s indicate that the population abundance of sharks and rays has declined substantially, particularly common skate (Figure 4.2.2.3) and corroborate the trends in commercial landings (Walker and Heessen, 1996). The decline in these species could be attributed to fishing pressure, given the low reproductive rate of these animals (Brander, 1981). Some other non-target species (greater weever, angler, grey gurnard) appear to have declined since the 1930s but, except for the grey gurnard, the information is anecdotal. The more extensive survey data available since 1970 indicate fluctuations in abundance which may be related to natural variability of the environment and/or anthropogenic activities. None of the studies reported have shown a direct relationship between trends in abundance and the effect of fishing.

In attempting to assess the effects of fishing on non-target shellfish populations we remain hampered by the lack of time series data. We have looked at the benthic community present on the Northumberland *Nephrops* ground. While there was no trend in macrofaunal abundance, the importance of inter-annual variation in organic flux was confirmed (Buchanan, 1993; Frid *et al.*, in press). The causes of the change in primary production in the area, from one of low inter-annual variability to higher mean values and higher variability, around 1980 have not been explained. This increased variability and subtle shifts in community structure are also apparent in zooplankton time series from the Continuous Plankton Recorder (e.g., Colebrook, 1986) and the Northumberland station (Frid and Huliselan, in press). Similar patterns are also apparent in time series of phytoplankton and macrofauna from the northeastern North Sea (Austen *et al.*, 1991) and the Wadden Sea (Lindeboom *et al.*, 1995). It would appear that in many components of the North Sea ecosystem, the observed changes over the last 20 to 30 years are driven by a wide variety of mechanisms including climatic/hydrographic factors, eutrophication/changes in primary productivity as well as pollution and fishing. That fishing effects do not appear to have been the primary cause of change in any of the time series on benthic communities could be the result of impacts occurring prior to sampling. It is unlikely that these areas were 'pristine' at the start of the time series. There is therefore a need to extract historical data from sources other than formal time series in order to provide a true 'pre-fishing' baseline.

With any approach which uses occasional records, be they casual observations of rare species or dedicated sampling programmes for some other purpose but which are later used for temporal comparison, there remains the problem of separating true differences from apparent differences produced by the data collection protocols. This problem is perhaps most apparent when one considers data based on occasional observations. The frequency of occurrence of a phenomenon may not change but if awareness of interest in it is raised, more reports follow. From this, it follows that we may be able to attach more weight to cases of disappearances of species, such as the elasmobranchs from the southern North Sea, than appearances. If we are actively seeking reports of a species and they are not forthcoming, yet the species was regularly reported at an earlier time, we can be confident that it is less common.

There are several interesting time series available that may yield information on possible trends of non-target organisms. However, in general these series were not created to be used for this sort of analysis, and caution is required in drawing any inferences from them. A possible approach could be to combine several of these series and to analyse the common trends. Given the large number of factors operating in the natural world, and the complexity of their interactions, it is difficult to answer questions of ecosystem function by testing simple hypotheses. This is particularly the case for processes which operate over large spatial and/or temporal scales. The most convincing evidence of ecosystem-level changes comes when similar patterns are found in different areas and/or from different studies.

The Working Group believes that there is scope for the use of historical data sets and the few available time series to identify changes which may be the result of fishing activities. Significant progress in the use of these data will require development and application of analytical procedures appropriate for them.

4.5 References

Aebischer, N.J., Coulson, J.C., and Colebrook, J.M. 1990. Parallel long-term trends across four marine trophic levels and weather. Nature, London, 347: 753–755.

Austen, M.C., Buchanan, J.B., Hunt, H.G., Josefson, A.B., and Kendall, M.A. 1991. Comparison of long-term trends in benthic and pelagic communities of the North Sea. J. mar. Biol. Ass. UK., 71: 179–190.

Brander, K. 1981. Disappearance of common skate, *Raia batis* from Irish Sea. Nature, London, 290: 48–49.

Buchanan, J.B. 1993. Evidence of benthic-pelagic coupling at a station off the Northumberland coast. J. Exp. Mar. Biol. Ecol., 72: 1–10.

Buchanan, J.B., and Moore, J.J. 1986. A broad review of variability and persistence in the Northumberland benthic fauna 1971-1985. J. Mar. Biol. Ass. UK., 66: 641–657.

Colebrook, J.M. 1986. Continuous plankton record: the distribution and standing crop of the plankton of the

shelf and ocean to the west of British Isles. Proc. Roy. Soc., Edin., 88B: 221–237.

Craeymeersch, J.A., Rijnsdorp, A.D., Buijs J., and van Leeuwen, P. 1995. Long-term faunal changes: an analysis of by-catch data. *In* Report of the Benthos Ecology Working Group. ICES CM 1995/L:3. Annex 11.

Davis, F.M. 1923. Quantitative studies on the fauna of the sea bottom. No. 1. Preliminary investigations of the Dogger Bank. Fishery Investigations. Ministry of Agriculture, Fisheries and Food (ser. 2) 6(2). 54 pp.

De Vooys, C.G.N., Witte, J.IJ., Dapper, R., van der Meer, J., and van der Veer, H.W. 1993. Lange termijn veranderingen op het Nederlands continentaal plat van de Noordzee: trends in evertebraten van 1931–1990. NIOZ-Rapport 1993 -17, NIOZ, Texel, Netherlands. 68 pp.

Ford, E. 1923. Animal communities of the level sea bottom in the waters adjacent to Plymouth. Journal of the Marine Biological Association of the United Kingdom, 13: 164–224.

Frid, C.L.J., and Huliselan, N.V. In press. Far field control of long-term changes in Northumberland (NW North Sea) coastal zooplankton. ICES Journal of Marine Science.

Frid, C.L.J., Buchanan, J.B., and Garwood, P.R. In press. Variability and stability in the benthos: 22 years of monitoring off Northumberland. ICES Journal of Marine Science.

Gaumiga, R., and Lagzdinsh, G. 1995. Macrozoobenthos. *In* Ecosystem of the Gulf of Riga between 1920 and 1990. Ed. by E. Ojaveer. Estonian Academy Publishers, Tallinn.

George, D.G., and Taylor, A.H. 1995. Relationship between the abundance of zooplankton in Windermere and north/south shifts of the Gulf Stream. Nature, Lond., 316: 536–539.

Greenstreet, S.P.R., and Hall, S.J. In press. Fishing and groundfish assemblages structure in the north-western North Sea: an analysis of long-term and spatial trends. Journal of Animal Ecology.

Heessen, J.L., and Daan, N. 1996. Long-term trends in ten non-target North Sea fish species. ICES Jounal of Marine Science, 53 (6).

ICES. 1994a. Report of the Study Group on the Life History, Population Biology, and Assessment of *Crangon*. ICES CM 1994/K:3.

ICES. 1994b. Report of the Study Group on the Life History and Assessment of Cephalopods. ICES CM 1994/K:7.

ICES. 1995a. Report of the Study Group on the Biology and Assessment of Deep-Sea Fisheries Resources. ICES CM 1995/Assess:4.

ICES. 1995b. Report of the Study Group on Elasmobranch Fishes. ICES CM 1995/G:3.

ICES. 1995c. Report of the Working Group on Ecosystem Effects of Fishing Activities. ICES CM 1995/Assess/Env.:1.

ICES. 1995d. Report of the Study Group on Ecosystem Effects of Fishing Activities. ICES Cooperative Research Report No. 200. 120 pp.

ICES. 1996. Report of the Study Group on the Biology and Assessment of Deep-Sea Fisheries Resources. ICES CM 1996/Assess:8.

Klein, R., and Witbaard, R. 1993. The appearance of scars on the shells of *Artica islandica* L. (Mollusca, Bivalvia) and their relation to bottom trawl fishery. BEON-report 1993-2. N1OZ, Texel. 25 pp.

Knijn, R.J., Boon, T.W., Heessen, H.J.L., and Hislop, J.R.G. 1993. Atlas of North Sea Fishes: based on bottom-trawl survey data for the years 1985–1987. ICES Cooperative Research Report No. 194. 268 pp.

Kotta, I. 1995. Nektobenthos (Mysidacea). *In* Ecosystem of the Gulf of Riga between 1920 and 1990 Ed. by E. Ojaveer. Estonian Academy Publishers, Tallinn.

Kröncke, I. 1990. Macrofauna standing stock of the Dogger Bank. A comparison: II. 1951–1952 versus 1985–1987. Are changes in the community of the northeastern part of the Dogger Bank due to environmental changes? Neth. J. Sea Res. 25: 189–198.

Lindeboom, H.J. 1995. Protected areas in the North Sea: an absolute need for future marine research. Helgoländer Meeresuntersuchungen, 49: 591–602.

Lindeboom, H.J., van Raaphorst, W., Beukema, J., Cadée, G., and Swennen, C. 1995. (Sudden) changes in the North Sea and Wadden Sea: oceanic influences underestimated? German J. Hydrogr., (Suppl. 2): 87–100.

Nielsen, R. 1994. Fish and shellfish—shared Nordic marine stocks. TemaNord 1994: 519.

Ojaveer, E. 1995. Large-scale processes in the ecosystem of the Gulf of Riga. *In* Ecosystem of the Gulf of Riga between 1920 and 1990 Ed. by E. Ojaveer. Estonian Academy Publishers, Tallinn.

Ojaveer, H. In press. Environmentally induced changes in fish aggregations on the coastal slope of the Gulf of Riga. Proceedings of the 14th BMB Symposium.

Ojaveer, E., and Gaumiga, R. 1995. Cyclostomes, fishes and fisheries. *In* Ecosystem of the Gulf of Riga between 1920 and 1990. Ed. by E. Ojaveer. Estonian Academy Publishers, Tallinn.

Ojaveer, H., and Lumberg, A. 1995. On the role of *Cercopagis* (*Cercopagis*) *pengoi* (Ostroumov) in the Parnu Bay and the NE part of the Gulf of Riga. Proc. Estonian Acad. Sci. Ecology, 5: 20–25.

Petersen, C.G.J. 1915. On the animal communities of the sea bottom in the Skagerrak, the Christiania Fjord, and the Danish waters. Report of the Danish Biological Station to the Board of Agriculture, No. XXIII. 33 pp.

Philippart, C.J.M., Lindeboom, H.J., van der Meer, J., van der Veer, H.W., and Witte, J.IJ. 1996. Long-term fluctuations in fish recruit abundance in the western Wadden Sea in relation to variations in the marine environment. ICES Journal of Marine Science, 53 (6).

Rogers, S.I., and Millner, R.S. 1996. Factors affecting the annual abundance and regional distribution of English inshore demersal fish populations: 1973 to 1995. ICES Journal of Marine Science, 53 (6).

Simon, J.E., and Frank, K.T. 1995. An assessment of the skate fishery in Division 4VsW. DFO Atlantic Fisheries Research Document 95/71.

Taylor, A.H. 1995. North-south shifts of the Gulf Stream and their climatic connection with the abundance of zooplankton in the UK and its surrounding seas. ICES Journal of Marine Science, 52: 711–721.

van der Veer, H.W., Craeymeersch, J.A., van der Meer, J., and Rijnsdorp, A.D. 1995. Trends in invertebrates in the North Sea: A preliminary comparison of data sets. BEON-report nr. 1995-3.

Walker, P. 1995. Ecoprofile rays and skates on the Dutch continental shelf and North Sea. NIOZ publication no. 3053. Netherlands Institute for Sea Research, Texel.

Walker, P.A., and Heessen, H.J.L. 1996. Long-term changes in ray populations in the North Sea. ICES Journal of Marine Science, 53 (6).

5 QUANTIFYING VULNERABILITY

Refine and implement approaches for quantifying the susceptibility of species to fishing effects.

5.1 Introduction

With a view towards implementing the principles of sustainable development and the precautionary principle to fisheries management, there is a need to assess the effects of fisheries on non-target species. One approach would be to survey areas before exposure to fishing and again after a period of fishing. Unfortunately, in the North Sea (ICES, 1995a; Gislason, 1994; Hall *et al.*, 1993) and elsewhere (e.g., Thrush *et al.*, 1995) natural, 'unfished' areas or pre-fishing data are not available. Here we discuss the approaches to assessing the effects of fishing on individuals and means of extending this to the population level. In previous studies, the use of the terms vulnerability, sensitivity, susceptibility, fragility, etc., have often been used indiscriminately. This has led to some confusion and impeded comparisons between approaches. We, therefore, define the following terms as a clarification of the 1994 WGECO approach (ICES, 1995c) and apply them throughout.

Sensitivity = A measure of an organism's innate ability to withstand damage and avoid death when exposed to an external factor beyond the range of environmental parameters normally experienced (an impact).

Susceptibility = The likelihood that an organism will be affected by exposure to an external factor. A combination of sensitivity and the intensity of impact.

The first term is fixed, whereas the second term varies with intensity of impact. For example, a sensitive organism in a non-impacted area will not be susceptible.

The underlying rationale for an analysis of the effect of fishing on a species is that, in some sense or other, we wish to determine effects on populations. With the exception perhaps of marine mammals, there are few species for which most people view the death of an individual as cause for concern *per se*. Rather, it is the idea that the levels of mortality generated by fishing are sufficiently high to put populations at risk which motivates our endeavour. For this risk we use the term vulnerability, defined below.

Vulnerability = The likelihood that a population will be affected by exposure to an external factor.

It follows, therefore, that we require tools to assess the sensitivity and susceptibility of species and methods for extending this to the population level.

5.2   Assessing Sensitivity and Susceptibility

As a consequence of an impact, species may either benefit (reactive species) or suffer (susceptible species). The increase of the polychaete species *Capitella capitata*, for instance, has been frequently cited as a characteristic of organic enrichment by eutrophication or hydrocarbon pollution (Pearson and Rosenberg, 1978; Sanders *et al.*, 1980; Parra *et al.*, 1994). Gravel extraction, on the other hand, negatively affects large bivalves such as *Dosinea exoleta* (van Moorsel, 1994). Here we deal only with species that suffer.

The Benthos Ecology Working Group (BEWG) has outlined criteria that should make it possible to provide a list of the most likely sensitive species for each habitat within each biogeographic province (ICES, 1994a; 1995a). The BEWG recognized that a complication that must be dealt with in this approach is that, for most regions of interest, the seabed has been trawled for so many years that it is difficult to know which species are absent (or have reduced density) due to physical disturbances and which species are absent due to some other habitat requirement that is not being met in that region. The species which are present in heavily trawled areas might be those which have survived and are not, therefore, very susceptible to trawl damage.

In order to identify sensitive species, the BEWG separated the effects on species into two main categories: direct effects of trawls on individuals, and long-term effects on the population of a species. Obviously, if individuals are not sensitive, the population will not be vulnerable. On the other hand, if individuals are susceptible in some way, then the vulnerability of the population needs to be assessed.

Direct effects on individuals include:

*   damage to the body, the tube or the burrow;
*   removal from their burrow or tube;
*   removal from the substratum;
*   burial by resuspended sediment;
*   clogging of feeding structures.

The population characteristics of sensitive species which may become vulnerable are:

*   low growth rate;
*   low regenerative capability;
*   low fecundity or infrequent recruitment;
*   low mobility of reproductive propagules;
*   loss of specialized habitat; and
*   narrow substrate tolerance.

Populations experiencing high mortality would be unlikely to recover if they have low fecundity. Species with a narrow substrate tolerance that are removed from the substratum would only recolonize the type of habitat that is being repeatedly disturbed and, given frequent removals, the species population would eventually be so reduced that a breeding population would not be large enough to maintain itself.

In contrast, long-term effects on species that suffer some direct effects from physical disturbances are unlikely if they exhibit some or all of the following characteristics:

*   highly mobile;
*   freely burrowing;
*   rapid burrowing capability (as in some bivalves);
*   wide tolerance to natural disturbance;
*   high densities, rapid reproduction;
*   high fecundity.

To provide a complete list of susceptible species, one needs:

a) to determine whether an individual will or will not suffer when exposed to fishing. The most obvious species suffering from fishing impacts are the commercial target species. But all by-catch fish species suffer as well. In fact, almost all fish returned to the sea after processing of the catch will die (de Groot and Lindeboom, 1994). Damage, mortality, and reduction in numbers have been reported for many benthic species (see, e.g., Annex 8 (ICES, 1994a) and Table 3.3.2 (ICES, 1994b)). But all species occurring in the North Sea should be checked for susceptibility. Entanglement of seabirds is likely to occur in those areas that support both fixed net fisheries and diving seabirds (ICES, 1995c). Grey and common seals may become trapped in towed gears, but most seal by-catch mortality is attributed to fixed type gears. Harbour porpoises are taken in gill nets and trapped in pound nets (ICES, 1995c).

b) to indicate whether sensitive species are likely to be vulnerable. This is based on the population characteristics given above. Data on life-history parameters of fish species are being compiled, but data for benthic species remain scattered through the literature. Information on selected benthic species is given by, e.g., ICES (1994b), Rees and Dare (1993), and Holtmann *et al.* (in press), but there is a need to further review and summarize the literature, including 'grey' literature. The Benthos Ecology Working Group should be asked (a) to provide information on the individual risk of being damaged or killed, and (b) to compile information on the population characteristics. Population parameters that help indicate if a bird species is sensitive to extra mortality are known for most North Sea species (Dunnet *et al.*, 1990). Most marine mammals show similar life history characteristics to seabirds (relatively long lives, long period of immaturity, but even lower reproductive rate).

c) to produce maps showing the distribution of sensitive species (but see remark above concerning absence of species). Information on the distribution of fish species is available from ongoing programmes

(International Bottom Trawl Survey; ICES Beam Trawl Surveys) (Knijn et al., 1993; ICES, 1995b). The IBTS data are stored in a database held by ICES. The ICES North Sea Benthos Survey provides data on the distribution of the benthic infauna (Künitzer et al., 1992), and data are held in a database at Yerseke, The Netherlands (BEDMAN; ICES, 1994a; 1995a). Knowledge of the larger infauna and epifauna is restricted. In their 1995 meeting the BEWG proposed a new survey focusing on this part of the benthic fauna. Maps of cetaceans have recently been published by Northridge et al. (1995). The distribution of seabirds is available from Stone et al. (1995) and from the European Seabirds at Sea Group.

d) to produce maps indicating the intensity and frequency of disturbances for different types of gears. Information on fishing intensity is available for different fisheries for each ICES statistical rectangle (see, e.g., Figure 5.6 in ICES, 1994b; McDonald et al., 1994; Lindeboom, 1995). But there might be large variations within an ICES statistical rectangle (Rijnsdorp et al., 1994) and, thus, information on fishing intensity on a smaller scale than the ICES rectangles is needed.

Given a) to d), the susceptibility of each species can be assessed. This information is necessary before we can begin to determine whether or not the population of a species will be directly affected. Two approaches to assess the vulnerability are given below. However, additional information will be required before any indirect effects can be determined.

5.3 Assessing Species Vulnerability: Population Modelling Approach

Fisheries science has mainly been concerned with the assessment of populations of fish species that are targeted by commercial fishing operations. This assessment involves techniques that estimate vital population parameters such as abundance, recruitment, and fishing mortality based on catch-at-age data. The classification of catch data into age classes usually improves the quality of inference but is associated with high costs. Reliable information at age is only available for the most important target species because they represent the bulk of the landed catch and fisheries institutes naturally focus their attention and resources on them. Here we investigate how the accumulated knowledge of fisheries assessment and management can be extended to evaluate vulnerability for non-target species. We mainly consider "quick and dirty" methods that can be applied to provide yard-stick estimations when limited information is available.

5.3.1 Analyses based on catch-at-length data

Although most fisheries assessment techniques employ age-based analyses, length-based analyses are increasingly used. Although catch-at-length data are

easier and cheaper to collect, such methods still require comprehensive time series of catch data and auxiliary information to provide reasonable starting values for a large number of model parameters. For most non-target fish species, sophisticated length-based methods (Schnute, 1987; Sullivan et al., 1990) are too data demanding to be of any use. When modest catch-at-length data and some biological information are available, the less exacting cohort analysis at length (LCA) can be used. LCA is a convenient way of converting an equilibrium catch-at-length distribution to an estimate of population size and fishing mortality at size (Jones, 1984). The major assumptions associated with LCA are that the population is in a quasi-steady state over time and grows according to the von Bertalanffy growth curve. It also assumes that recruitment to the fishery is knife-edged and constant and that natural mortality is known.

LCA has been applied to three by-catch species (common dab Limanda limanda, grey gurnard Eutrigla gurnardus, and dragonets Callionymous sp.) to estimate the level of fishing mortality imposed by the Dutch beam trawl fleet (MacDonald et al., 1994). The analysis was based on discard data collected by observers on board the commercial vessels and assumed that these data provide a reasonable estimate of the total catch and that discarded fish did not survive. A similar study has been carried out for the North Sea and Scottish west coast populations of common dab and long rough dab Hippoglossoides platessoides based on Scottish discard information from trawls and seines (Stratoudakis, 1996). The results of both these analyses have shown small to moderate levels of F, lending support to Pope's postulate (MacDonald et al., 1994) which suggests that fishing mortality for non-target fish species will be equal to or below the one imposed on the target species of the fishery under consideration.

5.3.2 Index of relative vulnerability for populations

One quantitative measure of the relative vulnerability of a species population to fishing can be obtained by comparing the current level of fishing mortality to the one under which the spawning population per recruit would be reduced below a certain proportion of the virgin level. As an illustration, a critical value of F could be defined as the value which would reduce the population to 5% of its unfished fish level. MacDonald et al. (1994) suggest a way of inverting the LCA technique to produce estimates of the critical F values given the length structure of the population and the current level of F. Applying this technique to the common dab, grey gurnard, and the dragonet populations and knowing that other fish species with similar life history characteristics can sustain even higher levels of F, they concluded that these species are currently in little danger of suffering population collapses due to fishing. If these estimations are correct, the levels of fishing mortality would have to

increase by three or more orders of magnitude to reach the above-defined critical value.

5.3.3 Extension to non-fish species

LCA can also be applied to epifaunal invertebrates that are caught as by-catch by demersal fishing gears (crustaceans, large molluscs, and echinoderms). In recent years discard surveys have been extended to collect information on large invertebrates and LCA could provide a first indication of the levels of the impact of fishing on such species. However, there are a number of smaller epibenthic or infaunal animals (annelids, bryozoans, cnidarians, etc.) for which LCA is not applicable because they are not retained in the net, yet they are being killed by the passage of demersal gears (Kaiser and Spencer, 1996). In this case it is important to remember that traditional fisheries concepts, such as fishing mortality, should also include the mortality imposed by fishing on that part of the population that is damaged by the gear and is left dead or dying on the seabed.

Although for species that are not actually caught, reconstruction of the population at size by catch data is impossible, the principle used in providing an index of relative vulnerability in the previous section can still be applied (ICES, 1994b). The underlying idea remains the estimation of a critical F value which, if exceeded, would put the population under the risk of collapse (potential jeopardy) given unfavourable recruitment conditions. This critical F value is estimated based on modest life history information for the species. If some estimate of the current F is also available, then the two figures can be compared to provide some tentative assessment of vulnerability.

When actual levels of fishing mortality cannot be directly estimated from catch information, an indication of F can be obtained by considering fishing mortality to be proportional to the product of two more tractable components (ICES, 1994b):

1) the probability of encounter of an individual of a species with the gear; and

2) the probability of being killed as a result of the encounter.

The probability of encounter can be estimated if data on time spent fishing and spatial distribution of effort and the area covered are available. The probability of being killed can be estimated by experiments measuring the abundance of species prior to and after the passage of the gear. With these data a swept area technique can be applied. An illustration of the application of this technique in its crudest form is given in the following section.

5.3.4 Application of the swept area method to three benthic infaunal species

Beam trawl effort data from the Dutch fleet are available as days at sea per statistical rectangle for 1989. To apply the swept area method, days at sea are transformed to hours fishing by accepting an average of 13.6 hours fishing per day (MacDonald et al., 1994). The fleet is separated into two distinct groups with different gear characteristics. The first group contains larger vessels (>300 bhp) that use two beams of 12 m each and have an average towing speed of 7 knots. More recent information (Rijnsdorp, pers. comm.) suggests that this is probably an overestimation and the average speed was, therefore, reduced to 6 knots. The second group contains smaller vessels (<300 bhp) that have beams of 4 m each and have an average towing speed of 5 knots.

Indices of abundance for infaunal benthic species are available from the 1986 North Sea Benthos Survey (NSBS). The survey was carried out in the central and southern North Sea (IVb and IVc, respectively) between spring and early summer with box samplers and van Veen grabs (Künitzer et al., 1992). The data are available as numbers per m² for each sampling station. Stations are attributed to statistical rectangles in the North Sea and, in cases where more than one observation is available by rectangle, average indices are created. The analysis is restricted to a few relatively abundant species in the area fished by beam trawls and for which additional experimental data on the proportion killed due to the passage of the gear are available. These species are the ophiuroid *Amphiura filiformis*, the echinoid *Echinocardium cordatum*, and the bivalve mollusc *Tellimya ferruginosa*. An attempt was made to include information for the edible crab *Cancer pagurus* using the International Beam Trawl Surveys, but the degree of overlap between the survey data and the Dutch effort data was too low to allow any reasonable estimation.

To use the above data in the swept area method, a number of fundamental assumptions have to be made. Among these the most important are that the annual fleet effort within a rectangle is randomly distributed and that the index of abundance obtained from the benthos survey provides a reasonable estimate of the density of a species for the entire rectangle. Both of these assumptions are clearly violated. Recent studies on the spatial distribution of fishing effort within rectangles (Rijnsdorp et al., 1994) have demonstrated that fishers tend to distribute their effort patchily. In addition, it is far fetched to expect that a few grab samples can provide estimates of abundance that are representative of an area 30 miles x 30 miles. Data on fishing effort and species abundance at a much finer spatial resolution are needed to address the issue satisfactorily. However, such detailed information was not available to the Working Group and, therefore, we have used the swept area method on the scale of ICES statistical rectangles, bearing in mind the caveats mentioned above.

In the present study, the swept area per rectangle (SAR) for vessels over 300 bhp is given by the equation:

$$SAR = 2 \times 12 \text{ (m)} \times 6 \times 1852 \text{ (m hr}^{-1}) \times \text{Effort (hr)}.$$

The same equation is applied for the smaller vessels after reducing the size of the beams and the towing speed to 4 m and 5 knots, respectively. Assuming that everything in the path of the gear is killed and the species is randomly distributed within the statistical rectangle, the local fishing mortality (Fr) in an ICES rectangle is equal to the area swept by the gear in the rectangle (SAR) divided by the total area of the rectangle (AR):

$$Fr = SAR / AR.$$

The total fishing mortality (F) for a species in the entire North Sea from the gear under consideration can be obtained by weighting local mortality (Fr) with the local index of species abundance (Ur) (MacDonald *et al.*, 1994):

$$F = \frac{\sum_r (U_r F_r)}{\sum_r U_r}$$

As already mentioned, the above equation assumes that all individuals in the path of the gear are killed and represents, therefore, an upper limit of F. The actual proportion of the local population which will be killed by the gear is almost always less than one and, unless data are available, total F estimated by swept area is bound to provide an inflated estimate of fishing mortality. The method is initially used assuming that all individuals die when they encounter the fishing gear. In a second step, experimental data estimating mortality due to the passage of beam trawls (Bergman *et al.*, 1996) are used to provide more realistic estimates of F. The resulting levels of total fishing mortality from beam trawls in the North Sea are given in Table 5.3.4.1.

As expected, the results of Table 5.3.4.1 suggest that higher mortalities for all species are generated by the larger beam trawls for which both fishing effort and area coverage are higher. It is interesting that when it is assumed that all individuals in the trawl path die, all species seem to have similar F per gear category, apart from *Amphiura* in the smaller beam trawl fishery. The distribution of this species is known to be limited by the 30 m isobath in the southern North Sea (Künitzer *et al.*, 1992) and, therefore, it does not occur in a relatively large part of the area covered by the smaller beam trawls. The effect of this limited distribution seems to be diluted when the wider area in which large beam trawls operate is taken into consideration. In addition, the large reduction in fishing mortality observed for *Amphiura* when the assumption that all individuals that encounter the gear die is relaxed, highlights how misleading the lack of such information can be.

To assess the relative vulnerability of these infaunal species a theoretical value of critical F also has to be estimated. Although some information on life history parameters was available for the above three species, it was insufficient for a reliable estimate of critical F. Unfortunately, the lack of such data does not allow us to draw any inference on the relative vulnerability of the studied species. It does, however, highlight the fact that if such approaches are to be pursued in future, more reliable estimates on a number of life history parameters for benthic species are needed. According to the methodology proposed for the estimation of theoretical F in ICES (1994b), information on the following life history parameters is required:

- natural mortality;

- length at first maturity;

- length at first capture;

- growth parameters.

It is clear that some information on the life history of benthic species already exists. Nevertheless, it is also clear that the expression of life history parameters in the above-mentioned context is not often followed in benthic ecology. One possibility would be to use existing P/B ratios from populations in unfished areas to estimate the level of natural mortality for various species. More details on this issue can be found in ICES (1995c). An alternative approach would be to use data on longevity of the species. Finally, as increasingly more data become available on the damage that demersal gears impose on individuals, it seems feasible to combine and generalise such information by the application of appropriate statistical techniques. This can be done by modelling the mortality caused by the passage of gear as a function of explanatory variables that broadly capture the categories under which benthic species can be grouped in relation to their sensitivity. The results of such models can then be applied to species for which no such information is available.

5.4 Spatially Explicit Population Modelling (SEPM) Approach

At the scale of, for example, an ICES statistical rectangle, it is clear that fishing effort can be distributed in a number of ways. At one extreme there may be a totally uniform distribution of effort across the rectangle, or at the other extreme, only one, or a few, fished sites with a very localized effect. However, in reality, the distribution of fishing effort is likely to be between these extremes (Rijnsdorp *et al.*, 1991). The spatial distribution of benthos is also heterogeneous, but is patchy at scales smaller than the scale of distribution of fishing impacts. Therefore, we may gain additional insights into the mechanism via which fishing affects benthos outlined below. It is one of a number of modelling approaches (cf. previous section) which may be used to assess the vulnerability of benthic populations to fishing disturbance.

If the benthic ecosystem is viewed as a number of discrete interlinked areas (or elements) (Forman, 1983), then a population model can be applied to each element. If the parameters of these models vary, depending on the history of the element, i.e., mortality is higher in elements subjected to impact by fishing gear, then differences between elements can be explicitly accounted for. Summation over all of the elements gives a resultant response for the whole area. Such methods have been used to model disturbances to forests, allowing inclusion of spatial heterogeneity, and have been shown to give a high degree of realism and have led to a greater understanding of the dynamics of the system (Shugart and Seagle, 1986). A similar approach might be considered as a method of modelling the effects of fishing.

5.4.1 Features of the model

As a starting point, a model (Figure 5.4.1.1) is described which contains parameters for each element. The dimensions of each element (d) within the area depend on the scale of resolution required.

A discrete time model is proposed. When unfished, if individuals move at random and movement rates differ between elements, then their spatial distribution will become heterogeneous. By a choice of appropriate parameters it should be possible to match this degree of patchiness to that observed in the field. Then a set of elements may be randomly selected as those that are to be impacted by fishing. They may be distributed in a linear fashion, to simulate the passage of a trawl, or in any other pattern to represent the impact of a gear. The numbers of individuals in the impacted area are then reduced using a mortality term and the recovery from the fishing disturbance observed. This simulated fishing impact can be repeated at varying spatial and temporal intervals to assess the impact of multiple fishing events.

The following parameters are required:

D_f: Mortality of individuals as a result of fishing impact

> D_f describes the proportion of individuals within impacted elements that is killed as a result of the fishing activity that occurs in a time interval. This value is zero in all elements apart from the impacted ones. Vulnerable populations, that is those made up of sensitive individuals, would be expected to suffer higher proportional mortalities, so D_f for them may be up to 100%.

To derive this parameter, by-catch data, direct mortalities and survival rates are available from, for example, the CEC IMPACT-I study which examined the effects of beam trawling on both epi- and infauna (de Groot and Lindeboom, 1994). The impacts of otter trawling have been considered by the Dutch BEON group (BEON, 1990; 1991; 1992). However, these data only apply to species that are brought on deck. An alternative

assessment method of the mortality for small individuals, such as polychaetes, would require a different sampling regime, such as taking samples before and after trawling to assess effects on benthic populations (e.g., Kaiser and Spencer, 1996).

D_n: Natural mortality

> D_n describes the proportion of individuals in each element dying as result of natural mortality that occurs in each time interval. Some data are already available to describe this (e.g., Rees and Dare, 1993). However, the life history parameters for many species are unknown.

Birth of individuals

> B denotes the per capita birth rate.

Movement

> M describes the proportion of individuals in an element which move out of the element. Direction of movement of individuals is random. There are two rates of movement, one for disturbed, and another for undisturbed elements.

The group was not aware of any data that would allow parameterization of between-element movement terms. However, benthos have been shown to move in response to stresses other than those caused directly by fishing, for example, in anoxic conditions (reviewed in Diaz and Rosenberg, 1995), and such studies may provide useful data to define this term.

Movement of predators and scavengers (fish, crustaceans) onto trawled areas has already been shown (Kaiser and Spencer, 1995), and there is evidence that infaunal colonists also move rapidly onto azoic or disturbed sediments (e.g., Arntz and Rumohr, 1982; Zajac and Whitlatch, 1982; Arnoux et al., 1985; Bonsdorff et al., 1986). Such studies are useful in providing data on which this parameter can be based, although data on rates of movement appear to be less readily available.

R: Recruitment from a remote population

> R describes the proportion of individuals moving into an element from remote populations.

If adjacent elements are impacted, the only source of colonists may be from remote elements. An example of this would be the case of destruction of a whole Sabellaria reef. Since the mobility of many taxa is low, this would mean that an influx of planktonic larvae would be necessary. The extent to which larvae can be moved around in the water column depends on such factors as the water current speed, duration of the planktonic phase, and the ability of larvae to delay settlement until a suitable substrate is found.

The model could be parameterized by working out the time span of the planktonic larval stage, and then multiplying this by the residual current in the area under study, thus giving estimates of maximum distance travelled. However, flow dynamics are complex, and have the potential to cause large errors in any observations of this type. A more robust approach may be to carry out experimental exposure of azoic sediments, and thus estimate the extent of, and time scales of, colonization.

5.4.2 Usefulness of this approach

The principal advantage of the application of an SEPM model to fishing impacts is the ability to examine novel scenarios which would be impossible to do in the real world experiments. Such applications include modelling the effects of unimpacted fishery exclusion areas and defining suitable separation distances for control and impacted areas in experimental field studies on fishing effects. Such an approach allows exploration of the effects of sensitivity of species under different fishing strategies. It permits the inclusion of organisms with different life history parameters and mobilities. This should improve our understanding of the system being studied, and in particular the effects of varying faunal life history parameters and sensitivity to impacts, distributions of effort and fishing intensity.

The usefulness of any model is ultimately constrained by the quality and availability of the data needed for parameterization and the assumptions made in formulating the model. As was noted above, the data on life history characteristics of benthic organisms are sparse. As such data become available, the accuracy of the parameters will increase.

A principal drawback to the application of a SEPM to the marine benthos is the representation of the whole community in each model element. In the case of SEPMs used for forest ecosystems the element typically represents tree-sized spaces, so the scale of the patch matches the scale of the organism. The inclusion of parameters for all but the most abundant species is recognized as being impractical (ICES, 1995c). The model could be developed from a single species model by the addition of more than one dominant or functionally important species.

5.5 Genetic Effects on Fish Populations

Other aspects of species response to fishing are the changes in demographic characteristics of populations that fishing might impose. For the most part, work on this issue has focused on effects on target populations, since it is here that we would expect changes to be most apparent. There is a large number of life-history characteristics that may be affected by fishing, and these have been comprehensively discussed by Hutchings and Morris (1985), Jennings and Beverton (1991), Beverton (1992), and Stokes et al. (1993). Unfortunately, however,

analysis of many of these traits is likely to prove intractable owing to difficulties in estimation and the absence of historical records which would allow the construction of time series. In view of these difficulties, a new study is being initiated in France which has identified a subset of parameters which can be subjected to univariate and multivariate analytical approaches adopted by Stearns (1992), Pontier et al. (1993), and Harvey and Pagel (1991).

Data have now been assembled for approximately 50 separate (primarily commercial) fish stocks from the North Atlantic and the Mediterranean and preliminary analyses have been undertaken. These results suggest that there is a trade off between growth rate and reproductive life span which depends on the length of adult life. Thus, reduction in adult life span imposed by fishing may lead to changes in growth rate and age at first maturity (for example). Another preliminary result suggests that when ordinations are performed which place all species populations in a multi-dimensional space defined in terms of the selected set of life history traits, the volume of space occupied has decreased. It would appear, therefore, that the range (or diversity) of the life history attributes exhibited by populations has decreased over time. In the short term, the kind of phenotypic plasticity exhibited by individuals is not likely to alter the probability that a population will persist at low density. However, in the longer term selective pressure may be imposed which leads to reduced body sizes and genetic diversity. Ongoing work by the French group is aimed at 1) describing the changes in the life history composition between fish communities under different levels of exploitation, and 2) determining the limits of phenotypic plasticity and the proximity of populations to these limits.

5.6 Conclusions

1) All of the approaches outlined above require good data sets from which suitable parameter estimates can be derived. The absence of such data sets is a major deficiency that needs to be addressed. In particular, finer scale data on spatial distributions of fishing effort and benthos are required.

2) With respect to the life history parameters necessary to assess population vulnerability, we are still some way from arriving at a suitable set of metrics for benthic species. It is often difficult to see how life history metrics appropriate for fish, (e.g., L_∞) can be applied to benthic organisms. Nevertheless, some form of mapping to these parameters is necessary if progress is to be made.

3) Some direct estimates of mortality of non-target species due to mobile gears are now available and have been used above. However, the data available at present are limited to a few species and additional experimental data are required on mortality rates of sensitive species when exposed to a variety of gears

30

before these approaches can be applied with confidence.

4) The potential of fishing to exert an evolutionary selective pressure on populations remains an important issue that warrants further investigation.

5.7   References

Arnoux, A., Stora, G., and Diana, C. 1985. *In situ* experimental study of the evolution and recolonization of polluted sediments. Marine Pollution Bulletin, 16: 313–318.

Arntz, W.E. and Rumohr, H. 1982. An experimental study of macrobenthic colonization and succession, and the importance of seasonal variation in temperate latitudes. Journal of Experimental Marine Biology and Ecology, 64: 17–45.

BEON. 1990. Effects of beam trawl fishery on the bottom fauna in the North Sea. BEON Report No. 8.

BEON. 1991. Effects of beam trawl fishery on the bottom fauna in the North Sea II: the 1990 studies. BEON Report No. 13.

BEON. 1992. Effects of beam trawl fishery on the bottom fauna in the North Sea III: the 1991 studies. BEON Report No. 16.

Bergman, M.J.N., van Santbrink, J.W., Fonds, M., Craeymeersch, J.A., Fonteyene, R., and Damm, U. 1996. Direct mortality of invertebrate macrobenthos due to trawling with commercial beam and otter trawls in the North Sea. Concept data report 1995. IMPACT-II, Subproject 2 (Contract No. AIR 2 94 1664).

Beverton, R.J.H. 1992. Patterns of reproductive strategy parameters in some marine teleost fishes. Journal of Fish Biology, 41 (Suppl.B): 137–160.

Bonsdorff, E., Leppäkoski, E., and Östermann, C.-S. 1986. Patterns in post-impact successions of zoobenthos following physical disturbance in the Northern Baltic Sea. Publ. Water Res. Inst. Finland, 68: 117–121.

de Groot, S.J., and Lindeboom, H.J. (Eds.) 1994. Environmental impact of bottom gears on benthic fauna in relation to natural resources management and protection of the North Sea. NIOZ-Rapport 1994-11; RIVO-DLO Report CO26/94.

de Groot, S.J., and Lindeboom, H.J. 1994. Environmental impact of bottom gears on benthic fauna in relation to natural resources management and protection of the North Sea. NIOZ-Rapport 1994-11/RIVO-DLO Report CO26/94.

Diaz, R.J., and Rosenberg, R. 1995. Marine benthic hypoxia: A review of its ecological effects and the behavioural responses of benthic macrofauna. Ocean Mar. Biol. Ann. Rev., 33: 245–303.

Dunnet, G.M., Furness, R.W., Tasker, M.L, and Becker, P.H. 1990. Seabird ecology in the North Sea. Netherlands Journal of Sea Research, 26: 387–425.

Forman. 1983.

Gislason, H. 1994. Ecosystem effects of fishing activities in the North Sea. Marine Pollution Bulletin, 29:520–527.

Hall, S.J.; Robertson, M.R., Basford, D.J., and Heaney, S.D. 1993 The possible effects of fishing disturbance in the Northern North Sea—an analysis of spatial patterns in community structure around a wreck. Netherlands Journal of Sea Research, 31: 201–208.

Harvey, P.H., and Pagel, M.D. 1991. The comparative method in evolutionary biology. Oxford University Press, Oxford.

Holtmann, S.E., Groenewold, A., Schrader, K.H.M., Asjes, J., Craeymeersch, J.A., Duineveld, G.C.A., van Bostelen, A.J., and van der Meer, J. In press. Atlas of the zoobenthos of the Dutch Continental Shelf. Ministry of Transport, Public Works and Water Management, North Sea Directorate. Rijswijk, The Netherlands.

Hutchings, J.A., and Morris, D.W. 1985. The influence of phylogeny, size and behaviour on patterns of covariation in salmonid life histories. Oikos, 45: 118–124.

ICES. 1994a. Report of the Benthos Ecology Working Group. ICES CM 1994/L:4.

ICES. 1994b. Report of the Working Group on Ecosystem Effects of Fishing Activities. ICES CM 1994/Assess/Env:1.

ICES. 1995a. Report of the Benthos Ecology Working Group. ICES CM 1995/L:3.

ICES. 1995b. Report of the Beam Trawl Survey Study Group. ICES CM 1995/G:5.

ICES. 1995c. Report of the Study Group on Ecosystem Effects of Fishing Activities. ICES Cooperative Research Report No. 200. 120 pp.

Jennings, S., and Beverton, R.J.H. 1991. Intraspecific variation in the life history tactics of Atlantic herring stocks. ICES Journal of Marine Science, 48: 117–125.

Kaiser, M.J., and Spencer, B.E. 1986. The effects of beam trawl disturbance on infaunal communities in different habitats. Journal of Animal Ecology, 65(3).

Kaiser, M.J., and Spencer, B.E. 1995. A preliminary assessment of the immediate effects of beam trawling on a benthic community in the Irish Sea, pp. 87–94 In Environmental impact of bottom gears on benthic fauna in relation to natural resources management and protection of the North Sea. Ed. by S.J. de Groot and H.J. Lindeboom. NIOZ-Rapport 1994-11; RIVO-DLO Report CO26/94.

Kaiser, M.J., and Spencer, B.E. 1996. The effects of beam-trawl disturbance on infaunal communities in different habitats. Journal of Animal Ecology, 65(3).

Knijin, R.J., Boon, T.W., Heessen, H.J.L., and Hislop, J.R.G. 1993. Atlas of the North Sea Fishes: based on bottom-trawl survey data for the years 1985–1987. ICES Cooperative Research Report. No. 194. 268 pp.

Künitzer, A., Basford, D., Craeymeersch, J.A., Dewarumez, J.M., Dörjes, J., Duineveld, G.C.A., Eletheriou, A., Heip, C., Herman, P., Kingston, P., Niermann, U., Rachor, E., Rumohr, H., and de Wilde, P.A.J. 1992. The benthic infauna of the North Sea: species distribution and assemblages. ICES Journal of Marine Science, 49: 127–143.

Künitzer, A., Basford, D., Craeymeersch, J.A., Dewarumez, J.M., Dörjes, J., Duineveld, G.C.A., Eleftheriou, A., Heip, C., Herman, P., Kingston, P., Niermann, U., Rachor, E., Rumohr, H., and de Wilde, P.A.J. 1992. The benthic infauna of the North Sea: species distribution and assemblages. ICES Journal of Marine Science, 49: 127–143.

Lindeboom, H.J. 1995. Protected areas in the North Sea: an absolute need for future marine research. Helgoländer Meeresunters. 49: 591–602.

MacDonald, D.S.,. Pope, J.G., Daan, N., and Reynolds, J.R. 1994. Impact of fishing on non-target species. Report to EC Commission from MAFF, Dir. of Fisheries, University of East Anglia. 85 pp.

Northridge, S.P., Tasker, M.L., Webb, A., and Williams, J.M. 1995. Distribution and relative abundance of harbour porpoises Phocoena phocoena L., white-beaked dolphins Lagenorhynchus albirostris Gray, and minke whales Balaenoptera acutorostrata Lacapède around the British Isles. ICES Journal of Marine Science, 52: 55–66.

Parra, S., Dorrío, A.V., and López-Jamar, E. 1994. The "Aegean Sea" oil-spill: effects on the subtidal benthic communities. ICES CM 1994/L:4. Annex 13.

Pearson, T.H., and Rosenberg, R. 1978. Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. Oceanogr. Mar. Biol. Ann. Rev., 16: 229–311.

Pontier, D., Gaillard, J.M., and Allaine, D. 1993. Maternal investment per offspring and demographic tactics in placental mammals. Oikos, 66: 424–430.

Rees, H.L., and Dare, P.J. 1993. Sources of mortality and associated life-cycle traits of selected benthic species: a review. MAFF Fisheries Research Data Report No. 33. 36 pp.

Rees, H.L., and Dare, P.J. 1993. Sources of mortality and associated life-cycle traits of selected benthic species: a review. Fisheries Research Data Report No. 33. MAFF Directorate of Fisheries Research, Lowestoft.

Rijnsdorp, A.D., Buys, A.M., Storbeck, F., and Visser, E. 1994. De microverspreiding van de Nederlandse boomkorvisserij in 1993–1994. RIVO-DLO Rapport CO 18/94.

Rijnsdorp, A.D., Groot, P., and van Beek, F.A. 1991. The microdistribution of beamtrawl effort in the Southern North Sea. ICES CM 1991/G:49.

Sanders, H.L., Grassle, J.F., Hampson, G.R., Horse, L.S., Garner, P.S., and Jones, C. 1980. Anatomy of an oil spill: long term effects from the grounding of the barge Florida off West Falmouth, Massachusetts. Journal of Marine Research, 38: 265–380.

Schnute, J. 1987. A general fishery model for a size-structured fish population. Canadian Journal of Fisheries and Aquatic Science, 44: 924–940.

Shugart, H.H., and Seagle, S.W. 1985. Modeling forest landscapes and the role of disturbance in ecosystems and communities. In The Ecology of Natural Disturbance and Patch Dynamics, pp. 353–368. Ed. by S.T.A. Pickett and P.S. White. Academic Press, London.

Stearns, S.C. 1992. The evolution of life histories. Oxford University Press, Oxford. 250 pp.

Stokes, T.K.A., Law, R., and McGlade, J. (Eds.) 1993. The exploitation of evolving populations. Springer Verlag, New York. 264 pp.

Stone, C.J., Webb, A., Barton, C., Ratcliffe, N., Camphuysen, C.J., Reed, T.C., Tasker, M.L., and Pienkowski, M.W. 1995. An atlas of seabird distribution in North-west European Waters. Joint Nature Conservation Committee. Peterborough, UK.

Stratoudakis, Y. 1996. Mortality of common dab (Limanda limanda) and long rough dab (Hippoglossoides platessoides) in demersal fisheries of the North Sea and the west coast of Scotland.

Sullivan, P.J., Lai, H.L., and Gallucci, V.F. 1990. A catch-at-length analysis that incorporates a stochastic model of growth. Canadian Journal of Fisheries and Aquatic Science, 47: 184–198.

Thrush, S.F., Hewitt, J.E., Cummings, V.J., and Dayton, P.K. 1995. The impact of habitat disturbance by scallop dredging on marine communities: what can be predicted from the results of experiments. Mar. Ecol. Prog. Ser., 129: 141–150.

van Moorsel, G.W.N.M. 1994. The Klaverbank (North Sea): geomorphology, macrobenthic ecology and the effect of gravel extraction. Bureau Waardenburg, Culemborg, Report 94.24.

Zajac, R.N., and Whitlatch, R.B. 1982. Responses of estuarine infauna to disturbance, I. Spatial and temporal variation of initial recolonisation. Marine Ecology Progress Series, 10:1–14.

6   EFFECTS OF REDUCING FISHING MORTALITY

Analyse available data on system responses in areas where fishing has been reduced.

6.1   Introduction

Finding appropriate tests for the long-term effects of fishing disturbance is fraught with difficulty and perhaps the most fundamental of these difficulties is the limited spatial and temporal scale at which research into ecological processes can be conducted. For the question of fishing disturbance an ideal study requires analysis at the scale of a fishing ground (perhaps 100 km^2) and a time scale greater than the life-span of the longest lived species in the community. Unfortunately, practical and financial limitations are likely to preclude all but the crudest manipulative experiments on these scales. As an alternative, opportunistic analyses of communities subjected to varying levels of fishing disturbance are often attempted. Comparisons are usually between areas which are "fished" and those which are considered nominally "un-fished", but interpretation is difficult if un-fished areas differ from fishing grounds in aspects other than fishing. Such confounding effects may relate to, for example, substrate differences or different histories of use.

One variant on the above approach is to investigate localised areas in the middle of fishing grounds which are protected in some way. The presence of shipwecks afford such opportunities and the idea of using wreck sites as control areas to look for disturbance effects is an old one. Surprisingly, however, there are comparatively few published studies which adopt the approach, although as early as 1955 Graham postulated that circular areas (0.75 mile radii) around light vessels would be rarely trawled and used them as sites for comparison with adjacent fished grounds (Graham, 1955). Although Graham (1955) failed to detect differences between protected "reference" sites and adjacent fishing grounds, the power of this and other studies may well have been limited by the sampling designs that were adopted.

More recently, the methodology for using protected sites has been refined by employing accurate sample positioning capabilities (Hall *et al.*, 1993). It is postulated that the community nearest to a wreck should be close to an undisturbed state and that there is a gradient of increasing disturbance with increasing distance from the wreck. If this postulate is correct and disturbance by fishing does have effects on community structure, this should be reflected in samples collected along transects running outward from a wreck site, or in comparisons of samples taken near the wreck and those taken from closely adjacent fished areas. Although marked spatial patterns were found in this study, no effects that could be reasonably attributed to a gradient of fishing disturbance were detected. It must be recognised, however, that only limited inferences regarding the effects of fishing can be drawn from wreck studies of this kind.

Fishery closures can provide very powerful tests of the ecosystem effects of fishing activities on benthic communities and some parts of demersal fish assemblages (Lindeboom, 1995), provided that they are well designed and that appropriate sampling is conducted. As fishing is very intensive on continental shelves, closures may be the only way to obtain unfished control sites. Replication of the closure is necessary to factor out the effects of disturbance other than fishing. Areas are often closed to conserve fishery resources, and therefore the design of the closure may be suboptimal for assessing fishery impacts. The rationale and general design considerations for closed areas were expounded in ICES (1994a; 1994b) and need not be repeated here.

6.2   Methodological Considerations

The available studies of areas closed to mobile fishing gears have mainly focused on effects on target species and have not addressed effects on non-target species or the more general effects on the ecosystem (ICES, 1994b). Much of the focus of monitoring in most habitats is on detecting changes in the mean size, abundance, diversity of species or whatever variable is considered to be appropriate. Where a change can be identified and correlated with a particular human activity, it is often not clear whether the relationship between human activity and the perceived disruption of the environment is causal (Underwood and Peterson, 1988). ICES (1994a) stated that "It is only in conjunction with a well-designed monitoring programme that [closed] areas could produce insights into the effects of fishing on the marine environment," without providing specific guidance. This report therefore expands on the application of experimental design principles to fishery closures.

6.3 Experimental Design

To be able to connect differences in any variable to any particular human activity, there are strong logical arguments why a manipulative experiment (where the experimenter deliberately alters the level of fishing) is necessary. The fact that populations often vary considerably because of natural processes is something that must be accounted for in any study of putative impact. More precisely, the natural variability of populations must be controlled for, with proper replication and controls, both in time and space. These are basic requirements in experimental design but may nevertheless be a big problem in many studies of environmental impact. The worst feature of natural populations is that the time courses of abundances of a population are rarely the same from one place to another. A particularly powerful analytical tool to identify such interactive processes is the collection of procedures known as analysis of variance (Winer, 1971). A typical problem in studies of environmental impact is that the putative impact cannot be studied in replicate sites. This frequently leads to an asymmetrical design with only one impacted and several control locations. This problem can be avoided in the case of closed areas if research experiments are allowed in the area. If it is sufficiently large, the area can be randomly divided into impacted (trawled) and control locations, thus allowing a symmetrical design. This is preferable to a design with only one impacted site since it enables one to draw more general conclusions.

Recent developments in experimental design have, however, made possible logical and powerful analyses, both for symmetrical and asymmetrical designs, based on the detection of impacts that cause changes in temporal variance of populations, rather than their means (Underwood, 1991; 1992; 1993). These designs could, of course, also be used for other variables of interest (size, reproductive state, number of species, rate of growth etc.).

6.4 Hypotheses

Much of the concern about the effects of fishing activities is motivated by preconceptions that intense fishing degrades the quality of the ecosystem. However, the results of scientific investigations should be as free as possible from subjective value judgements. Fishing activities have multiple effects on the ecosystem, which may be beneficial or detrimental depending on which trophic level and which values are considered. To focus the scientific objectives for any experimental fishery closures, a set of testable hypotheses was listed in the 1994 WGECO report (ICES, 1994a). This list provides a useful framework against which to evaluate the results emerging from closed area studies.

The Working Group hypothesised that the following changes will occur in areas where fisheries are excluded relative to adjacent areas where fishing activities remain unchanged:

Species densities

Increases in population densities will occur for:

a) shallow burrowing and epibenthic species (e.g., *Modiolus modiolus*, Bryozoa, sea pens, anemones);

b) longer-lived benthic species (e.g., *Arctica islandica*);

c) demersal fish.

Deeper burrowing species, e.g., *Callianassa* spp., *Upogebia* spp. and those that have been shown to be less susceptible to mechanical damage by the passage of a trawl will show less marked changes than the species in (a) and (b) above.

Decreases in population densities will occur for:

a) demersal scavenger species which exploit discarded material and/or individuals that are damaged in the path of the trawl, owing to a reduction in food availability;

b) scavenging seabirds, because the closed area will provide less food;

c) seabird breeding success will decrease for birds nesting at colonies which previously exploited discards in the closed area.

Size and age structure

The mean age and size in the populations of long-lived species will show a general increase.

Community structure

The ratio of polychaetes to molluscs will decrease owing to the differential effect of trawling on the mortality rates of these two phyla and differences in their reproductive strategies.

The overall production-to-biomass ratio for the community will decrease owing to a shift in size structure toward large individuals.

The physical complexity of the closed area will increase owing to increases in populations of reef building species, such as *Sabellaria*.

Fish habits

The stomach contents of species which feed on the benthic community will change in comparison with species which feed outside the closed area.

There will be differences in dispersal rates of fish tagged within and outside closed areas due to differences in disturbance.

6.5 Results from Closed Areas

The 1994 WGECO report listed several areas that have been closed to fishing. Of this list, two areas are not discussed further because, at present, there are no data available with which to evaluate the community responses. These are the Western and Emerald Bank Haddock Nursery Area Closure and the Eastern Bering Sea Pot Sanctuary. Results from the other areas are summarised below, along with a number of other closed area studies which are now available for consideration.

6.5.1 Effects of scallop dredging on northern Georges Bank, Northwest Atlantic

This study is not strictly a fishing closure, but a comparison of community structure among areas of high and low dredging effort (Collie et al., 1996). In addition, large areas of Georges Bank have been closed to all bottom fishing since December 1994 to conserve the demersal fish stocks. The closed area on eastern Georges Bank includes one of the most heavily disturbed sites sampled in 1994. Subsequent resampling of the closed area will measure the recovery of a previously disturbed site.

The study site is located on the northern edge of Georges Bank at approximately 42° N, 67° W. This is an area of gravel sediments at depths of 40–100 m. Analyses of the benthos conducted to date are interpreted in the context of the above hypotheses.

Species densities

a) As predicted, the undredged sites had higher densities of shallow burrowing (e.g., *Modiolus modiolus*, *Musculus discors*) and epibenthic species (e.g., *Filograna implexa*, Bryozoa, Hydrozoa)

b) The long-lived species, *Modiolus modiolus*, was more abundant at the undredged sites.

c) The 1-m Naturalist dredge did not sample the entire demersal fish assemblage. However, it was possible to estimate the abundance of small fish (e.g., *Myoxocephalus* spp., *Liparis atlanticus*, *Aspidophoroides monopterygius*) and these were more abundant at the undredged sites.

Hard-shelled molluscs (e.g., *Astarte* spp., *Littorina obtusata*) were equally abundant at dredged and undredged sites; these taxa are apparently less susceptible to mechanical damage by dredging.

As predicted, scavenger species (*Asterias vulgaris*, *Cancer irroratus*, *Pagurus acadianus*) were abundant at the dredged sites, but they were also abundant at the undredged sites. This result suggests that the food available to scavengers is equal at dredged and undredged sites or that scavenger abundance is not food limited.

Size and age structure

There were no consistent differences in mean size and weight between dredged and undredged sites. Some mollusc species had larger mean size at the dredged site, presumably because larger individuals are less susceptible to physical damage than small ones. Many of the species that were abundant only at the undredged site are small species that live within the complex habitat of worm tubes and bushy epilithofauna. The main effect of dredging appears to be destruction of this complex habitat.

Community structure

Counter to predictions, many polychaete species were abundant only at the undredged sites, because of the complex habitat there. The ratio of polychaetes to molluscs was actually lower at the dredged sites.

Production-to-biomass ratios were not calculated but there was no evidence of larger mean size at undredged sites.

As predicted, the physical complexity of the habitat was higher at the undredged sites, owing to the presence of the colonial worm *Filograna implexa*, bushy bryozoans and hydroids.

Fish habits

The northern edge of Georges Bank is known to be an important nursery area of cod and haddock. Many of the megafaunal species that were abundant in the dredge samples have also been identified in fish stomachs collected on Georges Bank. Some of these prey species were more abundant at the undredged sites, while others were ubiquitous. Stomach samples have not yet been analysed on the spatial scale of the dredging disturbance.

6.5.2 Isle of Man, Irish Sea

In 1989 a small area off Port Erin, Isle of Man was closed to fishing by towed gear, although potting for crabs and lobsters was allowed to continue (Brand et al., 1991a). This area is part of the Bradda Inshore scallop *Pecten maximus* fishing ground, which is probably the most heavily dredged scallop ground in the Irish Sea (Brand et al., 1991b) and has been continuously exploited for more than 50 years. The area is adjacent to the rocky coast and is a little over 1 km². The closure was originally introduced in order to assess the potential benefits to the scallop fishery of both rotational closed area management and restocking with cultivated juveniles. Since April 1994 it has also been used in a

study to assess the environmental impact of scallop dredging (MAFF Contract No. CSA 2332).

Unfortunately, since the original purpose of the closure was concerned with the management of the scallop fishery, no detailed benthic monitoring studies were carried out prior to, or in the first four years after, the closure. However, since 1994, the benthic community and physical habitat (depth, particle size structure, and organic content) of the closed area have been compared with adjacent areas of the Bradda Inshore ground that have continued to be subjected to intense commercial dredging during the scallop fishing season (November to May, inclusive). In addition, since January 1995, two plots within the closed area have been experimentally dredged at two-month intervals, using standard commercial gear. In this study, four plots within the closed area (two experimentally dredged and two controls) and three plots in the adjacent fishing ground have been sampled at six-month intervals to determine seasonal patterns in the distribution and abundance of epifauna and infauna. The study will continue for at least one more year, but the results to date have been summarised in a progress report to MAFF (Brand and Hawkins, 1996).

Differences were detected in the epifaunal communities inside and outside the closed area, including a greater species number inside the closed area. There were more infaunal species inside the closed area and, although there was considerable seasonal variation, some species were consistently more abundant in undredged areas. Further analyses of these distributions showed that they were due to the absence of dredging inside the closed area and not to variations in sediment or depth. There is evidence from Multi Dimensional Scaling ordination that the infaunal communities from the dredged areas are more similar to each other than to samples from inside the closed area, which were more variable in composition. Furthermore, by July 1995, samples from the plots inside the closed area that had been experimentally dredged since January 1995 were similar in composition to the dredged sites outside the closed area.

Regarding the above hypotheses:

*Species densities

a) There were higher densities of shallow burrowing and epibenthic species at the undredged sites but particularly of species noted for their vulnerability to dredging (e.g., *Alcyonium digitatum, Anseropoda placenta, Luidia sarsi, Cellaria fistulosa* and *Echinus esculentus*).

b) There was no evidence of longer-lived benthic species at the undredged sites but this is not surprising in view of the relatively short period of time since effective closure of the area.

c) Sampling methods did not permit estimation of the demersal fish assemblage.

d) Scavenger species were common at both dredged and undredged sites, with *Asterias rubens* consistently more abundant on the dredged sites. Several scavenging and predatory species (*Porania pulvillus, Pagurus* spp., *Inachus dorsettensis*, and *Buccinum undatum*) increased in the experimentally dredged plots inside the closed area compared with the controls.

*Size and age structure

Not investigated, but samples are available for future study.

*Community structure

Contrary to the prediction, the ratio of polychaetes to molluscs was lower at the dredged sites on all four sampling dates over two years (April 1994, August 1994, January 1995, July 1995). This may be due to a greater habitat complexity favouring polychaetes in the closed area. However, it is also likely that infaunal bivalves are not adequately sampled by the Day grab on these relatively coarse substrates.

Production-to-biomass ratios were not calculated.

As predicted, there is evidence of an increase in complexity in the unfished areas with an increase in attached epifauna (e.g., *Cellaria fistulosa*, hydroids) and a more diverse infauna.

*Fish habits

No data on fish stomach contents or fish movements are available.

6.5.3   Grand Banks of Newfoundland Trawling Impact Study, Northwest Atlantic

6.5.3.1   Background

In 1990 a collaborative research project between the Scotia-Fundy and Newfoundland Regions of the Canadian Department of Fisheries and Oceans was established to study the potential impacts of mobile fishing gear on benthic marine ecosystems in Atlantic Canada. The long-term objectives are:

1) to develop new instrumentation for viewing and sampling a marine benthic habitat in order to quantify its productive capacity;

2) to obtain quantitative information on the impacts of mobile fishing gear on benthic habitat (both physical structure and biological communities); and

3) to obtain quantitative information on the recovery rate of benthic habitat after disturbance by mobile fishing gear.

Analysis of historical sidescan sonar records collected by the Atlantic Geosciences Centre has provided information on the degree and distribution of mobile gear disturbance off the Atlantic coast of Canada (Jenner et al., 1991; Harrison et al., 1991). In the Scotia-Fundy Region, less than 2% of the seabed surveyed showed any evidence of physical disturbance by mobile gear. Most was due to groundfish trawls and was restricted to areas of low sediment transport. On the Grand Banks, less than 10% of the total records showed evidence of trawling disturbance. The spatial pattern of trawling on the Grand Banks and Labrador Shelf has been estimated from data collected in the fisheries observer programme over the period from 1980 to 1992. The results indicate that trawling intensity varied widely with only relatively small areas being intensively trawled. The most intensive effort has been directed toward the edges of the Banks and especially along the northeast Newfoundland Shelf where the major spawning concentration of northern cod was found.

6.5.3.2 Grand Banks Experiment

A field experiment is currently being conducted at a site on the Grand Banks of Newfoundland, centred at 47° 10' N, 48° 17' W (about 60 km northeast of Hibernia at an average depth of 137 m). The area had not been subjected to heavy trawling in the past decade, could be closed to all mobile gear for an indefinite period, has a medium-fine sand that is easy to process and, most importantly, has an abundant and diverse community of benthic organisms, including a well-developed assemblage of epibenthic organisms. This experiment was conducted with an otter trawl since it is the most widely used mobile gear in Atlantic Canada. Cruises were made in 1991 and 1992 to test sampling equipment and to select a suitable site for the experiment.

The experimental design required the establishment of three 13-km long corridors in close proximity to each other but with different orientations (Rowell et al., 1993; Rowell et al., 1994). These corridors were trawled twelve times by the CSS Templeman with a rockhopper-equipped Engel 145 otter trawl. At the end of each pass, the trawl was retrieved so that the catch could be sorted and the number and biomass of organisms recorded. Sidescan sonar surveys were conducted by the CSS Parizeau before and after trawling along each corridor. Because of time restrictions, it was possible to collect biological samples before and after trawling, with the equipment described above, along just two of the three corridors. In addition, control samples were collected along control corridors parallel to and 300 m away from the centre line of the trawled corridors. Variables being measured include epifauna, macrofauna, meiofauna, bacteria, sediment grain size, organic carbon/nitrogen and acoustic structure of the sediments. Precise sampling

locations are pre-selected on a random basis to ensure that the same bottom is not sampled twice (50 m blocks for grabs and 250 m blocks for sleds).

The experiment was initiated in July 1993. The CSS Parizeau returned in September 1993 to investigate conditions two months after the initial trawling. During a third cruise in July 1994, samples ware collected to investigate conditions one year after initial trawling. Immediately thereafter, the corridors were trawled a second time in identical fashion and then another set of samples was collected to provide a second data set on immediate impacts. A fourth cruise in July 1995 was identical to the third.

6.5.3.3 Project status

It must be emphasised that the Grand Banks experiment is still in progress. As of spring 1995, 140 grab samples and 65 sled samples had been collected. Not all samples have been processed and databases are incomplete. Therefore, only limited data analysis and interpretation have been possible for this report (Don Gordon, Canadian Department of Fisheries and Oceans, Dartmouth, N.S., Canada, pers. comm.). It should be remembered that conclusions drawn from these results could change when the experiment is complete and the data fully analysed.

6.5.3.4 Preliminary results

Physical

Sidescan data indicate that the initial trawl disturbance was still visible on the seabed after one year. Grey-scale analysis of video information collected in July 1993 indicates a more homogeneous distribution of sediment/benthos surface properties in a trawled corridor than in the accompanying control corridor. Fractal analysis of acoustic imaging data indicates that small-scale structural properties of the sediment are altered by trawling to a depth of at least 4.5 cm (Schwinghamer et al., 1996).

Biological

The composition of the biological community at the study site is illustrated by the catches of the three principal sampling gears: trawl, epibenthic sled and 0.5 m^2 grab sampler.

Trawl catches were dominated by epibenthic invertebrate species, mainly snow crab *Chionoectes opilio*, basket star *Gorgonocephalus arcticus*, and sea urchins *Strongylocentrotus* spp. American plaice *Hippoglossoides platessoides* and thorny skate *Raja radiata* dominated the vertebrate catch with some Arctic cod *Boreogadus saida*, capelin *Mallotus villosus*, and a variety of incidental species. The fish catch was extremely small, the maximum being just 80 kg for a 2.5 hour tow over a bottom distance in excess of 13 km.

A total of 74 epibenthic species were encountered in the sled samples collected in July 1993. Eight species were common to all samples. Most occurred in low numbers in just a few samples. The most abundant epibenthic species collected by the sled at the study site are the brittle star *Ophiura sarsi*, sand dollar *Echinarachnius parma*, sea urchin *Strongylocentrotus* spp., and a tube- dwelling polychaete *Nothria conchylega*. The crustaceans such as the snow crab *Chionoectes opilio*, toad crab *Hyas coarctatus*, and hermit crab *Pagurus* spp. were common but much less abundant.

A total of 145 epibenthic and macrobenthic species have been identified in the grab samples collected in July 1993. Extrapolation by means of a species-area function predicts the presence of an additional 36 species. Twenty-seven species were common to all samples while forty-one species were encountered in only one sample. The mean number of individuals was 2476 m^{-2} while the mean biomass was 1171 g m^{-2} (wet weight). The most common taxa were Polychaeta, Crustacea, Mollusca and Echinodermata. Numerically, the Polychaeta dominated while most of the biomass was composed of Mollusca and Echinodermata.

In all three years, the invertebrate catch in the otter trawl decreased significantly in all corridors as the number of passes increased. It was clear from the amount of broken snow crab, basket star and other invertebrate parts on the mesh of the trawl belly and wings ahead of the liner that the organisms retained in the cod-end of the trawl represent only an unknown fraction of those coming into contact with the trawl. Therefore, the direct impacts of the trawl are clearly under-reported.

Invertebrate catch in the trawl generally declined over the three-year sampling period. Sand lance were not present in 1993 but appeared in low numbers in 1994. Capelin abundance increased in 1994. No data have been collected on the potential survivorship of organisms collected in the trawl if dumped overboard but estimates for some species are available in the scientific literature.

Sled samples collected a few days after trawling suggest that 7–12% of sea urchins are damaged and there is a 50% decrease in small snow crabs (large snow crabs are not adequately sampled by the sled). To date, there are no apparent effects on other common invertebrate species, either epifaunal or infaunal. However, there is some suggestion of impacts on smaller, rarer species. The potential for impacts will become clearer as more data become available and a complete statistical analysis is performed.

Trawling does not appear to affect the overall grain size distribution of sediments but there is a suggestion that the very fine (silt/clay) and coarse (gravel) components may be affected. There were no apparent differences in sediment water content attributable to trawling. The 1993 data suggest that the abundance of bacteria may increase immediately after trawling. Preliminary data also suggest

changes in the abundance of some species of nematodes. Sediment organic carbon and nitrogen concentrations appear to be unaffected by trawling based on 1993 Corridor A data.

6.5.3.5   Future plans

For the immediate future, priority is being given to the analysis of samples already collected and the development of databases. Once databases are complete, they will be subjected to various appropriate statistical tests to see if differences due to trawling can be detected above natural variability. Planning will hopefully start on a third field experiment and current thinking is to conduct a similar experiment after trawling on a coarser bottom within an area on the Scotian shelf which has been closed to trawling since 1987.

6.5.4   North Sea plaice box

In 1989 a coastal area in the southeastern North Sea (plaice box) was closed to trawling with vessels larger than 300 hp during the second and third quarter to improve the exploitation pattern of plaice (ICES, 1987; 1994c). In 1994 the box was extended to the fourth quarter and since 1995 the box has been closed for the whole year. The box has primarily affected the distribution of beam trawlers fishing for flatfish. Although beam trawling was still allowed in the plaice box with vessels of 300 hp and data on the effort of this fleet is not comprehensive (ICES, 1994c), the measure has undoubtedly reduced the beam trawl effort in the area.

This raises the question whether the demersal fish assemblage has changed in response to the reduction in beam trawling. RIVO-DLO recently initiated a research programme to analyse the changes in the demersal fish and epibenthos in the southern North Sea using data from the annual beam trawl surveys conducted in August– September in the southeastern North Sea since 1985 (ICES, 1995). The station grid is stratified by ICES rectangles and comprises the southern part of the plaice box as well as adjacent areas to the south and west. In the following sections, the preliminary results are described in relation to two working hypotheses that in the closed area there has been 1) a relative increase in the number of large fish, and 2) an increase in the diversity of the fish assemblage.

6.5.4.1   Size spectrum

To analyse changes in the size spectrum, the survey area was divided into three sub-areas comprising the plaice box and two reference areas (Figure 6.5.4.1). Only stations within the depth band 15–30 m were included in the analyses. Reference area N consisted of the ICES rectangles adjacent to the plaice box. Reference area K consisted of the ICES rectangles in the southern North Sea south of the plaice box. Reference area N consisted of offshore stations only, but reference area K comprised

both inshore and offshore stations. The number of stations varied between 9–20 in the plaice box, 6–13 in area N and 4–12 in area K.

The size spectra were calculated for the period before the plaice box closure (1985–1988) and the period of the plaice box (1989–1994) and indicate that in the plaice box the number of fish above 30 cm has increased (Figure 6.5.4.2). A similar increase was not observed in the rectangles adjacent to the plaice box (reference area N). In reference area K, a slight increase in the number of larger fish was observed. The increase in the abundance of larger fish was observed in both target (plaice and sole) and non-target species.

6.5.4.2   Diversity indices

Community parameters were calculated for the plaice box area and the two reference areas N and K, respectively. Since 1989, the number of species recorded in the plaice box was slightly higher than in the years prior to closure (Figure 6.5.4.3). In the two reference areas no changes were observed. The low number of species recorded in all three areas in 1985 is likely to be related to the smaller number of stations sampled. Species diversity (Shannon-Wiener) and evenness steadily increased in the plaice box after the box was closed in 1989 (Figure 6.5.4.3). A similar increase is observed in reference area K. The trend in reference area N was highly variable without a clear direction.

6.5.4.3   Discussion

The changes in the size spectra in the plaice box support the hypothesis that fishing reduces the relative abundance of the larger fish species. Although an increase in diversity was observed in the plaice box after closure, the interpretation in terms of the hypothesis is not straightforward. Inspection of the relative abundance of individual species revealed that the increase in diversity is related to the relative decrease in the numerical abundance of plaice and dab in the coastal waters of the southeastern North Sea (plaice box and reference area K). The contribution of these species to the total catch decreased from 80–90% in the period 1985–1990 to about 40–60% in the period 1991–1994. A similar decrease occurred in reference area K but not in reference area N. The changes in the numerical abundance of plaice and dab will be mainly determined by the recruitment success which is not directly related to the level of fishing effort in the plaice box. Whether the reduction in beam trawl effort in the plaice box has resulted in a more diverse fish assemblage, as suggested by the increased number of species recorded, needs further detailed investigation.

6.5.5   Loch Gareloch, Inverclyde, Scotland

6.5.5.1   Background

Loch Gareloch, Inverclyde, Scotland, is a sheltered fjordic sealoch in the upper reaches of the Firth of Clyde. Owing to the presence of the R.N. Faslane, Clyde Submarine Base, fishing in the loch is presently prohibited by the Inshore Fishing (Prohibition of Fishing and Fishing Methods) (Scotland) Order 1989. Prior to this Order, fishing was restricted by the Clyde Dockyard Port of Gareloch and Loch Long Order 1967. Anecdotal evidence suggests that although good catches of fish have been taken from the loch, very little trawling took place prior to the ban, and some areas may never have been trawled. The marine fauna supported by the loch has therefore remained undisturbed by fishing for at least 25 years. The continued absence of fishing activity offers a unique opportunity to study fishing effects experimentally in an area which has been closed to fishing for many years and to conduct a manipulative field experiment to examine the impact of fishing disturbance in a controlled manner.

To exploit the opportunity Loch Gareloch offers, a project is being undertaken by SOAEFD Marine Laboratory, Aberdeen, funded through the EC Impact Project. The objective of the project is to follow changes in benthic community structure during a 17-month controlled fishing experiment and for 15 months thereafter.

6.5.5.2   Studies to be undertaken

The work will include:

a) documenting the community that has developed after 25 years' protection from fishing;

b) conducting monthly experimental fishing disturbance with a trawl ground gear at a site on the west side of the inner loch, using an area on the eastern side of the loch as a reference (this experiment will run over 15 months);

c) conducting six-monthly surveys to measure the effects of experimental fishing on the epifaunal community (three surveys during the experiment and three afterwards).

The methodological details of the work programme and Impact II reports providing full details of progress are available (contact SOAEFD Marine Laboratory), but their contents are summarised below. It is too early, however, to report results within the framework of hypotheses discussed above.

6.5.5.3 Summary of progress to date

In November 1993, prior to the start of the experiment, a baseline survey of Loch Gareloch was conducted to map the topography, substrata, and benthic fauna. The objective of this survey was to broadly characterise the area with acoustic, underwater television, and benthic sampling techniques, and to obtain sufficient data for selecting an experimental and reference site within the loch.

Trawling disturbance at the experimental area on the western shore of Loch Gareloch commenced in January 1994 and continued one day per month until April 1995. On each visit ten tows (approximately 45 minutes in duration at a speed of 2 knots) over the treatment area were made using a modified rockhopper groundgear. Scanmar units have been deployed on the gear during some treatments to accurately measure the width of the disturbed track produced by each tow.

Further surveys to follow the effects of the disturbance were carried out in May 1994, October 1994, and May 1995. A comparative survey between the reference and experimental areas was carried out in October 1995, and the next survey will take place in May 1996. All surveys use the same techniques, examining physical effects on the seabed with acoustic seabed discrimination techniques and sidescan sonar, the epifaunal community with towed underwater TV, and the infaunal community with Day grab samples. Data from the preliminary and October 1994 surveys are now available and have been subjected to preliminary analysis. Acoustic and epifaunal data collected during May 1995 and October 1995 have also been analysed.

6.5.5.4 Results to date

The preliminary survey results from the acoustic seabed discrimination and Agassiz trawl studies indicate that the sediment and epifauna were relatively uniform throughout the surveyed areas, with the exception of the shallower region in the northeastern part of the loch, which appeared to have a rougher seabed. Owing to the marked differences that are apparent in this latter area it was excluded from consideration. Analysis of the TV survey failed to identify any distinct locations within the survey area. Epifaunal species densities were generally low and relatively uniform.

Physical effects

The sidescan surveys carried out during May and October 1994 showed evidence of considerable disturbance to the seabed in the experimental area, while the seabed in the reference area remained devoid of distinct topographic features. The acoustic discrimination surveys also provided evidence that trawling has had an effect on the seabed. The sidescan survey carried out during May 1995 continued to show evidence of considerable disturbance to the seabed in the experimental

area, while the seabed in the reference area remained devoid of distinct topographic features.

By October 1995, the treatment site had been undisturbed for 6 months. Very few trawl tracks were identified on the sidescan track, and those that were seen were very faint. Data from the acoustic discrimination survey would appear to indicate that the seabed in the treatment site is returning to its pre-disturbance condition.

Ecological effects

Analysis of the infaunal samples from the preliminary survey suggests that the communities from the two sides of the loch were different before the experiment started. A dendrogram of Bray-Curtis Similarity indicated that none of the samples were less that 70% similar, but despite this similarity between stations, statistical tests reveal that the treatment and control areas were significantly different.

Infaunal analysis shows that the communities differ both between treatment and reference sites and within each site between surveys. The mean densities of the 12 commonest infaunal species indicate that a number of species densities have changed at each site, but it is difficult to identify any clear pattern for individual species.

Analyses have also been undertaken for two abundant epifaunal taxa, the starfish *Asterias rubens* and the sea pen *Virgularia* sp. Differences between the disturbed and reference areas appear to be rather small on any given date, particularly in contrast to the seasonal differences that are apparent.

On the basis of the analyses conducted to date, no firm statements about the effects of the experimental disturbance can be made, except that they do not appear to be particularly marked for these two species. It should be stressed, however, that a much more complete analysis will be necessary before any firm conclusions can be drawn.

6.5.5.5 Discussion

It is important to stress that the results described for this project are very preliminary in nature. The analysis has been included to illustrate the progress that has been made with the project more than to draw firm conclusions. The problems of lack of replication that are inherent in the experimental design are recognised (see Sections 6.5.6.2 and 6.5.6.3 for a more complete discussion of these issues), but the scale of the experiment and the uniqueness of the study site preclude the possibility of multiple experimental areas. Importantly, however, the study shows that a manipulative field experiment to examine fishing effects is feasible and, despite the absence of spatial replication, valuable inferences can probably be drawn.

6.5.6 Trawl closure in the Gullmar fjord, Sweden

6.5.6.1 Introduction

Since 1 January 1990, trawling for shrimps *Pandalus borealis* has not been permitted in the Gullmar fjord on the Swedish Skagerrak coast. The closure was meant to protect Gullmar fjord as an important reference area for scientific purposes and to protect valuable reproduction areas for important fish stocks. The regulations against trawling in these areas are now about to be reconsidered. Standardized benthic sampling has been conducted since the early 1980s (Tunberg, 1994) but without a spatial control trends in the abundance of benthic fauna cannot necessarily be attributed to the trawl closure.

As background for further management decisions, it has been proposed that a study of trawling effects should be carried out in the area. The following design has been proposed for evaluating potential effects of trawling on benthic macrofauna, diversity, and sediment-related variables in the Gullmar fjord. These investigations will be performed mainly with sediment corers and video recordings with a ROV. However, the project is also intended to investigate potential effects on shrimp *P. borealis* populations and fish stocks. The fact that there has been prohibition of trawling for six years provides good opportunities to perform a manipulative experiment which is a prerequisite for establishing facts about cause and effect. Data from previous monitoring studies (e.g., Tunberg, 1994) may be used as valuable background information to design experiments which will allow powerful tests of the effects of trawling.

One basic approach is to expose parts of the closed area to trawling and to have some areas which are not exposed to trawling as controls. For practical reasons (connected to the trawling) it is only possible to have two exposed sites and two control sites in Gullmar fjord. Basically, this is a straightforward analysis of variance (ANOVA) design. The problem is that, for this kind of application, this design will often have low statistical power. The reason for this low power is the high 'natural' variability that we can expect, coupled to the low number of independent replicates ($n=2$) that are possible because of the above-mentioned constraints on trawling. Note that the number of cores taken in each impacted or control site does not affect the number of independent samples.

The way the Swedish group proposes to cope with the problem of low power is to design a study which detects the effects of trawling as an interaction between the treatment (trawling) and time (e.g., Underwood, 1991; 1992). In short, this means that if trawling has a significant and spatially consistent effect on benthic fauna, the sites which are exposed to trawling will behave differently in time from those sites that are not exposed to trawling. With this approach, it will be possible to

distinguish the 'natural' variability and to detect putative impact in several ways, as described below. Although the logic behind these tests for detecting such effects is simple, the technical aspects are not straightforward. A brief overview of the design is necessary to illustrate what is required for such a study.

6.5.6.2 Proposed design

With the exception of the proposed replication of trawled sites, the following design is similar to the ones suggested in Underwood (1991; 1992; 1993). These are multifactorial ANOVAs and the proposed design contains four main factors:

- Before/After impact,
- Treatment (trawling),
- Sites within Treatment, and
- Time within Before/After.

As described above, the study is constrained to include two impacted sites and two control sites. Thus, the possibilities to optimize sampling effort are by varying the number of times sampled (t) and the number of samples (n) taken each time. The spatial location of the experimental units should be randomly chosen. The sites do not need to be identical; they simply have to follow the normal requirements that they come from a population of apparently similar sites. With this design, an impact must appear as an interaction between the differences among locations before the impact starts and those differences prevailing after it begins. The details of how this interaction should be estimated for situations where there are trends over time in the populations (caused by factors other than the treatment) and when the variance of the populations is large relative to the treatment effect are complex. These issues are addressed in the literature cited above, but in general collaboration with biometricians is to be encouraged in the design phase of the experiments.

The statistical power of these tests depends on the variability of undisturbed populations. Peterman (1990) argued that the magnitude of change in abundance of a species that should be considered to be biologically important in some habitat should be defined in advance. This would then set the minimal change that should be detected and, thus, enable sampling to be designed with adequate power. Calculations of power also depend on knowledge of the natural rates of variation in abundance of a population and the dynamics of recovery. This knowledge is clearly necessary because it will influence the size of replication in the sampling programme and should be crucial in influencing the choice of species (variables) to monitor. Historical data could be used for such an *a priori* power analysis. Otherwise, a pilot study is needed. Using *a priori* information on variability makes it possible to efficiently allocate sampling effort by varying t and n. Note that this is not possible in the simple, one-way design.

6.5.6.3 Choice of species to monitor

Due to the different ecologial characteristics of species, the probability of detection of putative effects of trawling will depend on which organisms (variables) we choose to measure. Although this study also aims to investigate overall effects at the community level (Warwick, 1993), the previously sampled data may be used to select taxa which will give the most powerful tests. Analysis of data on 11 taxa from four locations in three years (Tunberg, 1994) showed that 7 of 11 investigated taxa varied in time or space or as a result of an interaction between the two. As an example of the importance of the choice of organism, two different taxa were selected for a priori power analysis (Peterman, 1990; Fairweather, 1991). Heteromastus filiformis showed a significant interaction in time and space. Thus, this organism varied significantly even in the absence of trawling and to detect significant trawling effects on the abundance of this species requires a stronger effect as compared to less variable species (e.g., Melinna cristata), given identical sampling design. Power calculations were made, both for the simple nested design and for the more extensive proposed design. In summary, these analyses indicate that less sampling- intensive designs will not have sufficient power (i.e., power will be below 0.8) unless the treatment effect is of the same order as the error variation in the particular test. On the other hand, a more sampling-intensive design ($n=10$; $t=5$) will be considerably more powerful (Power ≈ 0.8) even at effect sizes half as large as the natural variability. Compared with the power of tests in the simpler nested design, the tests in the more extensive model are considerably more powerful. For example, even if 24 samples were to be taken at each location with the simpler design, the power would be less than 0.2 for all species! The flexibility of this design is also more limited as the only thing that may be decided by the experimenter is the number of cores taken at each location.

After a study, if the analyses were not significant, it would not necessarily mean that the disturbance had had no effect (Toft and Shea, 1983; Peterman, 1990). Rather, it may indicate that any effects that have occurred were unlikely to be detected. It is very important not to draw conclusions as if the null hypothesis (H_0) were true. A false H_0 could have been missed because of a low-power experiment caused by small sample sizes or large sampling variability. Therefore, β or statistical power $(1 - \beta)$ has to be reported. β is the probability of making a type II error (not rejecting H_0 when it should have been). Quantification of the probability of making a type II error and reporting detectable effect sizes can clearly improve the inferences and recommendations that result from them.

Finally, these preliminary calculations emphasize the need for careful planning of studies of putative trawling effects. Power calculations can only be viewed as rough estimates indicating the approximate sampling design necessary to detect impacts of a certain magnitude, but when data are available as in this case, these calculations may successfully be used to optimize sampling design. It is hoped that final results from the study of trawling effects in the Gullmar fjord will be presented at the next meeting of WGECO.

6.6 Conclusions

Data on ecosystem responses where fishing mortality has been reduced are now becoming available from a number of studies within the ICES area. These studies have employed a variety of approaches including spatial comparisons between fished and unfished areas, closed areas, and experimental fishing disturbances in closed areas. WGECO recognises inherent spatial and temporal constraints in designing appropriate tests of the long-term effects of fishing disturbance. Nevertheless, a planned experimental design can achieve a substantial increase in statistical power compared with unplanned experiments and other spatial comparisons. A prerequisite for establishing closed areas is intensive sampling carried out well in advance of the closure. For example, four well-studied 'boxes' (black squares B, C, D, E in Figure 6.6.1) are situated in areas of the North Sea determined to be suitable as closed areas (white areas) (ICES, 1994a). For these boxes 10-year series of random fish samples, hydrography, and nutrient measurements are available.

The results reported here are consistent with the hypothesis that shallow-burrowing and epibenthic species are more abundant in unfished areas. There is, as yet, little evidence that longer-lived benthic species are more abundant at unfished sites. In the North Sea plaice box, the number of fish larger than 30 cm has increased. This shift in the size distribution is most probably a direct response to the decrease in exploitation rather than changes in the habitat. Demersal scavenger species were ubiquitous in all studies reported; some taxa occur at greater densities in fished areas. No data were presented on the effects of closed areas on scavenging seabirds.

No consistent differences in the size structures of benthic communities were found for one study in which organisms were measured. The hypothesis that the ratio of polychaetes to molluscs will be lower in unfished areas was rejected in the two studies on gravel habitats. There is evidence of increased physical complexity in unfished areas. The increase in habitat complexity is especially marked on gravel sediments where epifaunal taxa (e.g., Cellaria fistulosa, Fillograna implexa, hydroids, bryozoans) provide habitat for smaller and more fragile taxa (e.g., polychaetes, shrimps, and brittle stars). Data on fishing habits and movements were not presented. However, it should be noted that some of the benthic species enumerated in these studies are important prey of demersal fish. Therefore, there is a potential link between the disturbance of benthic communities and fish feeding.

The impacts of mobile gear depend on the kind of gear used, the type of bottom, and the composition of benthic communities. Therefore, it will be impossible to

generalise the results of a few experiments to the wide variety of habitats found in the ICES statistical areas. The severity of impacts for different scenarios is likely to vary over a considerable range from minor to extreme. Each habitat must be considered on the basis of its inherent natural features and the management objectives for that area.

The preliminary results of the Grand Banks experiment indicate impacts of otter trawling on epibenthic invertebrate communities on flat, uniform, well-sorted sandy bottoms. The abundance of large, attached epibenthic species easily susceptible to damage is limited but mobile epifauna are reduced by repeated trawling. Damage to surface-dwelling, tube-dwelling, and burrowing species has not yet been demonstrated, but damage to the infaunal habitat by homogenisation of surficial sediments occurs.

There is evidence from other research to suggest that biological damage could be greater on finer, more cohesive sediments (e.g., silt and clay) as well as on coarser sediments (e.g., gravel, cobble, and boulder). Such bottom types are common in the ICES area. Habitats with gravel sediments appear to be particularly sensitive to scallop dredging because of their well-developed epifaunal communities encrusting the pebbles and cobbles. These taxa are abraded by the dredge and by the gravel particles as they pass through the dredge rings. In heavily dredged areas the gravel is more homogeneous and free of encrusting organisms.

The management of any mobile gear demersal fishery should include habitat conservation as one of its objectives. A logical extension of the benthic habitat conservation perspective is to make better use of protected areas. Areas of the seafloor which are important spawning, egg deposition, nursery, and feeding areas for valued species should be protected from damage by mobile gear. In some cases, closure for just a few months a year may provide sufficient protection while in others year-round closure over a large area may be necessary.

6.7 References

Brand, A.R., and Hawkins, S.J. 1996. Assessment of the effects of scallop dredging on benthic communities. Interim report to MAFF. February, 1996.

Brand, A.R., Allison, E.H., and Murphy, E.J. 1991a. North Irish Sea scallop fisheries: a review of changes. *In* An International Compendium of Scallop Biology and Culture, pp. 204–218. Ed. by S.E. Shumway and P.A. Sandifer. World Aquaculture Society, Baton Rouge.

Brand, A.R., Wilson, U.A.W., Hawkins, S.J. Allison, E., and Duggan, D.A. 1991b. Pectinid fisheries, spat collection, and the potential for stock enhancement in

the Isle of Man. ICES Marine Science Symposium, 192: 79–86.

Collie, J.S., Escanero, G.A., and Valentine, P.V. 1996. Effects of bottom fishing on the benthic megafauna of Georges Bank. 1996 WGECO Meeting, Working Paper.

Fairweather, P.G. 1991. Statistical power and design requirements for environmental monitoring. Australian Journal of Marine and Freshwater Research, 42: 555–567.

Graham, M. 1955. Effect of trawling on animals of the sea bed. Deep Sea Research 3 (Suppl.): 1–6.

Hall, S.J., Robertson, M.R., Basford, D.J., and Heaney, S.J. 1993. The possible effects of fishing disturbance in the northern North Sea: an analysis of spatial patterns in community structure around a wreck. Netherlands Journal of Sea Research, 31: 201–208.

Harrison, P.H., Strong, K.W., and Jenner, K.A. 1991. A review of fishery related seabed disturbance on the Grand Banks of Newfoundland. Final contractor's report to DFO from Maritime Testing (1985) Ltd. 32 pp. plus figures and appendices.

ICES. 1987. Report of the *ad hoc* meeting of the North Sea Flatfish Working Group. ICES CM 1987/Assess:14.

ICES. 1994a. Report of the Working Group on Ecosystem Effects of Fishing Activities. ICES CM 1994/Assess/Env:1.

ICES. 1994b. Joint report of the ICES Advisory Committee on Fishery Management and the Advisory Committee on the Marine Environment. ICES Cooperative Research Report No. 203. 21 pp.

ICES. 1994c. Report of the Study Group on the North Sea 'Plaice Box'. ICES CM 1994/Assess:14.

ICES. 1995. Report of the Study Group on Beam Trawl Surveys. ICES CM 1995/G:5.

Jenner, V., Strong, K.W., and Pocklington, P. 1991. A review of fishery related seabed disturbance in the Scotia-Fundy Region. Industry Services and Native Fisheries Branch, Project Report No. 166.

Lindeboom, H.L. 1995. Protected areas in the North Sea: an absolute need for future marine research. Helgolander Meeresuntersuchungen, 49: 591–602.

Peterman, R.M. 1990. Statistical power analysis can improve fisheries research and management. Canadian Journal of Fisheries and Aquatic Sciences, 47: 2–15.

Rowell, T.W., Schwinghamer, P., and Gordon Jr., D.C. 1993. Canadian project to evaluate the impacts of mobile fishing gear on benthic habitat. Report to ICES.

Rowell, T.W., Schwinghamer, P., Gilkinson, K., Gordon Jr., D.C., Hartgers, E., Hawryluk, M., McKeown, D.L., Prena, J., Vass, W.P., and Woo, P. 1994. Investigating the impact of otter trawling of benthic communities of the Grand Bank. Report to ICES.

Schwinghamer, P., Guigné, J.-Y., and Siu, W.C. 1996. Quantifying the impact of trawling on benthic habitat using high resolution acoustics and chaos theory. Canadian Journal of Fisheries and Aquatic Science, 53: 288–296.

Toft, C.A., and Shea, P.J. 1983. Detecting community-wide patterns: estimating power strengthens statistical inference. American Naturalist, 122: 618–625.

Tunberg, B. 1994. Traleffekter i Gullmarsfjorden 1990–1993. Länsstyrelsen i Göteborgs och Bohus län. Report in Swedish.

Underwood, A.J. 1991. Beyond BACI: experimental designs for detecting human environmental impacts on temporal variations in natural populations. Australian Journal of Marine and Freshwater Research, 42: 569–587.

Underwood, A.J. 1992. Beyond BACI: the detection of environmental impacts on populations in the real, but variable, world. Journal of Experimental Marine Biology and Ecology, 161: 145–178.

Underwood, A.J. 1993. The mechanics of spatially replicated sampling programmes to detect environmental impacts in a variable world. Australian Journal of Ecology, 18: 99–116.

Underwood, A.J., and Peterson, C.H. 1988. Towards an ecological framework for investigating pollution. Marine Ecology Progress Series 46: 227–234.

Warwick, R.M. 1993. Environmental impact studies on marine communities: pragmatical considerations. Australian Journal of Ecology, 18: 63–80.

Winer, B.J. 1971. Statistical principles in experimental design. McGraw-Hill, New York. 907 pp.

7 ESTIMATING DISCARDS AND THEIR UTILIZATION

7.1 Summarise any new work undertaken on the estimation of by-catches and discards and their exploitation by scavengers.

'Again, the Kingdom of Heaven can be illustrated by a fisherman, he casts the net into the water and gathers in fish of every kind valuable and worthless. When the net is full he drags it up the beach and sits down and sorts the eatable ones into crates and throws the others away.'

Matthew 13: 47–48

7.2 Definitions

In order to clarify the use of terminology, the following definitions have been adopted by the Working Group. These follow McCaughran (1992) and the Fifth North Sea Conference.

Target catch = The catch of a species or species assemblage which is primarily sought in a fishery.

Incidental catch = Retained catch of non-targeted species.

Discards = The portion of the catch returned to the sea as a result of economic, legal or personal considerations.

By-catch = Discards plus incidental catch.

A global assessment of by-catch and discards reviews more than 800 articles published prior to 1994 (Alverson *et al.*, 1994).

7.3 Estimation of By-catches

7.3.1 Incidental catch of fish: demersal fisheries

By-catch may be divided into commercial species (incidental catch), non-commercial fish, shellfish and crustaceans, mammals, birds, and benthos. Components of the by-catch have been assessed in several studies. The species that comprise the incidental catch vary among fisheries (a species that is a target in one fishery may be described as incidentally caught in another). Table 7.3.1.1 categorises fish caught in Scottish groundfish surveys as to whether or not the species is targeted by Scottish fishing vessels and quantifies the landings in 1994 by UK vessels into Scotland. A total of 19,400 tonnes of non-target fish were landed; this compares with 394,000 tonnes landed from targeted fish stocks. The incidental catch thus comprises 5% of total landings into Scotland by UK vessels. Note that the industrially fished species (e.g., sandeels, Norway pout) were fished for in the area but not landed in Scotland.

7.3.2 Incidental catch of fish: industrial fisheries

The Working Group on the Assessment of Demersal Stocks in the North Sea and Skagerrak provide information on the by-catch of species in the industrial fisheries of the North Sea for the years 1974 to 1994 (ICES, 1996a). Information for 1994 is summarised in Tables 7.3.2.1 and 7.3.2.2. The target species in these

fisheries are Norway pout, sandeel, and sprat. The catch in 1994 of these three species was 1,222,000 tonnes, with an incidental catch totalling 85,000 tonnes (6.5%). The incidental catch in 1994 was lower than the average over the years 1974 to 1993, however. Over these years the mean target catch size was 1,207,000 tonnes, with an incidental catch of 249,000 tonnes (17%).

7.3.3 Marine mammals

There has been considerable interest in the scale of cetacean by-catch in fisheries. The Study Group on Seals and Small Cetaceans in European Seas recently reviewed (ICES, 1996b) knowledge on the by-catch of harbour porpoises *Phocoena phocoena* in bottom-set gill nets in the North and Celtic Seas, and the relationship of the by-catch to estimates of stock size (Table 7.3.3.1). For a variety of reasons, these by-catches were almost certainly an underestimate of the total number drowned in these fisheries. Further studies of harbour porpoise by-catch are being undertaken in the North, Baltic, and Celtic Seas.

Northridge (1996) has reviewed methods of monitoring cetacean by-catch and concluded that independent on-board observer schemes provide the most accurate and reliable statistics. Mechanisms such as port interviews, collection or counting of marine mammal carcasses when they are brought to port, or logbook schemes which rely on fishermen to report by-catches were found to be unreliable for estimating total catch, but may be useful for more limited purposes.

7.4 Estimation of Discards

7.4.1 Methodological studies

There have been a number of recent studies attempting to improve methods for quantifying discards. Most of these studies have been designed to improve fish stock assessment and fisheries management, rather than to provide full quantification of amounts of discards returned to the sea. Consequently, these studies tend to focus on commercial species of fish.

Cotter (1995) discusses practical aspects of sampling discards and attempts to estimate components of variance in available discarding data in order to optimise survey design. This report proposes a general, yet practical and efficient, strategy for estimating discards (of target species) in a fishery that includes sampling by fishermen as well as by trained technicians, minimising the number of sampling strata, sampling vessels with probability proportional to size but sampling trips as opportunities permit and the continuing development of models to estimate missing values and otherwise improve precision.

New discard sampling schemes have been carried out in the northern Irish Sea from August 1993 to September 1994 (Tamsett, 1995) and started recently for the main Irish fleets operating in the waters around the Irish coast

(Connolly *et al.*, 1995). These schemes complement continuing studies off Scotland (Reeves, 1995), Northern Ireland (Briggs, 1995), France (Celtic Sea and Bay of Biscay) (Peronnet, 1995), and Ireland (Connolly *et al.*, 1995).

7.4.2 New estimates of discard amounts

Peréz *et al.* (in press) have assessed quantities of species discarded by the Spanish fleets in 1994 in ICES Divisions VIa, VIb, VIIb, VIIc, VIIj, VIIk, VIIIa, VIIb, VIIIc, and IXa (Table 7.4.2.1). No marine mammals and very few birds were recorded in the samples. Trawlers generated a very high proportion of the amounts discarded and caught many more species than other fishing methods (Table 7.4.2.2).

Peronnet (1995) reports some results of studies of discarding from the French *Nephrops* and demersal fisheries in the Celtic Sea. Connolly *et al.* (1995) give some information on discard rates of whiting from the Irish Sea *Nephrops* fishery. A key point which emerges from these studies is that much of the variation in discarding rates from both fisheries can be attributed to variations in year class size of the species most discarded.

Garthe *et al.* (in press) bring together estimates of discard totals, including those reviewed in the last report of the Working Group on Ecosystem Effects of Fishing Activities (ICES, 1994a). As noted in ICES (1994a), there is insufficient information to fully quantify discard amounts from all North Sea fisheries, so Garthe *et al.* (in press) made estimates. They excluded the coastal zone fisheries, such as those for brown shrimp *Crangon crangon* in the Wadden Sea. Three main sources of discards and offal in the North Sea were a) demersal trawlers and seiners catching gadids, b) pelagic trawlers and seiners, and c) beam trawlers. Totals of 56,000 tonnes of haddock, 36,000 tonnes of whiting, 15,000 tonnes of cod, and 10,000 tonnes of plaice were estimated to be discarded annually by the demersal fisheries for gadids in the North Sea. In addition, the work of Jensen *et al.* (1994) indicates that a further 15,000 tonnes of roundfish, 23,000 tonnes of flatfish, and 10,000 tonnes of elasmobranchs of little economic interest were discarded by these fisheries. Totals of 95,000 tonnes of herring and 68,000 tonnes of mackerel discards were estimated on the basis of catch data from the pelagic fisheries. The beam trawl fishery was estimated to discard 120,000 tonnes of dead invertebrates, 251,000 tonnes of flatfish, and 19,000 tonnes of roundfish. Amounts of offal discharged are estimated as in ICES (1994a). These figures are summarised and divided between areas used to assess seabird consumption (see Section 7.5.2) in Table 7.4.2.3.

Weber (1995) estimated the amount of discarded cod in a directed trawl fishery in the German Bight (ICES Division IVb) over a period of 12 years. Depending on year class strengths, this fishery may discard up to 750

tonnes of cod (two million individuals). Studies continue on different sectors of the German fishery, including beam trawls. First results show that about 60% of the total catch consists of benthic organisms (mainly *Asterias rubens*, *Ophiura* sp., and *Echinocardium*) (Table 7.4.2.4). Between 45% and 70% of the remaining fish portion of the catch is discarded, especially plaice *Pleuronectes platessa* and common dab *Limanda limanda*.

Norway and the EU have established three fish stock Working Groups that will, amongst other tasks in the next year, examine the discard issues in Norwegian fisheries. The quantity of discards from Norwegian demersal fisheries in the North Sea is limited owing to the relatively small scale of this fishery. The Norwegian fishery for mackerel may generate considerable quantities of discards due to the price difference between small and large mackerel. A preliminary estimate of the discards of mackerel that would occur if discarding were allowed in this Norwegian fishery is in the order of 50% of landed catch, given the present market, availability and effort.

7.5 Exploitation by Scavengers

7.5.1 Survival of discarded species

A proportion of the organisms discarded from fishing vessels will survive and will not be available to scavengers (although some injured organisms may be eaten or may be more easily consumed due to displacement from their usual habitat). It is necessary to know the proportion that survives in order to calculate the availability of discards to scavengers. De Groot and Lindeboom (1994) and Kaiser and Spencer (1995) investigated the subsequent survival of animals discarded after being caught in a 4 metre beam trawl. Relatively high proportions of starfish, hermit crabs, and molluscs survived capture, while fish (except dogfish), sea urchins, and swimming crabs suffered higher mortality after capture. Gear type is likely to be important in determining survival rates.

7.5.2 Seabirds

A comprehensive study of the consumption of discards and offal by seabirds in the North Sea (not including the Channel) has been undertaken by Camphuysen *et al.* (1995). Their results are updated and summarised by Garthe *et al.* (in press). These studies included experiments to determine the fate of fish individually discarded during daylight from research vessels.

Based on the calculations of amounts of offal and discarded fish and benthic organisms (presented in Section 7.4.2), Garthe *et al.* (in press) estimate that a maximum of 3.4 x 10^{12} kJ (800,000 t) of discarded items would be available to seabirds. This would potentially be enough to satisfy the energy demands of the entire scavenging seabird community in the North Sea. However, the distributions of discards and of seabirds are

different; maximum availability of discards is in the southern North Sea, while maximum numbers of seabirds are present in the northwestern North Sea (ICES, 1994b). Several other assumptions would also need to be met if fishery waste were to support all scavenging seabirds; these include the need for waste to be relatively evenly available through time, which is unlikely, given the seasonal nature of landings. Actual conditions that would enable seabirds to consume all wastes discarded are also rarely met; waste is often discharged in bulk (swamping the ability of seabirds to swallow them before sinking), ship manoeuvring and discharging by night (when some birds will not feed). Experimental discarding of individual items revealed that the proportion of fishery waste consumed by seabirds varied between areas, and by season, and type of waste (Table 7.5.2.1). Overall, offal and roundfish were consumed in the highest percentages. If the percentages of each type of waste consumed in the different sub-regions shown in Figure 7.5.2.1 are applied to the amounts available (as summarised in Table 7.4.2.3), then around 55,000 tonnes of offal, 206,000 tonnes of roundfish, 38,000 tonnes of flatfish, 2,000 tonnes of elasmobranchs, and 9,000 tonnes of benthic invertebrates would be consumed by seabirds in the North Sea. However, as indicated above, this is likely to be an overestimate since waste discharge practices differ between the study of Camphuysen *et al.* (1995) and on fishing vessels.

Spaans *et al.* (1994) examined the diets of lesser black-backed gulls *Larus fuscus* breeding on the Dutch coasts. Clupeids, gadids, and flatfish of size ranges not naturally available to plunge-diving gulls formed a considerable portion of the diet. These fish probably came from the Dutch beam trawler fleet working off these coasts, corroborating the findings of Camphuysen *et al.* (1995). Lesser black-backed gulls nesting on the Dutch coast rely to a great extent on discarded fish.

7.5.3 Other scavengers

The studies on seabird scavengers in the North Sea (Camphuysen *et al.*, 1995; Garthe *et al.*, in press) indicate that more than 7,800 tonnes of offal, 56,200 tonnes of roundfish, 261,200 tonnes of flatfish, 13,000 tonnes of elasmo-branchs, and 140,800 tonnes of benthic invertebrates sink beyond the reach of seabirds and are potentially available to other scavenging organisms. Some of these discarded organisms will survive the process of being caught and discarded, while the mortality rate of some species is very high once caught. There have been no recent direct studies of this part of the North Sea scavenger community.

In a study of the diet of fish in ICES Division VIIIc, Olaso (in prep.) examined more than 36,000 stomach contents of 28 species of demersal fish between 1988 and 1995. Much of the diet of the fish studied comprises silvery pout *Gadiculus argenteus* and blue whiting *Micromesistius poutassou*, the most abundant fish species in the area studied. In most years, 19 of the 28

species preyed on blue whiting. Eight of these species are active hunters, but the other eleven are opportunists, preying on dead or moribund fish. Blue whiting comprises the highest percentage (24% by weight) of any species discarded from fisheries in the area. It therefore seems likely that the main scavenger fish species in the area are supported to some extent by discards from the fishery.

In ICES Division VIIIc the proportion of the dogfish *Scyliorhinus canicula* in catches has increased from 3% in 1983 to 7% in 1994. A study of the survival of dogfish after being discarded in this area began in 1994. A new study will start soon examining the importance of discards in the diet of this species. Present information indicates that blue whiting represents 30% of total food. Much of this is likely to come from discards since blue whiting (of the sizes consumed) usually occurs only below 90 m. Studies of changes in community structure (and size distribution) between 1983 and 1995 are also under way.

Pedersen (1995) describes the diet of starry rays *Raja radiata* in west Greenland waters. Considerably more redfish *Sebastes* spp. were found in ray diets than would be expected from estimates of local abundance. Large quantities of redfish are discarded from, or damaged by, the local shrimp fishery. Pedersen (1995) concludes that fishing activities make redfish considerably more available to rays than would otherwise be the case.

Beam trawling generates two food sources for scavenging species: exposed and damaged infauna, and dead discarded material. It is often difficult to separate the attraction of these two foods, and the same species may often be involved. Kaiser and Spencer (1994; 1996) and de Groot and Lindeboom (1994) describe the behaviour of scavenging species that move into an area following beam trawling in the Irish and North Seas. Species that move into the area include whiting *Merlangius merlangus*, bib *Trisopterus luscus*, plaice *Pleuronectes platessa*, dabs, gurnards *Aspitrigla cuculus* and *Eutrigla gurnardus*, hermit crabs, common starfish, and whelks.

In order to find out which species of scavenger were feeding on what kind of discards, recent studies in the southern North Sea, under the EC project "Impact II", baited traps with dead fish, mollusc meat, crushed crustaceans, or echinoderms. The traps were set on the seabed for two days in five different locations in the southern North Sea. Traps baited with fish primarily caught swimming crabs, hermit crabs, and starfish. Shrimp *Crangon crangon* and *Cirolana* arrived in smaller numbers with occasional *Cancer* and nemertines. Traps baited with mollusc meat mainly caught swimming crabs, hermit crabs, whelks *Buccinum*, and gadoid fish (e.g., *Trisopterus, Gadus*). Fish were particularly attracted when sea-mouse *Aphrodite* was used as bait. Traps baited with dead crustaceans attracted thousands of amphipods *Orchomene nana* and *Scopelacheirus hopei*. Laboratory investigations of the daily food consumption

of some benthic scavengers are in progress. Dead fish that were not directly consumed by scavengers were found to decay in about 8 days at 15 °C, 11 days at 10 °C, and 16 days at 5 °C.

7.6   References

Alverson, D.L., Freeberg, M.H., Murawski, S.A., and Pope, J.G. 1994. A global assessment of fisheries bycatch and discards. FAO Fisheries Technical Paper No. 339, Rome, FAO. 233 pp.

Anon. 1995. Scottish sea fisheries statistical tables. Scottish Office Agriculture and Fisheries Department, Edinburgh.

Berrow, S., Tregenza, N., and Hammond, P.S. 1994. Marine mammal bycatch on the Celtic Shelf. Report to the European Commission DG XIV-C-1 under study contract 92/3505.

Briggs, R. 1995. Discard sampling in Northern Ireland. *In* Assessment of discarding rates for commercial species of fish, pp. NI1.1–NI1.5. Compiled by J. Cotter. Final report to the Commission of the European Communities for contract BIO-ECO/93/003.

Camphuysen, C.J., Calvo, B., Durinck, J., Ensor, K., Follestad, A., Furness, R.W., Garthe, S., Leaper, G., Skov, H., Tasker, M.L., and Winter, C.J.N. 1995. Consumption of discards by seabirds in the North Sea. Final report EC DG XIV research contract BIOECO/93/10. NIOZ Report 1995-5, Netherlands Institute for Sea Research, Texel. 202 + 56 pp.

Connolly, P., Keatinge, M., and Woods, F. 1995. The assessment of discarding rates for commercial species of fish. *In* Assessment of discarding rates for commercial species of fish, pp. I1.1–I1.31. Compiled by J. Cotter. Final report to the Commission of the European Communities for contract BIO-ECO/93/003.

Cotter, J. (compiler) 1995. Assessment of discarding rates for commercial species of fish. Final report to the Commission of the European Communities for contract BIOECO/93/003.

Garthe, S., Camphuysen, C.J., and Furness, R.W. In press. Amounts discarded by commercial fisheries and their significance as food for seabirds in the North Sea. Marine Ecology Progress Series.

Greenstreet, S.P.R., and Hall, S.J. 1996. Fishing and the ground-fish assemblage structure in the northwestern North Sea: an analysis of long-term and spatial trends. Journal of Animal Ecology.

de Groot, S.J., and Lindeboom, H.J. (Eds.) 1994. Environmental impact of bottom gear on benthic

fauna in relation to natural resources management and protection of the North Sea. NIOZ Rapport 1994-11, RIVO-DLO Report C026/94. Netherlands Institute for Sea Research, Texel. 255 pp.

ICES. 1994a. Report of the Working Group on Ecosystem Effects of Fishing Activities. ICES CM 1994/ Assess/Env:1.

ICES. 1994b. Report of the Study Group on Seabird/Fish Interactions. ICES CM 1994/L:3.

ICES. 1996a. Report of the Working Group on the Assessment of Demersal Stocks in the North Sea and Skagerrak. ICES CM 1996/Assess:6.

ICES. 1996b. Report of the ICES Study Group on Seals and Small Cetaceans in European Seas. ICES CM 1996/N:1.

Jensen, H., Emslie, D., and Coull, K. 1994. Discards of fish species of no or very little economic interest. Final report to the Commission of the European Communities for contract 92/3508.

Kaiser, M.J., and Spencer, B.E. 1994. Fish scavenging behaviour in recently trawled areas. Marine Ecology Progress Series, 112: 41–49.

Kaiser, M.J., and Spencer, B.E. 1995 Survival of bycatch from a beam trawl. Marine Ecology Progress Series, 126: 31–38.

Kaiser, M.J., and Spencer, B.E. 1996. Behavioural responses of scavengers to beam trawl disturbance. *In* Aquatic predators and their prey. Ed. by S.P.R. Greenstreet and M.L. Tasker. Blackwell Science, Oxford.

McCaughran, D.A. 1992. Standardized nomenclature and methods of defining bycatch levels and implications. *In* Proceedings of the National Industry Bycatch Workshop, pp. 200–201. Ed. by R.W. Schoning, R.W. Jacobson, D.L. Alverson, T.G. Gentle and J. Auyong. Natural Resources Consultants, Seattle.

Northridge, S.P. 1996. A review of marine mammal bycatch observer schemes with recommendations for best practice. JNCC Report 219.

Pedersen, S.A. 1995. Feeding habits of starry ray (*Raja radiata*) in West Greenland waters. ICES Journal of Marine Science, 52: 43–53.

Peréz, N., Pereda, P., Uriarte, A., Trujillo, V., Olaso, I., and Lens, S. In press. Discards of the Spanish fleet in ICES Divisions. Final report of CEC Study Contract DG XIV, PEM/93/005. Instituto Español de Oceanografia, Vigo.

Peronnet, I. 1995. Estimation of discards by the French trawler fleet fishing in the Celtic Sea and the Bay of Biscay. *In* Assessment of discarding rates for commercial species of fish, pp F1.1–F1.15. Compiled by J. Cotter. Final report to the Commission of the European Communities for contract BIO-ECO/93/003.

Reeves, S. 1995. The Scottish discard sampling scheme. *In* Assessment of discarding rates for commercial species of fish, pp S1.1–S1.11. Compiled by J. Cotter. Final report to the Commission of the European Communities for contract BIOE-CO/93/003.

Spaans, A.L., Bukacinska, M., Bukacinski, D., and van Swelm, N.D. 1994. The relationship between food supply, reproductive parameters and population dynamics in Dutch lesser black-backed gulls *Larus fuscus*: a pilot study. IBN Research Report 94/9/BEON Report No. 94-15. Institute for Forestry and Nature Research, Wageningen. 65 pp.

Tamsett, D. 1995. English studies of discarding. *In* Assessment of discarding rates for commercial species of fish, pp. E1.1–E1.5. Compiled by J. Cotter. Final report to the Commission of the European Communities for contract BIOECO/93/003.

Vinther, M. 1996. Incidental catch of harbour porpoise (*Phocoena phocoena*) in the Danish North Sea gill-net fisheries: preliminary results. Proceedings of the Scientific Symposium on the North Sea Quality Status Report, 1994.

Weber, W. 1995. Estimation of cod discards caused by the fishery on roundfish in the German Bight 1982–1994. ICES CM 1995/B+G+H+J+K:6.

8 FIFTH INTERNATIONAL CONFERENCE ON THE PROTECTION OF THE NORTH SEA

Prepare work as required for the Intermediate Ministerial Meeting on Fisheries and the Environment (1997) and ultimately for the Fifth International Conference on the Protection of the North Sea, and address the implications of applying the precautionary principle to fisheries activity.

8.1 Intermediate Ministerial Meeting (1997)

The Working Group considered twelve questions from the *Reporting format on fisheries and fisheries-related species and habitat issues* prepared by the North Sea Secretariat. The questions and their respective responses are attached as Annex 4 to this report.

8.2 Precautionary Principle

8.2.1 Introduction

In the past, the precautionary principle in a marine context has most often been connected with the introduction of environmentally harmful substances. As an example, the Oslo and Paris Commissions (OSPAR) in their 1992 convention text (Article 2, paragraph 2a) described the precautionary principle as follows:

"The precautionary principle, by virtue of which preventive measures are to be taken when there are reasonable grounds for concern that substances or energy introduced, directly or indirectly, into the marine environment may bring about hazards to human health, harm living resources and marine ecosystems, damage amenities or interfere with other legitimate users of the sea, even when there is no conclusive evidence of a causal relationship between the inputs and effects."

Following on from the precautionary principle is the precautionary approach. This has been defined and used in a number of contexts.

The precautionary approach is mentioned in the 1992 Rio Declaration of the UN Conference on Environment and Development in which Principle 15 states:

"In order to protect the environment, the precautionary approach shall be widely applied by states according to their capabilities. Where there are threats of serious or irreversible damage, lack of full scientific certainty shall not be used as a reason for postponing cost-effective measures to prevent environmental degradation."

Another example is from the global London Convention (Resolution LDC.44 (14)), which states that:

"Contracting Parties shall be guided by a precautionary approach to environmental protection whereby appropriate preventive measures are taken when there is reason to believe that substances or energy introduced in the marine environment are likely to cause harm even when there is no conclusive evidence to prove a causal relation between inputs and their effects".

There is, however, no reason why the principle, and any approaches deriving from it, should not be applied to all human activities which can potentially harm the environment. In relation to fisheries the FAO has elaborated a Code of Conduct for Responsible Fisheries (Garcia, 1994; FAO, 1995a). This characterises the concept of precaution, and provides specific guidelines for various aspects of fisheries. In elaborating the precautionary approach, it declares that:

"States should apply the precautionary approach widely to conservation, management and exploitation of living aquatic resources in order to protect them and preserve the aquatic environment. The absence of adequate scientific information should not be used as a reason for postponing or failing to take conservation and management measures".

"In implementing the precautionary approach, States should take into account uncertainties relating to the size and productivity of the stocks, reference points, stock condition in relation to such reference points, levels and distribution of fishing mortality and the impact of fishing activities, including discards, on non-target and associated or dependent species, as well as environmental and socioeconomic conditions."

"States should take appropriate measures to minimize waste, discards, catch by lost or abandoned gear, catch of non-target species, both fish- and non-fish species, and negative impacts on associated or dependent species, in particular endangered species... Such measures may include technical measures related to fish size, mesh size or gear, discards, closed seasons and areas, zones preserved for selected fisheries."

The main concerns about the application of the precautionary approach to fisheries stem from a lack of clarity on what the approach would mean in practice. Within the fishing industry there have been concerns that its application would mean the closing down of certain fisheries, and the adoption of regulations that would be unnecessarily restrictive.

In general a precautionary approach is intended to give more weight to long-term considerations in the management of fisheries and the ecosystem in which they operate, and seeks to consider the needs of future generations and the avoidance of changes that are not potentially reversible.

It is acknowledged that all fishing activities have environmental impacts and that it is inappropriate to assume that these impacts are negligible until they have been proven to be important. Even so, a precautionary approach does not imply that no fishing can take place until all potential impacts have been assessed and found to be negligible. The approach requires that all fishing activities should be subject to review and authorization, and that a management plan should be put in place that specifies the management objectives and contains plans for assessing and monitoring the impact. When authorizations are given to a fishery it also requires that the standard of proof for possible negative effects should be commensurate with the potential risk to the resource and the ecosystem in general.

A precautionary approach requires that management objectives are specified and that a management plan is elaborated which contains explicit considerations of the actions that will have to be taken to avoid specific undesirable outcomes. Although science does have an important role in developing the objectives it is the

managers' responsibility, with the help of the scientific community, that objectives for both fisheries and the wider ecosytem are clearly stated and transferred into operational targets and constraints.

A precautionary approach should also involve procedures for estimating the uncertainty involved in the assessment and monitoring of the impacts. The marine ecosystem is complex and subject to a lot of variation due to natural fluctuations. In most cases it is difficult to distinguish between changes due to natural and to anthropogenic causes. As a consequence, the FAO guidelines (FAO, 1995b) put considerable emphasis on establishing ways to deal with uncertainty in a management context. When the statistical power of the assessment and monitoring system is low, management should be cautious.

8.3 Implications for Targets in Fisheries Management

With respect to management measures it should be recognised that, for a number of species, fishing effort is above the so-called 'Minimum Biologically Acceptable Level', and reductions in fishing effort have been recommended by various ICES assessment groups. One of the greatest effects of fishing on the ecosystem is the direct effect of harvesting target species. Therefore, reductions in effort can serve both the target fisheries and the ecosystem. Such reductions in fishing effort would constitute one of the most significant steps that could be taken to implement a precautionary approach for the ecosystem while simultaneously improving fishery yields and the sustainability and continuity of fisheries.

The specification of Minimum Biologically Acceptable Levels (MBALs) of spawning stock biomass can be considered an application of the precautionary principle to fisheries management. The management objective is to maintain biomass above MBAL. However, a clear specification of the acceptable level of risk of having stocks reduced below MBAL is lacking from the current management scheme. The implied statistical power of this comparison is 0.5—a stock is overfished if the point estimate of biomass is less than MBAL.

8.4 Implications for Targets in Ecosystem Management

Under the precautionary principle, biological components of the ecosystem and their habitats should be protected from adverse fishing effects. This requires that objectives be set, expressed in measurable terms, not only for target species but also for non-target species. It will be necessary to establish both measurable targets and constraints. Measures will then need to be taken to meet these objectives and to manage the ecosystem. Measures could include the elimination/avoidance of by-catch, reduction in discarding and in the direct mortality of non-target species. This also includes measures to reduce adverse effects of some types of fishing gear on benthic species and communities.

8.5 Implications for the Working Group on Ecosystem Effects of Fishing Activities

One of the more novel aspects of the precautionary approach to fisheries is the integration of fisheries and ecosystem management. As outlined above, ecosystem management will need objectives and measurable targets. These would need to exist alongside the longer-established fisheries targets. The Working Group on Ecosystem Effects of Fishing Activities has attempted in several cases to use information gathered for fisheries management purposes to describe the ecosystem and its components. Some of these descriptors may be of particular use in establishing practical measurable targets for the ecosystem and the effects of fisheries on it. Any such descriptor would have the additional benefit of not needing further resources to gather relevant information.

Although WGECO has not devoted a great deal of time to specific management issues, it could start by examining such questions as: Are there ecosystem objectives that would not be met by satisfying the single-species fisheries management objectives? What level of confidence that an activity is having no effect should be chosen? What type of actions might be of particular value in managing the ecosystem? Which particular actions could be taken to improve the position of (damaged) areas of the environment? What measurable properties of the ecosystem should be managed for? Is it possible to elaborate a framework to identify the most important components of the marine environment?

8.6 References

FAO. 1995a. Code of Conduct for Responsible Fisheries. Food and Agriculture Organization of the United Nations. Rome. 41 pp.

FAO. 1995b. Precautionary approach to fisheries. Part 1: Guidelines on the precautionary approach to capture fisheries and species introductions. Elaborated by the technical consultation on the precautionary approach to capture fisheries (including species introductions). Lysekil, Sweden, 6–13 June 1995. FAO Fisheries Technical Paper: 350(1). Rome, FAO. 47 pp.

Garcia, S.M. 1994. The precautionary approach to fisheries with reference to straddling fish stocks and highly migratory fish stocks. FAO Fisheries Circular No. 871. FAO, Rome.

9 ICES AREA

Review relevant activities within the entire ICES area with a view to extending the activities of the Working Group to ecosystems outside the North Sea.

The origins of WGECO lie partly in the North Sea Ministerial Conference processes. Many terms of reference given to the group have related directly to the

North Sea. Other terms of reference have not focused on the North Sea, but in several cases the majority of the responses to them has been made using North Sea material since that was all that was available to the group. However, studies from outside the North Sea have been used when available, as can be seen in most parts of the current report.

In most cases, descriptions of ecosystem effects of fishing activities in various areas require both a description of the components of the ecosystem and a study of the processes in those areas. Table 9.1 indicates those parts of the ecosystem where the group knows of descriptions of the fauna/flora adequate for some assessment of the effects of fishing activities, and where there are known relevant studies under way. The group would welcome further information to improve this table. For this exercise, the eastern ICES area has been divided into the regions to be used by OSPAR in their forthcoming quality status reports on the Northeast Atlantic (see Figure 9.1), with the addition of regions for the Baltic Sea (see Figure 9.2) and Northwest Atlantic (see Figures 9.3.a and 9.3.b). Note that Table 9.1 does not indicate availability or quality of the information.

The group was unable to prepare a comprehensive response to this term of reference in the time available. However, from the brief analysis summarised in Table 9.1, it is clear that there is considerable research activity on this topic outside the North Sea. WGECO believes that the increased opportunities for contrasting findings from the North Sea with those from other systems would be very valuable and it would welcome the opportunity to further broaden its activities to other areas.

9.1   References

ICES. 1980. Extension to the Baseline Study of Contaminant Levels in Living Resources of the North Atlantic. ICES Cooperative Research Report, No. 95. 57 pp.

ICES. 1987. An updated description of the ICES Statistical Area (North), Statistical Sub-Areas, Divisions, and Sub-Divisions. ICES CM 1987/D:22. 17 pp.

OSPAR. 1995. Assessment and monitoring: the Joint Assessment and Monitoring Programme. Oslo and Paris Commissions, London. 45 pp.

10   FOOD FOR THOUGHT

10.1   Towards a Better Way of Investigating the Impact of Fishing on Species Assemblages

The approaches to describing fish assemblage structure used to date and summarised in this report are effectively data-driven. The data are assembled and analysed using a number of techniques such that all conceivable aspects of the structural pattern are captured. Consequently, the

questions asked are broad, e.g., does the assemblage structure change in some way over time or between areas. The problems arise, however, when it comes to determining the causes of any change and, in particular, trying to decide whether the changes discerned are consistent with what might be expected of a species assemblage undergoing long-term chronic fishing disturbance.

Without a theory as to how the processes which mould the structure of the fish assemblage operate, it is difficult to predict what the likely effect of fishing might be; testable hypotheses cannot be developed. The effects of disturbance on community characteristics, such as species diversity, are far from clear (e.g., Death and Winterbourne, 1995) and may be particularly complex in marine ecosystems (Russo, 1982; Menge and Sutherland, 1987). Nutrient enrichment events have been shown to result in both increases and decreases in species diversity (Mirza and Gray, 1981; Russo, 1982). Without a testable hypothesis we cannot distinguish between several opposing explanations.

Initially, community structure was thought to be decided through competitive equilibrium processes (MacArthur, 1970; MacArthur and Levins, 1967), an idea supported by many laboratory and field studies (e.g., Park, 1948; 1954; Pianka 1973; 1975; 1983; Fenchel, 1975; Davidson, 1977). However, communities were identified which, while apparently stable, were clearly not at competitive equilibrium (e.g., Paine 1966; Patrick, 1975). Predation was claimed to be the dominant process governing the structure of communities. In its absence communities progressed to competitive exclusion; in its presence an alternative stable state always existed (Connell, 1975).

Huston (1979) developed a model—the 'dynamic equilibria model'—of species diversity, which took into account the opposing influences of competition and predation. The fundamental predictions of this model are encompassed in Figure 10.1.1. The theory suggests that at low disturbance frequency, diversity is highest at low–intermediate productivity. At the lowest levels of productivity population growth is so slow that extinction is a distinct possibility. At higher productivity the scope for growth is sufficiently high that populations of the dominant competitors will grow fast enough that they will be able to capture an unequal share of the resources and inhibit population growth of the subordinate competitors. These species will be excluded before the next disturbance event. At higher disturbance frequencies, diversity will be lower at low productivity levels; some populations will be unable to grow fast enough to recover sufficiently between disturbances and they will be driven to extinction. At intermediate productivity levels populations will be able to grow fast enough to avoid extinction, but the growth of dominant competitors will be checked, thereby preventing competitive exclusion. Species diversity in areas of high disturbance frequency and high productivity will be

lower than in areas of intermediate levels of both because the high frequency of disturbance will probably eliminate k-selected species and so reduce the total species pool available. Huston (1994) examined a large number and variety of species assemblage data sets and generally found that variation in species diversity could be explained in terms of the dynamic equilibria model.

What insights might such a model provide with respect to examining changes in the structure of fish assemblages and relating these to variations in fishing activities? Firstly, it demonstrates that, without understanding the other factors which interact with and affect assemblage structure, we can have little idea how the assemblage will respond to variations in fishing disturbances. Given this, it is difficult to state categorically that a particular alteration in assemblage structure is a direct response to the disturbance regime. Figure 10.1.2 illustrates this, showing that the relationship between species diversity and disturbance is strongly influenced by environmental productivity. Figure 10.1.2 indicates that in rich environments increasing disturbance results in **increased** species diversity up to high levels of disturbance. Conversely, in poor environments species diversity **declines** as disturbance increases. In moderately productive environments diversity increases as disturbance rates rise to intermediate levels, and thereafter decreases. Secondly, the model illustrates that it may be difficult to determine any relationship between assemblage structure and disturbance. Figure 10.1.3 examines the mean (and standard deviation) species diversity predicted by the model at three levels of fishing disturbance (the disturbance levels are indicated on Figure 10.1.1). This graph suggests that significant variations in species diversity would be unlikely to be detected from sets of random samples taken across the range of possible levels of productivity.

Models such as this can provide testable hypotheses. For example, the ICES North Sea Benthos database holds data for benthic biomass density (g m^{-2}) over the whole North Sea (Heip et al., 1992; Basford et al., 1993), and production can be estimated from these data using empirical relationships (Duineveld et al., 1991; Bryant et al., 1995). Thus, for the benthivorous component of the groundfish assemblage we can use the model to predict the relationships between fish species diversity and benthic production at varying levels of fishing disturbance. Such relationships are shown in Figure 10.1.4 for the same three levels of fishing disturbance examined in Figure 10.1.3 and indicated on the model (Figure 10.1.1.). The model predicts specific relationships. At low benthic production levels fish species diversity behaves as we might naively expect; it is high at low levels of disturbance and decreases as disturbance increases. At intermediate benthos production levels the fish assemblage diversity ranking starts to change so that when benthic production is high, the order is completely reversed; fish diversity is greatest where fishing disturbance is most prevalent and *vice versa*. If such relationships are found in our data then we

can be more confident regarding the role of fishing disturbance in influencing fish assemblage structure.

10.2 References

Basford, D.J., Eleftheriou, A., Davies, I.M., Irion, G., and Soltwedel, T. 1993. The ICES North Sea benthos survey: the sedimentary environment. ICES Journal of Marine Science, 50: 71–80.

Bryant, A.D., Heath, M.R., Broekhuizen, N., Ollason, J.G., Gurney W.S.G., and Greenstreet, S.P.R. 1995. Modelling the predation, growth and population dynamics of fish within a spatially-resolved shelf sea ecosystem model. Netherlands Journal of Sea Research, 33: 407–421

Connell, J.H. 1975. Some mechanisms producing structure in natural communities. In Ecology and Evolution of Communities, pp. 460–490. Ed. by M.L. Cody and J.M. Diamond. Belknap Harvard University Press, Cambridge, Massachusetts.

Davidson, D.W. 1977. Species diversity and community organization in desert seed-eating ants. Ecology, 58: 711–724.

Death, R.G., and Winterbourne, M.J. 1995. Diversity patterns in stream benthic invertebrate communities: the influence of habitat stability. Ecology, 76: 1446–1460.

Duineveld, G.C.A., Künitzer, A., Niermann, U., de Wilde, P.A.J., and Gray, J.S. 1991. The macrobenthos of the North Sea. Netherlands Journal of Sea Research, 28: 53–65.

Fenchel, T. 1975. Character displacement and coexistance in mud snails (Hydrobiidae). Oecologia, 20: 19–32.

Heip, C., Basford, D., Craeymeersch, J.A., Dewarumez, J.-M., Dorjes, J., de Wilde, P., Duineveld, G., Eleftheriou, A., Herman, P.M.J., Niermann, U., Kingston, P., Künitzer, A., Rachor, E., Rumohr, H., Soetaert K., and Soltwedel, T. 1992. Trends in biomass, density and diversity of North Sea macrofauna. ICES Journal of Marine Science, 49: 13–22.

Huston, A.H. 1979. A general hypothesis of species diversity. American Naturalist, 113: 81–101.

Huston, A.H. 1994. Biological diversity: the coexistence of species on changing landscapes. Cambridge University Press. 681 pp.

MacArthur, R.H. 1970. Species packing and competitive equilibrium for many species. Theoretical Population Biology, 1: 1–11.

MacArthur, R.H., and Levins, R. 1967. The limiting similarity, convergence and divergence of coexisting species. American Naturalist, 101: 377–385.

Menge, B.A., and Sutherland, J.P. 1987. Community regulation: variation in disturbance, competition and predation in relation to environmental stress and recruitment. American Naturalist, 130: 730–757.

Mirza, F.B., and Gray, J.S. 1981. The fauna of benthic sediments from the organically enriched Oslofjord, Norway. Journal of Experimental Marine Biology and Ecology, 54: 181–207.

Paine, R.T. 1966 Foodweb complexity and species diversity. American Naturalist, 100: 65–75.

Park, T. 1948 Experimental studies of interspecific competition. I. Competition between populations of the flour beetles *Tribolium confusum* Dival and *Tribolium castaneum* Herbst. Ecological Monographs, 18: 267–307.

Park, T. 1954. Experimental studies of interspecific competition. II. Temperature, humidity and competition in two species of *Tribolium*. Physiological Zoology, 27: 177–238.

Patrick, R. 1975. Structure of stream communities. *In* Ecology and evolution of communities, pp. 445–459. Ed. by M.L. Cody and J.M. Diamond. Belknap Harvard University Press, Cambridge, Massachusetts.

Pianka, E.R. 1973. The structure of lizard communities. Annual Review of Ecology and Systematics, 4: 53–74.

Pianka, E.R. 1975. Niche relations of desert lizards. *In* Ecology and evolution of communities, pp. 292–314. Ed. by M.L. Cody and J.M. Diamond. Belknap Harvard University Press, Cambridge, Massachusetts.

Pianka, E.R. 1983. Evolutionary Biology, 3rd Edition. Harper & Row, New York.

Russo, A.R. 1982. Temporal changes in fish community structure near a sewage ocean outfall, Mokapu, Oahu, Hawaii. Marine Environmental Research, 6: 83–98.

11 ELECTION OF CHAIRMAN

Steve Hall intimated that, in view of his new job in Australia, he would be unable to continue as WGECO Chairman after the current meeting. Two nominations for his replacement were put forth: Mark Tasker and Jake Rice. The Chairman informally asked the views of members of the working group, most of whom had indicated their willingness to work with either nominee. The Chairman decided that no vote would be taken and the names of both nominees would be forwarded to the two parent committees, ACFM and ACME, for consideration at their May and June meetings, respectively.

12 RECOMMENDATIONS

WGECO agreed to several recommendations; these are provided in Annex 5.

13 APPROVAL OF REPORT

WGECO considered and approved the report of the meeting. The meeting closed at 14.00 hrs on 21 March 1996.

AGENDA

1) Opening of the meeting.

2) Adoption of the agenda.

3) Review of intersessional activities.

4) Communities and assemblages
    Analyze survey data in terms of appropriate summary parameters for species assemblages, with a view to continuing the study of biodiversity and changes in community structure.

5) Non-target fish populations.
    Examine trends in abundance of non-target fish species using trawl survey data and other appropriate data sets.

6) Quantifying vulnerability.
    Refine and implement approaches for quantifying the susceptibility of species, including their juvenile stages, to fishing effects.

7) The effects of reducing fishing mortality.
    Analyze available data on system responses in areas where fishing mortality has been reduced.

8) Estimating discards and their utilization
    Summarize any new work undertaken on the estimation of discards or their utilization by scavengers.

9) Responses to CONSSO Questionnaire on fisheries and fisheries-related species and habitat issues.
    Prepare work appropriate for the Intermediate Ministerial Meeting on Fisheries and the Environment (1997) and ultimately for the Fifth International Conference on the Protection of the North Sea and address concerns about the [possible] application of the precautionary principle in a fisheries context.

    Q 17–18
    Q 21–23
    Q 25
    Q 26

10) The precautionary principle.

11) Food for thought.

12) Proposals for a further meeting.

13) Election of new chairman.

14) Consideration and approval of the report.

ANNEX 2

LIST OF PARTICIPANTS

Name	Address	Telephone	FAX	E-mail
Jean Boucher	IFREMER Pêche Aquaculture BP 70 29280 Plouzane France	+33 982 24615	+33 9822 4547	jboucher@ifremer.fr
Andy Brand	Port Erin Marine Laboratory University of Liverpool Port Erin Isle of Man IM9 6JA, UK	+44 1624 832027	+44 162 835788	arbrand@liverpool.ac.uk
Jeremy Collie	University of Rhode Island Graduate School of Oceanography Narragansett, RI 0282-1197 USA	+1 401 874 6859	+1 401 874 6160	jcollie@limanda.gso.uri.edu
Johan Craeymeersch	Netherlands Institute of Ecology Centre for Estuarine and Coastal Ecology Vierstraat 28 4401 EA Yerseke Netherlands	+31 1135 71902	+31 1135 73616	craeymer@cemo.nioo.knaw.nl
Petter Fossum	Institute of Marine Research P.O. Box 1870 Nordnes N-5024 Bergen Norway	+47 5523 8500	+47 552 38584	petter.fossum@imr.no
Chris Frid	Dove Marine Laboratory University of Newcastle Cullercoats, North Shields NE30 4PZ, UK	+44 191 252 4850	+44 191 252 1054	c.l.j.frid@ncl.ac.uk
Henrik Gislason	Danmarks Fiskeri- og Havundersøgelser Charlottenlund Slot 2920 Charlottenlund Denmark	+45 33 963361	+45 33 963333	hg@dfu.min.dk
Simon Greenstreet	SOAEFD, Marine Laboratory P.O. Box 101, Victoria Road Aberdeen AB9 8D3 UK	+44 224 295417	+44 224 295511	greenstreet@marlab.ac.uk
Bas de Groot	Netherlands Institute for Fishery Investigations P.O. Box 68 1970 AB IJmuiden Netherlands	+31 2555 64646	+31 2555 64644	

Name	Address	Telephone no.	Fax no.	E-mail
Steve Hall* (Chairman)	SOAEFD, Marine Laboratory P.O. Box, 101, Victoria Road Aberdeen AB9 8DB Scotland	+44 1224 295456	+44 1224295511	hallsj@marlab.ac.uk
John Hall	Dove Marine Lab. University of Newcastle Cullercoats North Shields, Tyne and Wear NE30 4PZ, UK	+44 191 252 4850	+44 191 252 1054	john.hall@ncl.ac.uk
Hans Hallbäck	Institute of Marine Research Box 4 453 21 Lysekil Sweden	+46 523 18715	+46 523 13977	
Henrik Jensen	Danmarks Fiskeri- og Havundersøgelser Charlottenlund Slot 2920 Charlottenlund	+45 33 963370	+45 33 963333	hj@fimdfh.fim.dk
Han Lindeboom	Netherlands Institute for Sea Research P.O. Box 59 1790 AB Den Burg, Texel Netherlands	+31 2223 69410	+31 2223 19674	hanl@nioz.nl
Henn Ojaveer	Estonian Marine Institute Lai Str. 32 EE-0001 Tallinn Estonia	+372 2601 529	+372 6313 004	
Ignacio Olaso	Instituto Español de Oceanografía Laboratorio de Santander Aptdo 240 39080 Santander, Spain	+34 42 275033 " " 225062	+34 42 275 072	
Jake Rice	Department of Fisheries & Oceans Pacific Biological Station Hammond Bay Road Nanaimo, B.C. V9R 5K6 Canada	+1 604 7567136	+1 604 756 7053	ricej@pbs.dfo.ca
Adriaan Rijnsdorp	Netherlands Institute for Fishery Investigations P.O. Box 68 1970 AB IJmuiden Netherlands	+31 2555 64646	+31 2555 64644	a.d.rijnsdorp@rivo.agro.nl
Yorgos Stratoudakis	SOAFD, Marine Laboratory P.O. Box, 101, Victoria Road Aberdeen AB9 8DB Scotland	+44 1224 295522	+44 1224295511	stratoudy@marlab.ac.uk

,*As of 1 July 1996:

School of Biological Sciences   Telephone: +61 8 201 2229
The Flinders University of S.A.   Telefax: +61 8 201 3015
GPO Box 2100   E-mail: stephen.hall@flinders.edu.au
Adelaide 5001, S.A.
Australia

Name	Address	Telephone no.	FAX no.	E-mail
Mark Tasker	Seabirds and Cetaceans Branch Joint Nature Conservation Committee Dunnet House Thistle Place Aberdeen AB1 1U2 Scotland, UK	+44 1224 655701	+44 1224 621488	seabirds.jncc@aberdeen. ac.uk
Daniel Valentinsson	Institute of Marine Research Box 4 45321 Lysekil Sweden	+46 5231 8712	+46 523 13977	d.valentinsson@imr.se
Wolfgang Weber	Bundesforschungsanstalt für Fischerei Palmaille 9 22767 Hamburg Germany	+49 40389 05246	+49 403 8905263	
Wanda Zevenboom	RWS-North Sea Directorate P.O. Box 5807 2280 HV Rysivyk Netherlands	+31 703366883 or +31 7033 66600	+31 703 900691	

ANNEX 3

LIST OF WORKING DOCUMENTS

WGECO 1996/2/1   Draft agenda

WGECO 1996/4/1   Hall, S.J. Trends in body size and species richness in marine fish assemblages.

WGECO 1996/4/2   Greenstreet, S.P.R. Towards a better way of investigating the impact of fishing disturbance on fish species assemblages.

WGECO 1996/4/3   Gislason, H., and Rice, J. Modelling the effect of changes in fishing effort on the size and diversity spectra of exploited fish assemblages.

WGECO 1996/6/1   Hall, S.J., and Frid, C. Refine and implement approaches for quantifying the susceptibility of species, including their juvenile stages.

WGECO 1996/6/2   Stratoudakis, Y. Mortality of common dab (*Limanda limanda*) and long rough dab (*Hippoglossoides platessoides*) in demersal fisheries of the North Sea and the west coast of Scotland.

WGECO 1996/6/3   Collie, J.S., Escanero, G.A., and Valentine, P.C. Effects of bottom fishing on the benthic megafauna of Georges Bank.

WGECO 1996/6/4   Rochet, M.-J. Effets de la peche sur la diversite des strategies et tactiques demographiques des poissons.

WGECO 1996/8/1   Olaso, I. Discards of the Spanish fleet in ICES Divisions, and importance of blue whiting discards in the diet of demersal scavengers fish species.

ANNEX 4

FIFTH INTERNATIONAL CONFERENCE ON THE PROTECTION OF THE NORTH SEA
Reporting format on fisheries and fisheries-related species and habitats issues

At the North Sea Ministerial Conference in June 1995, the ministers of countries bordering the North Sea agreed to set up a Secretariat in the Ministry of Environment in Oslo to prepare for the next North Sea Conference scheduled for 2000–2002 and, more immediately, for the Intermediate Ministerial Meeting (IMM) to be held in early 1997. The Intermediate Ministerial Meeting will focus on the relationship between fisheries and the environment. In preparation for IMM 1997, a *Questionnaire on fisheries and fisheries-related species and habitats issues* was prepared by the North Sea Secretariat and ICES was invited to provide material and information relevant to some of the questions. The ICES Council agreed at the 1995 Annual Science Conference that ICES should respond to this request. The Working Group on Ecosystem Effects of Fishing Activities (WGECO) was given the remit to consider the following questions at their 1996 meeting and to draft responses to the questions for the Advisory Committee on Fishery Management (ACFM) and the Advisory Committee on the Marine Environment (ACME). This response is based primarily on data synthesised and presented in reports of this and previous meetings of the Working Group.

Question 14.a. Indicate the quantity of offal discarded at sea in tonnes per year.

Fishery	Quantity of offal discarded at sea (tonnes/year)	Comments
International North Sea estimate	65,000–70,000	see below

Comments

- The estimate is for the early 1990s.

- The estimate is based on combined landings data (gutted weight) from roundfish, flatfish and shellfish (trawls) fisheries where most fish species are usually landed gutted. We are unable to divide the total amount between these fisheries. Most other important fisheries (pelagic and industrial) fisheries are not thought to generate significant amounts of offal because fish are landed whole.

- Estimations use species-specific conversion factors to get total from gutted weight (for details see ICES CM 1994/Assess/Env:1).

- Larger quantities of offal are expected in areas where large individuals of species with higher conversion factors are targeted. In that sense flatfish fisheries in ICES Division IVc (North Sea) will produce smaller quantities of offal than roundfish fisheries in offshore grounds of ICES Divisions IVa and IVb (North Sea).

- The above indirect method of estimation assumes that [all] landed fish are gutted and the offal discharged at sea. The former is not always the case in the Scottish fisheries but extrapolations to other national fleets cannot be made. In Scotland, haddock and, mainly, whiting can be landed in large proportions as "rounders" (ungutted fish), especially in periods where strong year classes reach marketable size. In these cases vessels that target fish around minimum landing size (mainly inshore areas) land a good deal of the catch ungutted due to the increased labour involved with gutting.

Question 14.b. Indicate the quantity of offal landed and utilised for some purpose.

No data available to the group.

Question 15.a.   Indicate for the different fisheries, the percentages of the total fish catches which are discarded at sea..

Fishery	0–25%	25–50%	50–100%
Roundfish		**	
Flatfish		**	
Shellfish (trawls)			**
All other categories	**		

Additional comments

• The percentages refer to total numbers of all fish species (target and by-catch). Percentages in biomass are somewhat lower.

Question 15.d.   Rank the main reasons for the discarding of fish in the different fisheries. Rank from 1 to 5, where 1 is the most important reason and 5 is insignificant.

Fisheries	No commercial value	High grading	Quotas exceeded	Juvenile fish	Undersized fish
Roundfish	2	4	3	5	1
Flatfish	2	4	3	5	1
Shellfish Trawls	1	5	4	2	1
Shellfish Dredge	1	-	-	-	-
Purse seine	4	2	3	-	1
Gill net	1	-	-	-	-

Additional comments

• In roundfish fisheries undersized fish from target species are extensively caught despite the continual increase of nominal mesh size over the past 15 years. The impact of minimum landing size (MLS) on the discarding decisions seems to be much higher for fishers operating in inshore grounds where fishers tend to land everything they legally can. High grading is mainly observed in offshore areas where the catch consists of more and larger fish and considerations about MLS become secondary. In the North Sea high grading decisions are mainly driven by market forces and considerations of potential quota restrictions rather than hold capacity.

• In certain shellfish trawl fisheries the use of a smaller mesh size leads to the capture of more species and more small fish. In such fisheries fish are considered by-catch and there are restrictions on the amounts that can be landed.

Question 15.e.   Rank the main reasons for the discarding of fish species. Rank from 1 to 5 if possible, where 1 is the most important reason and 5 is insignificant.

Fish species	No commercial value	High grading	Quotas exceeded	Juvenile fish	Undersized fish
Haddock	5	3	2	4	1
Whiting	5	2	3	4	1
Cod	4	3	1	-	2
Saithe	4	2	1	-	3
Plaice	4	2	3	-	1
Sole	4	2	3	-	1

• Fishers often claim that quota limitations is one of the main reasons for discarding fish. It is however unclear to what extent quota restrictions lead to increased discarding or to "un-reported" and "over quota" landing. Discard sampling schemes are unable to address fully the above issue due to the lack of complete randomisation in the selection of the sampled vessels.

Question 18.a.   Which marine species including seabirds and marine mammals benefit, directly or indirectly, from fishing activities.

[Note: The term 'benefit' is taken to mean an increase in population size.]

Apart from target species and seabirds there is little published information on actual trends in the abundance of marine species that may benefit from fishing activities. Furthermore, even if such trends have been established it remains uncertain whether there is a direct or indirect relationship with fisheries. Species may benefit from the discards created, but at the same time increased mortality due to fishing can counteract any benefit gained. With the above caveats in mind we have drawn the following conclusions:

Species most likely to benefit

• Seabirds: scavengers on floating discards (mainly fish). Eight species utilise fishery offal and discards on a large scale: fulmar (*Fulmarus glacialis*), gannet (*Morus bassanus*), great skua (*Catharacta skua*), common gull (*Larus canus*), lesser black-backed gull (*Larus fuscus*), herring gull (*Larus argentatus*), great black-backed gull (*Larus marinus*) and kittiwake (*Rissa tridactyla*) (Camphuysen *et al.*,1995).

Species which probably benefit

• Starfish (*Asterias rubens*)

Species which possibly benefit

• Opportunistic benthic species like spionid worms.

Question 18.b.   How do these species benefit from fishing activities.

Species benefit from fisheries because they are scavengers on dead discards returned to the sea. Other organisms scavenge on dead, damaged or exposed benthic animals in the trawl path and escape relatively undamaged through the meshes of the gear (starfish). [Note: Few seabirds are killed by fisheries in the North Sea.]

Other species may benefit from reduced competition with, or predation by, more vulnerable species, or because their scavengers are removed by fisheries. However, we are aware of no conclusive data on these possibilities.

Question 19.a.   Which non-target species are negatively affected, directly or indirectly, by fishing activities?

[Note: The term 'negative' is taken to mean a decrease in the size of a population.]

For many non-target species there is little published information on actual trends in their abundance, although some species like oysters, lobsters, greater weever, rays, and sharks have completely disappeared from certain areas of the North Sea. However, even if decreasing trends have been established it remains uncertain whether there is a direct or indirect relationship with fisheries. Also other influences (e.g., climatic changes, pollution, eutrophication) may affect the abundance. On the other hand, it is clear that if fisheries impose an increased mortality on non-target species, there is a direct short-term negative effect on that species. Whether this has long-term negative implications on the population is very difficult to establish and depends, among other things, on the local fishing intensity and gears used. It should also be borne in mind that not all fisheries target all species so, for example, the crabs which are targets for static gears may be negatively affected by towed gears for which they are non-target. Such cases are marked with an asterisk (*) in the following lists.

With the above caveats in mind we have drawn the following conclusions:

Species most likely to be affected

- Oyster* *(Ostrea edulis)*
- Lobster* *(Homarus gammarus)*
- Sessile erect epifauna (e.g., sponges, hydrozoa, anemones, Bryozoa, reef building and tube dwelling Polychaetes)
- Common skate* *(Raja batis)*
- Stingray *(Dasyatis pastinaca)*
- Thornback ray* *(Raja clavata)*
- Spurdog *(Squalas acanthus)*
- Smooth hound *(Mustulus mustulus)*
- Harbour porpoise *(Phocoena phocoena)*
- Common scoter *(Melanitta nigra)*
- Common eider *(Someteria mollissima)*

Species probably affected

- Bivalves living at or in the upper 10 cm (e.g., *Arctica islandica, Modiolus modiolus, Dosinia lupinus, Dosinia exoleta, Acanthocardium,*and *Thracia).*
- Gastropod *(Neptunea antiqua)*
- Edible crab* *(Cancer pagurus)*
- Dogfish *(Scyliorhinus canicula)*

Species possibly affected

- Whelk* *(Buccinum undatum)*
- Helmet crab *(Corystes cassivelaunus)*
- Cuttlefish* *(Sepia officinalis)*
- Arctic tern *(Sterna paradisea)*
- Guillemot *(Uria aalge)*
- Kittiwake *(Rissa tridactyla)*

Question 19.b.   How are these species negatively affected?

For all species the overall basis for any negative effect is that population growth rates are insufficient to compensate for the increased mortality that fishing imposes. For some species, part of this effect is due to the destruction of suitable habitats.

Species most likely to be affected

- Oyster. Oyster were common in many parts of the North Sea until about the 1870s, at which time there was an active fishery for the species. Many factors may have influenced their decline, including hydrographic changes, disease and overfishing. Given that their numbers are now so low that fishing is uneconomic, they are no longer a target species and their numbers probably remain low through the action of mobile gears. The destruction of suitable habitats through the removal or burial of stones may also play a role.

- Lobster. Incidental mortality from mobile gear and habitat destruction (burial and removal of stones).

- Sessile erect epifauna. These taxa are vulnerable and easily damaged or killed in the trawl path due to contact with the hard parts of mobile gears. Sessile species are also found on the hard substratum and the removal and burial of stones and larger shells also reduces habitat availability.

- Common skate, stingray, thornback ray, smooth hound, spurdog. These species are caught as by catch in mobile gear and are particularly vulnerable owing to their life-history characteristics. Also, eggs are detached from the seabed and are damaged by mobile gear.

- Harbour porpoise. Drowned in bottom set fixed nets.

- Common scoter, common eider. In the southern North Sea/Wadden Sea, harvesting of shellfish (mussels, cockles and *Spisula*) may have limited the amount of food available in the past decade.

Species probably affected

- Bivalves living at or in the upper 10 cm and *Neptunia antiqua.* These taxa are vulnerable and easily damaged or killed in the trawl path due to contact with the hard parts of mobile gears.

- Edible crab. Incidental mortality from mobile gear and habitat destruction (burial and removal of stones).

Species possibly affected

- Whelk. Eggs are detached from the seabed and adults can be damaged by contact with the hard parts of mobile gear.

- Helmet crab. Incidental mortality from mobile gear and habitat destruction (burial and removal of stones).

- Cuttlefish. Eggs are detached from the seabed and are easily damaged by mobile gear.

- Arctic tern, guillemot, kittiwake. On a local scale breeding success or winter mortality of populations may be affected by fishing for their prey species. Overall, however, populations of these species have increased during the 20th century.

Question 20.   Give examples of individual species or groups of species which are positively or negatively affected by fishing activities and that could be the subject for more comprehensive case reviews.

[Note: This list is restricted to non-target species.]

Groups   Sharks and rays, maerl (calcareous algae), seaduck, auks, non-seabird scavengers, erect sessile biota.

Species   *Arctica islandica, Phocoena phocoena, Rissa tridactyla, Sterna paradisea, Sabellaria, Modiolus, Limanda limanda*

Question 25.a. What is the evidence or indication of damage to seabeds and their communities from the use of different types of towed gears?

General Remarks

Direct physical disturbance and mortality of benthic fauna in the trawl path due to various types of trawling in different areas of the North Sea have been demonstrated by video, side scan sonar, REMOTS, bottomcore analyses, and analyses of the benthos in the trawl path (see *ICES Coop. Res. Rep.*, No. 200, 1995, and No. 203, 1994; BEON, 1994; IMPACT I, 1995, IMPACT II, in progress reports, 1995, 1996).

The effects will depend on the type of gear, type of sediment, and the vulnerability of the species (signal species).

Rough estimates of penetration depth into the seabed, total area swept, and species killed (direct mortality above 15 %) are given in Table 25.a.

Question 25.b. If such information does not exist, what is being done in order to obtain such information?

The time and space components of gear and trawl impacts need to be critically evaluated. The respective impacts of the various types of gears need to be taken into account when balancing the need for resource harvesting.

The demonstrated physical and biological impacts reduce habitat complexity and diversity. Reductions in habitats, including losses of nursery areas, may lead to negative effects on recruitment. Impacts of fishing gear have to be understood, not simply in terms of removal of the targeted species but, more importantly, in terms of their impacts on ecosystem productivity and diversity.

Table 25.a. Damage to the seabed and its communities from the use of different types of towed gears.

Type of fishing gear	Evidence or indication of damage on seabed		Affected area	
	penetration depth	*species affected*	*area swept*	*location*
Otter trawl	• ground rope <5 cm (soft bottom) <2 cm (hard bottom) • trawl door 6–20 cm (soft bottom)	epifauna (e.g., Crustaceae: *Corystes* and *Eupagurus*; Molluscs: *Abra alba, Arctica islandica, Donax vitatis, Spisula subtrancata, Placopecten*; Echinoderms: juvenile *Echinocardium, Psammmmechinus miliaris*; Cnidaria: hydroids, *Alcyonium digitatum*)	• 98,956 km² per year (1989)	• whole of the North Sea where otter trawl is used

Beam trawl	• penetration depth
	    5–8 cm (soft bottom)
	    3–6 cm (hard bottom)
		same epibenthic fauna as mentioned under otter trawl, as well as the following infauna species: various Polychaetes: *Pectinaria* sp., *Aphrodite aculeata*; Sipunculids and Tunicates; Molluscs: *Tellinya ferrigunosa, Turritella communis, Chamelea gallina, Dosinia lupinus, Mactra corallina*	• 323,009 km² per year (in 1989)	• central and southern North Sea

Demersal pair trawl	• ground rope 5–8 cm	**same as for otter trawl**	• 107,549 km² per year (in 1989)	• whole North Sea
Twin trawl	**see otter trawl**	**see otter trawl**		• central and northern North Sea
Seines and ring nets	• zero	mimimal effects on benthos	• 245 km²	• whole North Sea
Pair seine	**see seines and ring nets**	**see seines and ring nets**		

Dredges	• mussel dredge
	    5–25 cm
	 • cockle dredge
	    5 cm
	 • scallop dredge
	    3–10 cm (> 15 cm)
		see beam trawl [Note: The effects of the multiple scallop dredges markedly increases the swept area.]		• estuarine and coastal areas of the North Sea

Shrimp beam trawl	• 4–5 cm	in addition to benthos killed in the trawl path, a high mortality of benthos and juvenile fish in small-mesh sized nets		• estuarine and coastal areas of the North Sea
Prawn trawl	**see shrimp beam trawl**			• northern North Sea

Industrial trawls	**see otter trawl**	epibenthic fauna (**see otter trawl**)	• industrial pair trawl
			    10,648 km²
			 • industrial single trawl
			    127,165 km²
				• industrial pair trawl central North Sea
				 • industrial single trawl whole North Sea

RECOMMENDATIONS

Recommendations to the ICES Council

The Working Group on Ecosystem Effects of Fishing Activities recommends that it meet from 17–21 March 1997 at the ICES Secretariat in Copenhagen, Denmark, to:

a)   review the data available with regard to establishing long-term trends in fishing impact by area and fishing gear;

b)   continue development on the underlying theory on the behaviour of size and diversity spectra of groundfish data in order to more confidently relate variation in these spectra to changes in fishing activities;

c)   pursue further analyses of species abundance and species biomass relations;

d)   review the data and hypotheses on community responses in closed areas;

e)   examine alternative measures and reference points which might be used for including ecosystem considerations when applying the precautionary approach;

f)   explore the quantitative relationship between the level of fishing effort and the impact on other parts of the ecosystem; and

g)   collaborate with the Multispecies Assessment Working Group (MAWG) to estimate the level of fish predation on the benthos (using in particular the 1981 and, if possible, the 1991 year of the stomach data), with a view to ultimately estimating the long-term variation in fish predation impact due to changes in the size of dominant fish populations.

Justification for Recommendations

Analyses conducted during this meeting highlighted the need for further efforts to reconstruct histories of fishing intensity for areas of interest. The group recognized that efforts to compile existing effort data have already been undertaken by ICES groups, but felt that additional work should be done. (a)

WGECO has directed considerable effort to examining various metrics of community structure and to determining the long-term temporal and spatial patterns of variation in these metrics when applied to a variety of groundfish databases. WGECO now believes that if these patterns of variation are to be fully interpreted, especially if the link between fishing effort and changes in fish assemblage structure are to be established, we need to develop the underlying theory as to how these metrics behave. (b) and (c)

The potential utility of closed areas as a tool for understanding the effects of fishing activity is high. (d)

Discussion of the precautionary approach and its applications continues. WGECO should contribute to the issue. (e) and (f)

WGECO has directed some effort to determining the direct impact of fishing activities on benthic communities in recent time. But WGECO is aware that the large changes in fish population size that have occurred since the 1960s will also have affected the benthos. (g)

Recommendations for other Working Groups

1) Efforts to assess the vulnerability of populations are hampered by insufficient quantitative information on the sensitivity of benthic species and on the life history parameters which determine population vulnerability.

   The WGECO **recommends** that the Benthos Ecology Working Group (BEWG) continue to review experimental data on benthic mortalities and arrive at estimates of life history attributes that might be applied to methods such as those outlined in Sections 5.3 and 5.4.

2) Our analyses of variation on the number of fish species at various abundance levels showed that the number of rare species being recorded in later years has increased. In order to determine whether this is a "real" result, or simply an artifact of the data, we need to determine the consistency of recording species.

   The WGECO **recommends** that the International Bottom Trawl Survey Working Group (IBTSWG) review the taxonomic recording of species to establish consistency of reporting, particularly with respect to procedures for sampling the catch and the taxonomic level of identification. It should also attempt to apply a consistent scheme retrospectively to the existing data sets.

Table 3.4.4.1. Slopes and intercepts of regressions of ln(numbers) *versus* ln(length) for North Sea fish. Data from the IBTS survey.

Year	R-Square	Intercept	St. error	Slope	St. error
1973	0.87	16.67	1.27	-4.87	0.56
1974	0.96	15.55	0.68	-4.65	0.30
1975	0.99	17.11	0.39	-5.23	0.17
1976	0.95	16.89	0.80	-5.29	0.36
1977	0.92	15.28	0.93	-4.50	0.41
1978	0.97	15.41	0.53	-4.50	0.23
1979	0.97	15.23	0.55 .	-4.96	0.24
1980	0.81	13.36	1.13	-3.38	0.50
1981	0.96	18.16	0.83	-5.98	0.37
1982	0.96	15.60	0.62	-4.52	0.27
1983	0.98	17.35	0.49	-5.33	0.22
1984	0.98	16.16	0.42	-4.79	0.19
1985	0.97	16.60	0.56	-4.92	0.25
1986	0.95	17.40	0.82	-5.26	0.36
1987	0.97	17.40	0.69	-5.37	0.30
1988	0.96	19.20	0.84	-6.20	0.37
1989	0.97	18.90	0.69	-6.07	0.30
1990	0.98	18.26	0.52	-5.92	0.23
1991	0.96	18.60	0.83	-5.94	0.37
1992	0.96	19.13	0.82	-6.19	0.36
1993	0.96	19.02	0.82	-6.08	0.36

Table 3.4.4.2. Slopes and intercepts of regressions of ln(numbers) *versus* ln(length) for North Sea fish. Data from the EGFS survey (5 cm length classes in the range 20-80 cm).

Year	R-Square	Intercept	St. error	Slope	St. error
1977	0.99	17.73	0.27	-4.68	0.56
1978	0.98	17.23	0.47	-4.47	0.30
1979	0.96	17.72	0.61	-4.67	0.17
1980	0.95	18.16	0.72	-4.76	0.36
1981	0.93	18.35	0.91	-4.85	0.41
1982	0.95	19.49	0.90	-5.46	0.23
1983	0.98	18.35	0.52	-5.00	0.24
1984	0.98	18.93	0.52	-5.08	0.50
1985	0.96	18.91	0.68	-5.29	0.37
1986	0.91	17.23	0.98	-4.43	0.27
1987	0.94	17.74	0.80	-4.77	0.22
1988	0.97	19.33	0.71	-5.71	0.19
1989	0.98	20.26	0.64	-6.07	0.25
1990	0.98	19.78	0.60	-5.85	0.36

Table 3.4.4.3. Goodness of fit indicator (r^2) for annual regressions on ln(numbers) on ln(length) when different ranges of length classes are used.

Survey	EGFS			IBTS		
Year	Range (cm)			Range (cm)		
	10–70	20–80	40–95	10–70	20–80	40–95
1973				0.90	0.87	0.71
1974				0.98	0.96	0.89
1975				0.96	0.99	0.96
1976				0.94	0.95	0.92
1977	0.92	0.99	0.91	0.95	0.92	0.82
1978	0.92	0.98	0.92	0.93	0.97	0.95
1979	0.93	0.96	0.91	0.97	0.97	0.96
1980	0.83	0.95	0.89	0.91	0.81	0.56
1981	0.91	0.93	0.92	0.91	0.96	0.93
1982	0.86	0.95	0.89	0.93	0.96	0.95
1983	0.93	0.987	0.89	0.93	0.98	0.97
1984	0.83	0.98	0.87	0.96	0.98	0.97
1985	0.83	0.96	0.84	0.93	0.97	0.95
1986	0.81	0.91	0.82	0.91	0.95	0.94
1987	0.75	0.94	0.85	0.94	0.96	0.92
1988	0.85	0.97	0.88	0.92	0.96	0.97
1989	0.89	0.98	0.87	0.94	0.97	0.95
1990	0.91	0.98	0.89	0.92	0.98	0.95
1991				0.91	0.96	0.89
1992				0.92	0.96	0.98
1993				0.91	0.96	0.95

Table 3.4.4.4. Slopes and intercepts of regressions of diversity *versus* length for North Sea fish. Data from the IBTS survey (5 cm length classes in the range 40–95 cm).

Year	R-Square	Intercept	St. error	Slope (x10^2)	St.err. (x10^2)
1973	0.28	1.44	0.44	-6.29	3.15
1974	0.39	1.95	0.21	-3.92	1.54
1975	0.01	1.01	0.35	0.59	2.53
1976	0.48	1.65	0.21	-4.59	1.50
1977	0.15	1.73	0.52	-5.02	3.76
1978	0.06	1.42	0.38	-2.25	2.73
1979	0.85	2.12	0.13	-6.66	0.90
1980	0.28	2.19	0.46	-6.62	3.33
1981	0.82	2.63	0.24	-11.56	1.70
1982	0.09	1.37	0.38	-2.64	2.72
1983	0.24	2.51	0.40	-5.13	2.88
1984	0.35	2.72	0.44	-7.24	3.14
1985	0.14	2.28	0.36	-3.27	2.60
1986	0.29	2.38	0.26	-3.82	1.90
1987	0.64	2.74	0.28	-8.39	1.97
1988	0.65	2.68	0.23	-7.21	1.68
1989	0.82	3.31	0.24	-11.54	1.73
1990	0.70	3.28	0.34	-11.87	2.44
1991	0.61	2.86	0.26	-7.38	1.86
1992	0.87	3.91	0.28	-16.37	2.01
1993	0.72	3.20	0.27	-9.89	1.95

Table 3.4.4.5. Slopes and intercepts of regressions of diversity *versus* length for North Sea fish. Data from the EGFS survey (5 cm length classes in the 40–95 cm).

Year	R-Square	Intercept	St. error	Slope (x10^2)	St.err. (x10^2)
1977	0.30	2.18	0.17	-2.52	3.15
1978	0.82	2.76	0.14	-6.61	1.54
1979	0.82	2.77	0.16	-7.78	2.53
1980	0.66	2.62	0.20	-6.23	1.50
1981	0.21	2.29	0.32	-3.79	3.76
1982	0.00	1.53	0.27	-0.30	2.73
1983	0.01	1.64	0.16	-0.28	0.90
1984	0.09	1.81	0.22	-1.54	3.33
1985	0.22	2.01	0.29	-3.58	1.70
1986	0.41	2.82	0.53	-10.06	2.72
1987	0.48	2.72	0.42	-9.06	2.88
1988	0.78	3.11	0.26	-11.23	3.14
1989	0.83	3.16	0.21	-10.57	2.60
1990	0.63	2.94	0.26	-7.79	1.95

Table 3.4.4.6. Goodness of fit indicator (r^2) for annual regressions of diversity on length class, when different ranges of length classes are used.

Survey	EGFS			IBTS		
Year	Range (cm)			Range (cm)		
	10–70	20–80	40–95	10–70	20–80	40–95
1973				0.27	0.41	0.28
1974				0.022	0.17	0.39
1975				0.089	0.10	0.014
1976				0.19	0.70	0.48
1977	0.056	0.061	0.35	0.23	0.25	0.15
1978	0.0	0.17	0.82	0.26	0.29	0.058
1979	0.075	0.38	0.81	0.050	0.44	0.85
1980	0.151	0.051	0.66	0.12	0.18	0.28
1981	0.171	0.15	0.21	0.26	0.56	0.82
1982	0.000	0.11	0.013	0.21	0.33	0.091
1983	0.0	0.043	0.015	0.47	0.039	0.24
1984	0.252	0.66	0.096	0.12	0.082	0.35
1985	0.13	0.17	0.22	0.57	0.058	0.14
1986	0.47	0.65	0.41	0.24	0.050	0.29
1987	0.34	0.36	0.48	0.051	0.42	0.64
1988	0.029	0.26	0.78	0.53	0.022	0.65
1989	0.046	0.072	0.83	0.059	0.037	0.82
1990	0.018	0.073	0.63	0.028	0.093	0.70
1991				0.011	0.010	0.61
1992				0.040	0.12	0.87
1993				0.17	0.009	0.72

Table 3.4.5.1.a. Individual Type I sum-of-squares tables for annual glm models of ln(length), Area, and ln(Area) fit to ln(numbers) for EGFS and IBTS data. The areas used are North Sea Task Force Areas. The ln(length)(Area) or length(Area) estimated separate slopes for each Area.

Source	DF	Type I SS	Mean square	F Value	Pr > F
1977					
Intercept	1	2618.97290	2618.97290	6198.25	0.0001
ln(length)	1	414.52728	414.52728	981.05	0.0001
Area	9	133.67193	14.85244	35.15	0.0001
ln(length)(Area)	9	25.46301	2.82922	6.70	0.0001
1978					
Intercept	1	2722.64219	2722.64219	7451.43	0.0001
ln(length)	1	419.12298	419.12298	1147.07	0.0001
Area	9	93.09078	10.34342	28.31	0.0001
ln(length)(Area)	9	22.00315	2.44479	6.69	0.0001
1979					
Intercept	1	2534.28097	2534.28097	5659.24	0.0001
ln(length)	1	493.36705	493.36705	1101.72	0.0001
Area	9	158.82811	17.64757	39.41	0.0001
ln(length)(Area)	9	18.01812	2.00201	4.47	0.0001
1980					
Intercept	1	2960.78293	2960.78293	5693.01	0.0001
ln(length)	1	506.37148	506.37148	973.65	0.0001
Area	9	133.31494	14.81277	28.48	0.0001
ln(length)(Area)	9	13.00226	1.44470	2.78	0.0058
1981					
Intercept	1	2701.64767	2701.64767	4559.27	0.0001
ln(length)	1	430.59297	430.59297	726.66	0.0001
Area	9	197.49221	21.94358	37.03	0.0001
ln(length)(Area)	9	25.96492	2.88499	4.87	0.0001
1982					
Intercept	1	2525.98437	2525.98437	3167.59	0.0001
ln(length)	1	575.24804	575.24804	721.36	0.0001
Area	8	84.38218	10.54777	13.23	0.0001
ln(length)(Area)	8	19.30866	2.41358	3.03	0.0046
1983					
Intercept	1	2374.70045	2374.70045	4233.68	0.0001
ln(length)	1	464.31273	464.31273	827.79	0.0001
Area	9	205.58206	22.84245	40.72	0.0001
ln(length)(Area)	9	31.83719	3.53747	6.31	0.0001
1984					
Intercept	1	2831.38172	2831.38172	3261.60	0.0001
ln(length)	1	618.36281	618.36281	712.32	0.0001
Area	9	91.58634	10.17626	11.72	0.0001
ln(length)(Area)	9	59.87661	6.65296	7.66	0.0001
1985					
Intercept	1	2408.02899	2408.02899	3193.75	0.0001
ln(length)	1	567.12988	567.12988	752.18	0.0001
Area	9	92.10474	10.23386	13.57	0.0001
ln(length)(Area)	9	33.95501	3.77278	5.00	0.0001

Table 3.4.5.1.a. Continued.

Source	DF	Type I SS	Mean square	F Value	Pr > F
1986					
Intercept	1	2389.78031	2389.78031	2638.97	0.0001
ln(length)	1	553.92304	553.92304	611.68	0.0001
Area	9	125.52596	13.94733	15.40	0.0001
ln(length)(Area)	9	30.72783	3.41420	3.77	0.0004
1987					
Intercept	1	2442.49739	2442.49739	3944.60	0.0001
ln(length)	1	569.97262	569.97262	920.50	0.0001
Area	9	92.65160	10.29462	16.63	0.0001
ln(length)(Area)	9	29.69277	3.29920	5.33	0.0001
1988					
Intercept	1	1977.98121	1977.98121	3387.17	0.0001
ln(length)	1	583.16258	583.16258	998.63	0.0001
Area	9	133.52926	14.83658	25.41	0.0001
ln(length)(Area)	9	33.35785	3.70643	6.35	0.0001
1989					
Intercept	1	2247.42850	2247.42850	3260.92	0.0001
ln(length)	1	725.28638	725.28638	1052.36	0.0001
Area	9	90.29474	10.03275	14.56	0.0001
ln(length)(Area)	9	17.83323	1.98147	2.88	0.0046
1990					
Intercept	1	2135.20438	2135.20438	4172.90	0.0001
ln(length)	1	697.67141	697.67141	1363.48	0.0001
Area	9	83.64305	9.29367	18.16	0.0001
ln(length)(Area)	9	43.02494	4.78055	9.34	0.0001

Table 3.4.5.1.b. Individual Type I sum-of-squares tables for annual glm models of length, area, and length(Area) fit to ln(numbers) for EGFS and IBTS data. The areas used are North Sea Task Force Areas. The ln(length)(Area) or length(Area) estimated separate slopes for each area.

Source	DF	Type I SS	Mean square	F Value	Pr>F
1977					
Intercept	1	191.869194	191.869194	2027.90	0.0001
ln(length)	1	1.938419	1.938419	20.49	0.0001
Area	9	8.113715	0.901524	9.53	0.0001
ln(length)(Area)	10	6.979933	0.697993	7.38	0.0001
1978					
Intercept	1	204.692017	204.692017	2274.50	0.0001
ln(length)	1	5.769369	5.769369	64.11	0.0001
Area	9	10.982588	1.220288	13.56	0.0001
ln(length)(Area)	10	9.758428	0.975843	10.84	0.0001
1979					
Intercept	1	184.294559	184.294559	1679.18	0.0001
ln(length)	1	10.815397	10.815397	98.54	0.0001
Area	9	5.597280	0.621920	5.67	0.0001
ln(length)(Area)	10	7.718033	0.771803	7.03	0.0001
1980					
Intercept	1	183.955105	183.955105	1640.84	0.0001
ln(length)	1	4.456922	4.456922	39.75	0.0001
Area	9	5.875433	0.652826	5.82	0.0001
ln(length)(Area)	10	12.418766	1.241877	11.08	0.0001
1981					
Intercept	1	166.888095	166.888095	1399.53	0.0001
ln(length)	1	2.596106	2.596106	21.77	0.0001
Area	9	6.704015	0.744891	6.25	0.0001
ln(length)(Area)	10	9.149293	0.914929	7.67	0.0001
1982					
Intercept	1	140.604278	140.604278	1549.64	0.0001
ln(length)	1	5.296307	5.296307	58.37	0.0001
Area	8	4.951309	0.618914	6.82	0.0001
ln(length)(Area)	9	3.847388	0.427488	4.71	0.0001
1983					
Intercept	1	142.304382	142.304382	1144.41	0.0001
ln(length)	1	3.957928	3.957928	31.83	0.0001
Area	9	7.384948	0.820550	6.60	0.0001
ln(length)(Area)	10	5.713328	0.571333	4.59	0.0001
1984					
Intercept	1	167.464397	167.464397	1861.12	0.0001
ln(length)	1	5.039318	5.039318	56.00	0.0001
Area	9	4.529463	0.503274	5.59	0.0001
ln(length)(Area)	10	7.633275	0.763328	8.48	0.0001
1985					
Intercept	1	143.666271	143.666271	1192.60	0.0001
ln(length)	1	6.896730	6.896730	57.25	0.0001
Area	9	4.568940	0.507660	4.21	0.0001
ln(length)(Area)	10	8.297860	0.829786	6.89	0.0001

Table 3.4.5.1.b. Continued.

Source	DF	Type I SS	Mean square	F Value	Pr>F
1986					
Intercept	1	171.201965	171.201965	1512.29	0.0001
ln(length)	1	12.028054	12.028054	106.25	0.0001
Area	9	12.134819	1.348313	11.91	0.0001
ln(length)(Area)	10	8.288125	0.828813	7.32	0.0001
1987					
Intercept	1	143.282080	143.282080	1047.31	0.0001
ln(length)	1	17.819902	17.819902	130.25	0.0001
Area	9	5.679841	0.631093	4.61	0.0001
ln(length)(Area)	10	6.526509	0.652651	4.77	0.0001
1988					
Intercept	1	149.650725	149.650725	936.52	0.0001
ln(length)	1	14.411917	14.411917	90.19	0.0001
Area	9	8.446437	0.938493	5.87	0.0001
ln(length)(Area)	10	4.113051	0.411305	2.57	0.0081
1989					
Intercept	1	173.551545	173.551545	1123.23	0.0001
ln(length)	1	9.882472	9.882472	63.96	0.0001
Area	9	8.104915	0.900546	5.83	0.0001
ln(length)(Area)	10	4.853967	0.485397	3.14	0.0016
1990					
Intercept	1	180.285989	180.285989	1480.54	0.0001
ln(length)	1	14.711976	14.711976	120.82	0.0001
Area	9	9.622569	1.069174	8.78	0.0001
ln(length)(Area)	10	5.553398	0.555340	4.56	0.0001

Table 3.4.7.1. Counts of number of species in each category of abundance, by year, from EGFS data. The total number of possible species was counted over the entire time series, and the zero category is that total less the number in all non-zero categories for that year. Data are presented as separate counts for fish > 40 cm, fish < 40 cm and all fish.

					Number per haul					
Year	0	<1	<2	<4	<8	<16	<32	<64	<128	>128
					Fish < 40 cm					
1977	15	5	6	10	6	12	4	2	6	5
1978	12	7	8	11	4	10	2	7	2	7
1979	15	5	7	5	9	8	5	6	3	8
1980	20	10	2	5	6	6	9	1	5	7
1981	21	7	6	3	9	5	6	1	4	9
1982	25	8	4	7	4	8	2	1	4	8
1983	15	15	5	6	5	5	4	3	5	8
1984	16	13	5	5	5	4	4	2	4	8
1985	20	15	7	4	4	5	4	2	3	7
1986	18	12	7	5	5	7	4	5	1	6
1987	22	11	7	5	8	4	3	4	3	4
1988	16	14	7	4	6	3	4	3	2	7
1989	11	15	10	8	5	4	5	3	5	5
1990	6	23	10	5	3	4	7	4	4	5
					Fish > 40 cm					
1977	8	6	12	7	5	3	0	1	0	0
1978	3	5	15	9	4	5	0	1	0	0
1979	8	11	6	9	5	2	1	0	0	0
1980	7	7	9	7	5	5	2	0	0	0
1981	9	7	6	12	2	5	1	0	0	0
1982	14	9	7	6	3	1	0	1	1	0
1983	10	9	10	7	1	4	1	0	0	0
1984	9	15	4	6	4	1	0	1	0	0
1985	12	13	5	5	5	1	0	1	0	0
1986	8	16	7	6	3	0	2	0	0	0
1987	12	14	8	1	5	1	1	0	0	0
1988	13	12	7	4	4	1	0	0	0	0
1989	9	17	8	5	3	0	0	0	0	0
1990	5	16	13	3	2	3	0	0	0	0
										19
					All Fish					
1977	17	4	8	13	8	11	6	2	7	5
1978	13	7	11	11	7	12	3	6	3	7
1979	17	8	5	8	11	6	8	6	3	8
1980	17	7	5	9	7	7	11	1	5	7
1981	24	5	5	9	9	7	8	1	4	9
1982	28	7	8	9	3	9	2	1	5	8
1983	2	14	8	6	6	7	5	3	5	8
1984	17	17	8	10	4	6	4	2	4	8
1985	17	13	10	5	7	5	5	2	3	7
1986	18	16	9	6	6	7	6	5	1	6
1987	22	16	9	5	6	7	4	4	3	4
1988	15	16	8	7	6	4	5	3	2	7
1989	15	16	13	9	7	3	6	3	5	5
1990	8	21	17	6	4	5	7	4	4	5

Table 3.4.7.2. Counts of number of species in each category of abundance, by year, from IBTS data. The total number of possible species was counted over the entire time series, and the zero category is that total less the number in all non-zero categories for that year. Data are presented as separate counts for fish > 40 cm, fish < 40 cm, and all fish.

Year	Number per Haul								
	0	<1	<2	<4	<8	<16	<32	<64	<128	>128
Fish>40 cm
1973	61	23	2	5	4	5	2	4	2	7
1974	51	20	16	7	7	7	1	6	0	6
1975	49	17	12	9	4	4	6	3	5	6
1976	47	21	6	13	8	7	5	0	2	8
1977	42	20	15	12	8	5	4	1	2	5
1978	37	19	16	15	9	5	5	1	1	7
1979	52	26	11	7	6	6	1	0	1	5
1980	44	23	7	16	4	6	6	2	1	6
1981	43	29	7	15	2	5	3	4	1	6
1982	43	22	12	11	10	1	6	2	2	6
1983	26	22	11	19	12	8	6	3	0	8
1984	23	23	6	19	12	13	6	3	4	6
1985	26	26	8	9	15	8	7	6	1	7
1986	21	24	5	22	11	8	8	4	4	8
1987	33	17	8	11	8	12	7	6	5	8
1988	28	24	7	19	6	12	8	2	1	8
1989	22	22	9	18	9	9	8	3	6	9
1990	26	20	7	14	16	9	5	7	2	9
1991	11	21	11	19	22	10	6	2	4	7
1992	22	17	5	16	19	11	7	6	5	7
1993	24	19	9	17	9	17	6	4	2	8
Fish < 40 cm
1973	29	4	4	2	1	0	1	2	0	0
1974	17	12	5	3	1	1	4	0	0	0
1975	22	6	5	3	5	0	1	0	0	1
1976	20	10	5	4	1	1	2	0	0	0
1977	12	15	6	4	3	2	0	0	1	0
1978	19	12	6	1	2	1	0	2	0	0
1979	19	11	7	1	4	1	0	0	0	0
1980	11	12	5	8	3	0	3	0	1	0
1981	20	9	6	4	2	1	1	0	0	0
1982	14	12	7	5	0	2	1	2	0	0
1983	7	13	3	11	5	1	2	1	0	0
1984	3	17	6	7	6	3	0	1	0	0
1985	1	16	7	11	5	1	1	0	1	0
1986	8	13	8	6	2	4	2	0	0	0
1987	8	14	11	1	6	1	2	0	0	0
1988	7	18	4	4	6	4	0	0	0	0
1989	8	13	7	5	4	4	2	0	0	0
1990	7	15	8	5	5	2	1	0	0	0
1991	3	18	7	6	4	2	3	0	0	0
1992	5	16	8	7	6	0	1	0	0	0
1993	10	13	6	10	1	2	2	0	0	0

Table 3.4.7.2. Continued.

Year	0	<1	<2	<4	<8	<16	<32	<64	<128	>128
					All fish					
1973	78	27	6	7	5	5	3	6	2	7
1974	56	32	15	10	8	8	5	6	0	6
1975	59	23	17	12	9	4	7	3	5	7
1976	53	31	11	17	9	8	7	0	2	8
1977	42	35	21	16	11	7	4	1	3	6
1978	44	31	22	16	11	6	5	3	1	7
1979	59	37	18	8	10	7	1	0	1	5
1980	43	35	12	24	7	6	9	2	2	6
1981	51	38	13	19	4	6	4	4	1	6
1982	45	34	19	16	10	3	7	4	2	6
1983	21	35	14	30	17	9	8	4	0	8
1984	14	40	12	26	18	16	6	4	4	6
1985	15	42	15	20	20	9	8	6	2	9
1986	17	37	13	28	13	12	10	4	4	8
1987	29	31	19	12	14	13	9	6	5	8
1988	23	42	11	23	12	16	8	2	1	8
1989	18	35	16	23	13	13	10	3	6	9
1990	21	35	15	19	21	11	6	7	2	9
1991	2	39	18	25	26	12	9	2	4	9
1992	15	33	13	23	25	11	8	6	5	7
1993	21	32	15	27	10	19	8	4	2	8

Table 3.4.7.3. Maximum likelihood analysis of variance table for log linear model fit to tabulation of species by abundance group.

Term		df	Chi squared	P
EGFS				
Year		13	15.15	0.972
Group	4		22.65	0.0
Year x Group		52	77.50	0.0125
IBTS				
Year		20	14.63	0.797
Group	4		745.0	0.0
Year x Group		80	303.7	0.0

Table 3.4.7.4. Maximum likelihood analysis of variance table for log linear model fit to tabulation of species by abundance group, and size (>40 cm or <40 cm).

Term	df	Chi squared	P
EGFS			
Year	13	2.47	0.999
Size	1	16.41	0.001
Year x Size	13	1.04	0.999
Group	4	223.06	0.0
Year x Group	52	98.4	0.001
Size x Group	4	36.9	0.001
Residual likelihood	37	25.19	0.929
IBTS			
Year		13.65	0.873
Size		114.19	0.0
Year x Size		1.19	0.999
Group		342.26	0.0
Year x Group		273.75	0.0
Size x Group		51.58	0.0
Residual likelihood		71.93	0.139

Table 3.4.7.5.a. Counts of number of species in each category of biomass, by year, from EGFS data. The total number of possible species was counted over the entire time series, and the zero category is that total less the number in all non-zero categories for that year.

					Kg. per haul					
Year	0	< 0.125	< 0.25	<0.5	<1	<2	<4	<8	<16	<32
1978	13	11	0	2	11	10	6	11	6	10
1979	17	5	6	5	4	8	13	6	7	9
1980	20	6	2	6	3	10	6	11	5	11
1981	22	8	2	2	4	7	10	8	7	10
1982	28	6	1	4	7	6	6	9	5	8
1983	18	9	4	5	7	6	13	3	8	7
1984	17	10	5	3	8	5	9	10	6	7
1985	22	13	1	6	8	4	7	5	8	6
1986	18	9	6	6	6	5	8	10	7	5
1987	22	13	3	4	7	6	7	9	4	5
1988	19	14	6	3	8	5	5	10	5	5
1989	13	13	4	7	7	10	9	8	5	4
1990	7	17	4	7	9	12	6	8	5	5

Table 3.4.7.5.b. Counts of number of species in each category of biomass, by year, from IBTS data. The total number of possible species was counted over the entire time series, and the zero category is that total less the number in all non-zero categories for that year.

					Kg per haul					
Year	0	< 0.125	< 0.25	<0.5	<1	<2	<4	<8	<16	<32
1973	72	9	4	5	4	7	4	3	1	8
1974	55	15	2	2	11	9	8	5	0	10
1975	54	14	3	6	6	7	6	8	4	9
1976	53	17	3	5	7	9	6	7	2	8
1977	44	20	4	8	11	10	3	5	4	8
1978	42	23	8	7	8	10	4	5	3	7
1979	56	12	5	9	11	3	3	4	2	5
1980	45	19	1	11	10	6	7	4	1	13
1981	56	18	8	5	4	8	5	3	3	7
1982	48	20	5	10	6	7	8	3	2	8
1983	28	26	8	7	11	6	5	8	8	10
1984	25	22	7	7	11	12	11	9	3	10
1985	25	21	7	8	14	10	6	10	7	9
1986	24	25	10	6	12	10	8	6	5	11
1987	33	19	6	7	10	11	7	10	5	9
1988	31	16	8	14	11	9	7	5	5	11
1989	27	19	11	8	10	11	7	8	4	12
1990	26	26	5	10	8	8	10	11	7	6
1991	12	30	8	16	13	8	9	5	3	13
1992	21	20	10	9	9	16	10	10	4	8
1993	29	14	11	9	11	13	9	7	6	8

Table 3.4.7.6. Maximum likelihood analysis of variance table for log-linear model fit to tabulation of species by biomass class.

Term	df	Chi squared	P
EGFS			
Year	13	4.18	0.981
Group	4	145.61	0.0
Year x Group	52	60.12	0.205
IBTS			
Year	20	18.14	0.578
Group	4	231.23	0.0
Year x Group	80	214.19	0.0

Figure 3.2.2.1. Number and diversity spectra for various levels of fishing intensity. Predictions from non-interactive model with constant recruitment.

Number spectra
Recruitment constant

Diversity spectra
Recruitment constant
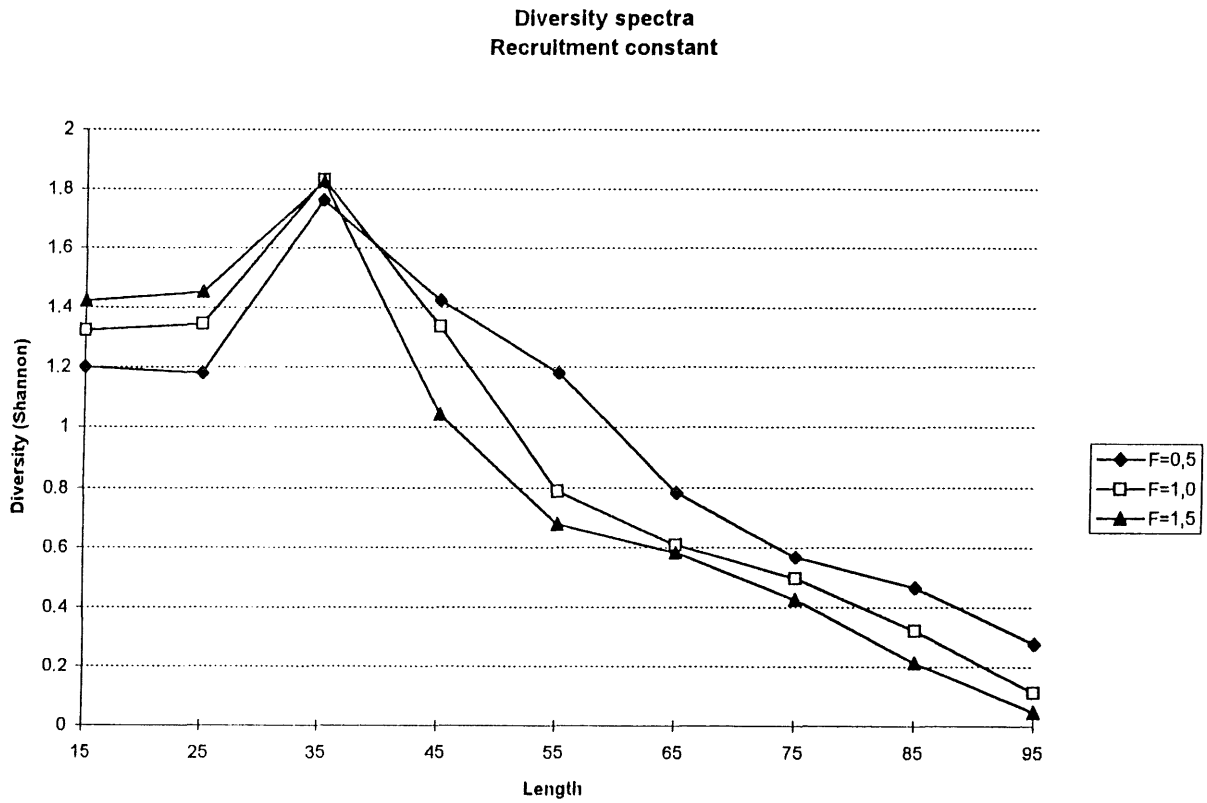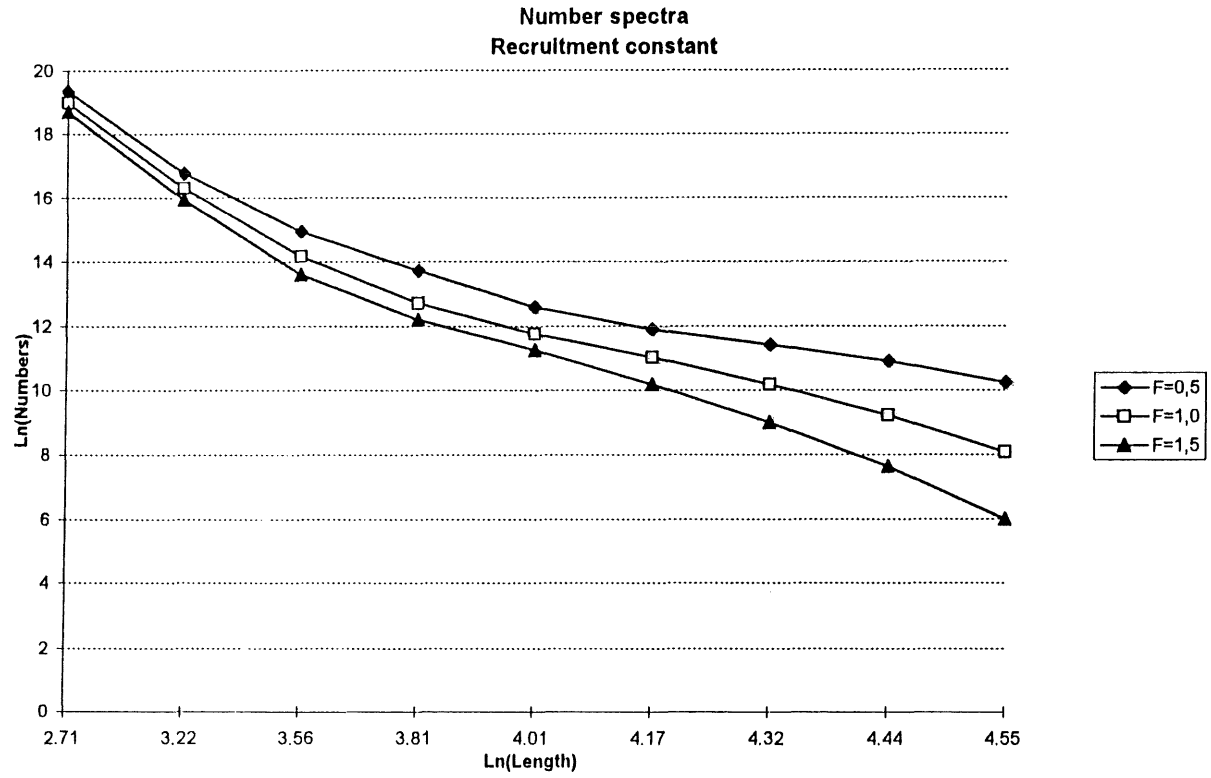
Figure 3.2.2.2. Slopes and intercepts of number and diversity spectra as a function of fishing intensity. Predictions from non-interactive model with constant recruitment.

Slope of number spectrum versus F

y = -1.6535x - 4.0546
R² = 0.9997

Slope of diversity spectrum versus F

y = 0.0003x - 0.0246
R² = 0.0128

Intercept of number spectrum versus F

y = 4.5243x + 30.16
R² = 0.9967

Intercept of diversity spectrum versus F

y = -0.2202x + 2.8064
R² = 0.7397
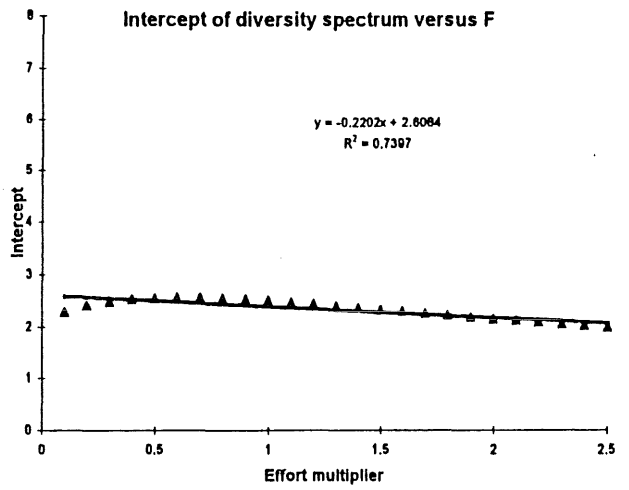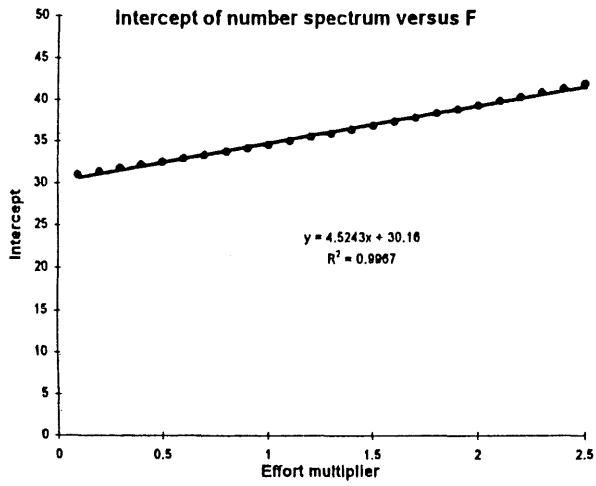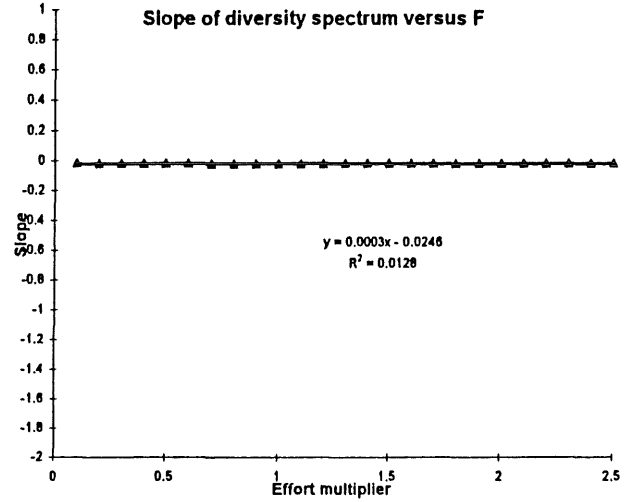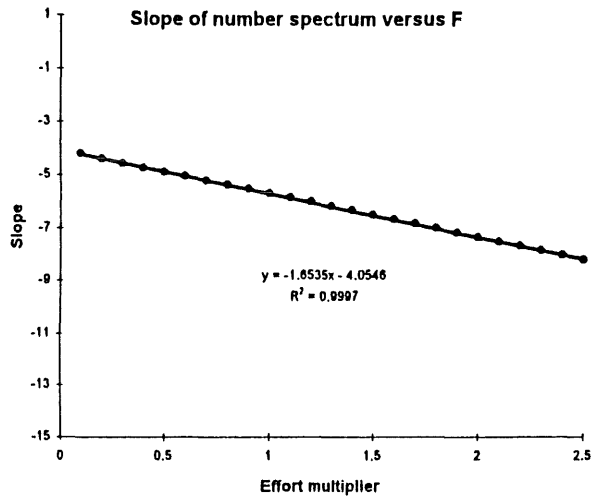
Figure 3.3.1.1. Map of the area used for the analysis in Sections 3.3 and 3.5. Area 1 = East Shetland; Area 2 = Northwest Central; Area 3 = Scottish East Coast.
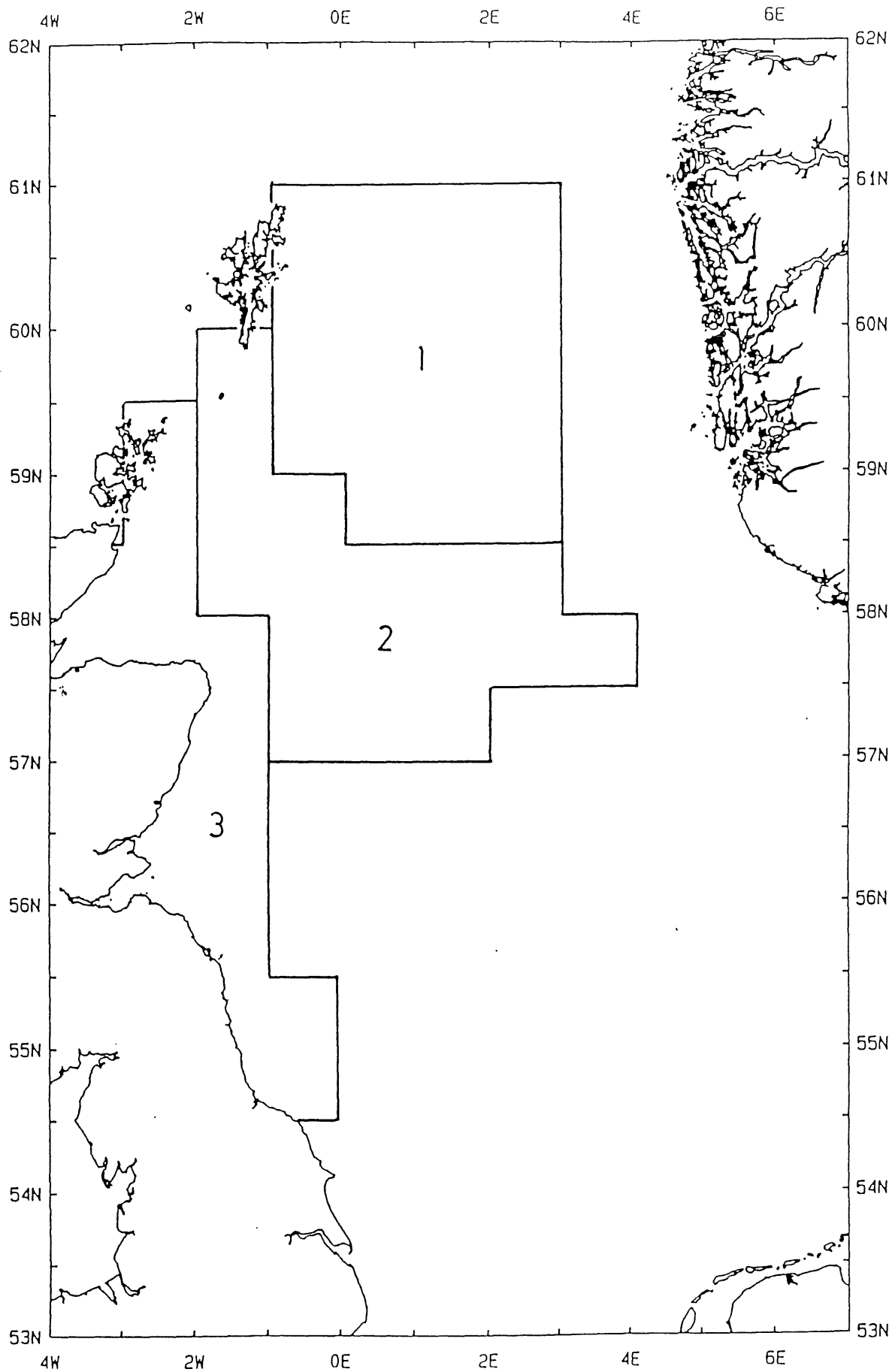
Figure 3.3.3.1. K-dominance curves for Scottish Survey data from the three areas shown in Figure 3.3.1.1. Curves are cumulative frequency of percentage of species as a function of species rank (order of abundance on a log 10 scale).
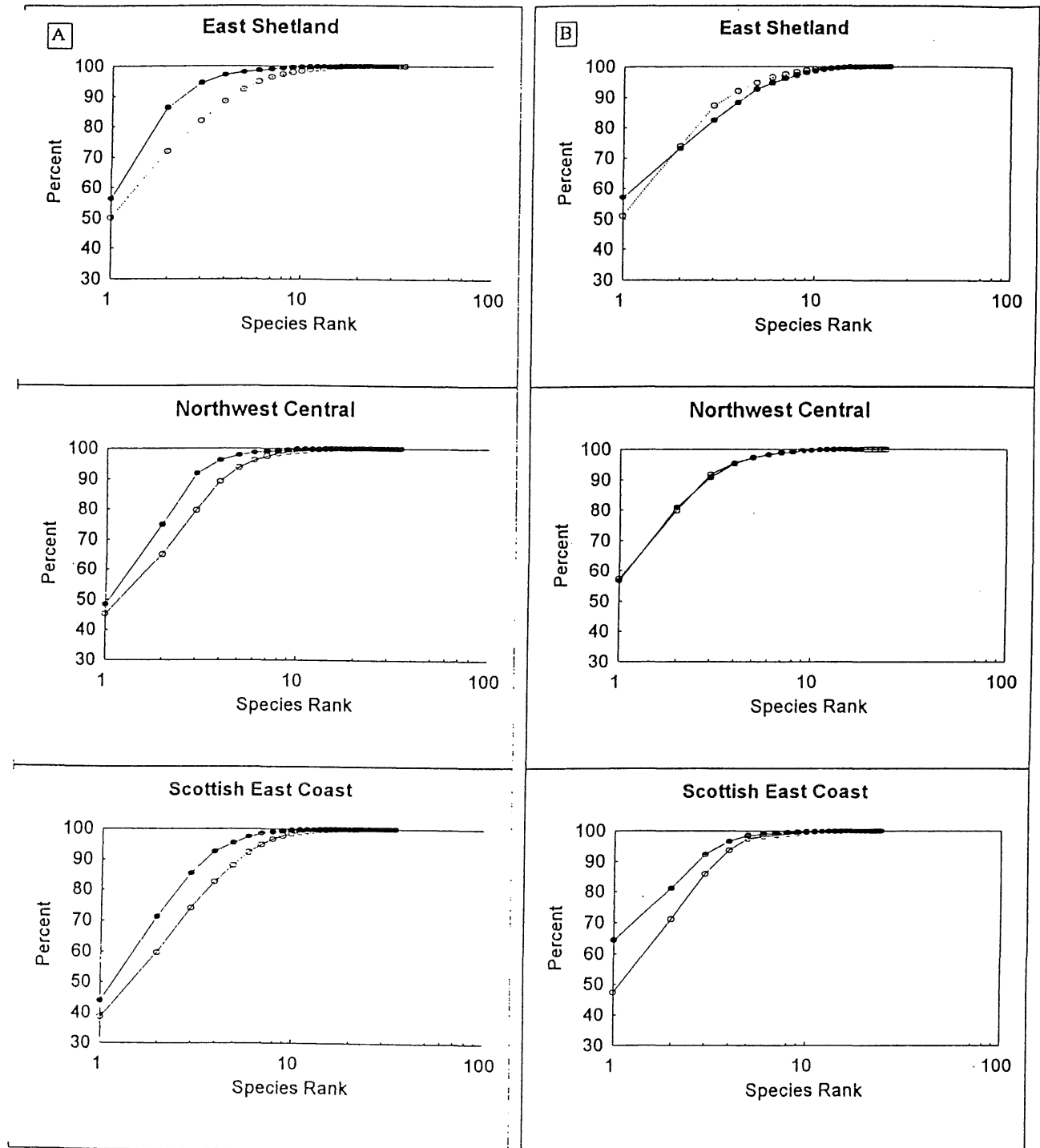
Figure **3.3.3.2.** Scatterplots of results from MDS ordinations of target (upper) and non-target (lower) species occurrences in Scottish survey data. Solid symbols are used for the period 1980–1993; open symbols are for 1929–1953. Squares = Area 1; Circles = Area 2; Diamonds = Area 3 in Figure 3.3.1.1.
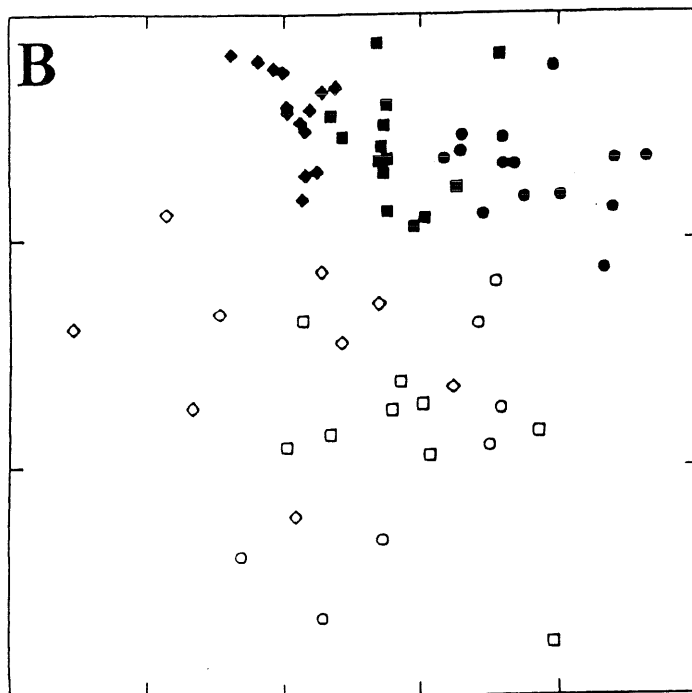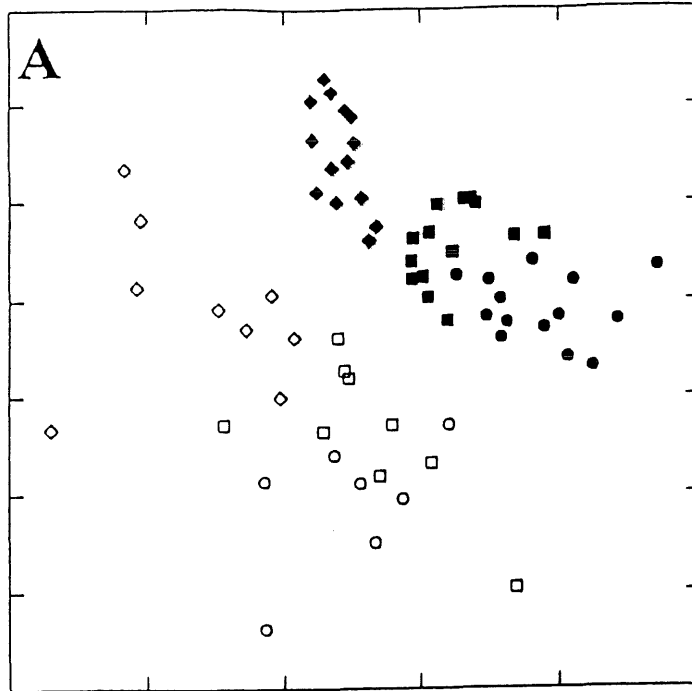
Figure 3.3.3.3.a. Pie chart of percentage occurrence of species for the whole groundfish assemblage from Scottish Survey data. Archive data set refers to the period from 1929–1953; AGFS data set refers to the period from 1980–1993. GGU= grey gurnard; SPU = spurdog; CDA = common dab; HAD = haddock; LRD = long rough dab; NPO = Norway pout; WHI = whiting.

All groundfish species
Archive data-set

OTHERS (7.86%)
GGU (3.29%)
SPU (1.59%)
CDA (5.61%)

LRD (9.34%)

NPO (18.45%)

HAD (37.01%)

WHI (16.84%)

All groundfish species
AGFS data-set

OTHERS (3.19%)
GGU (0.32%)
SPU (0.05%)
CDA (5.75%)
LRD (2.17%)

NPO (35.30%)

HAD (30.59%)

WHI (22.63%)
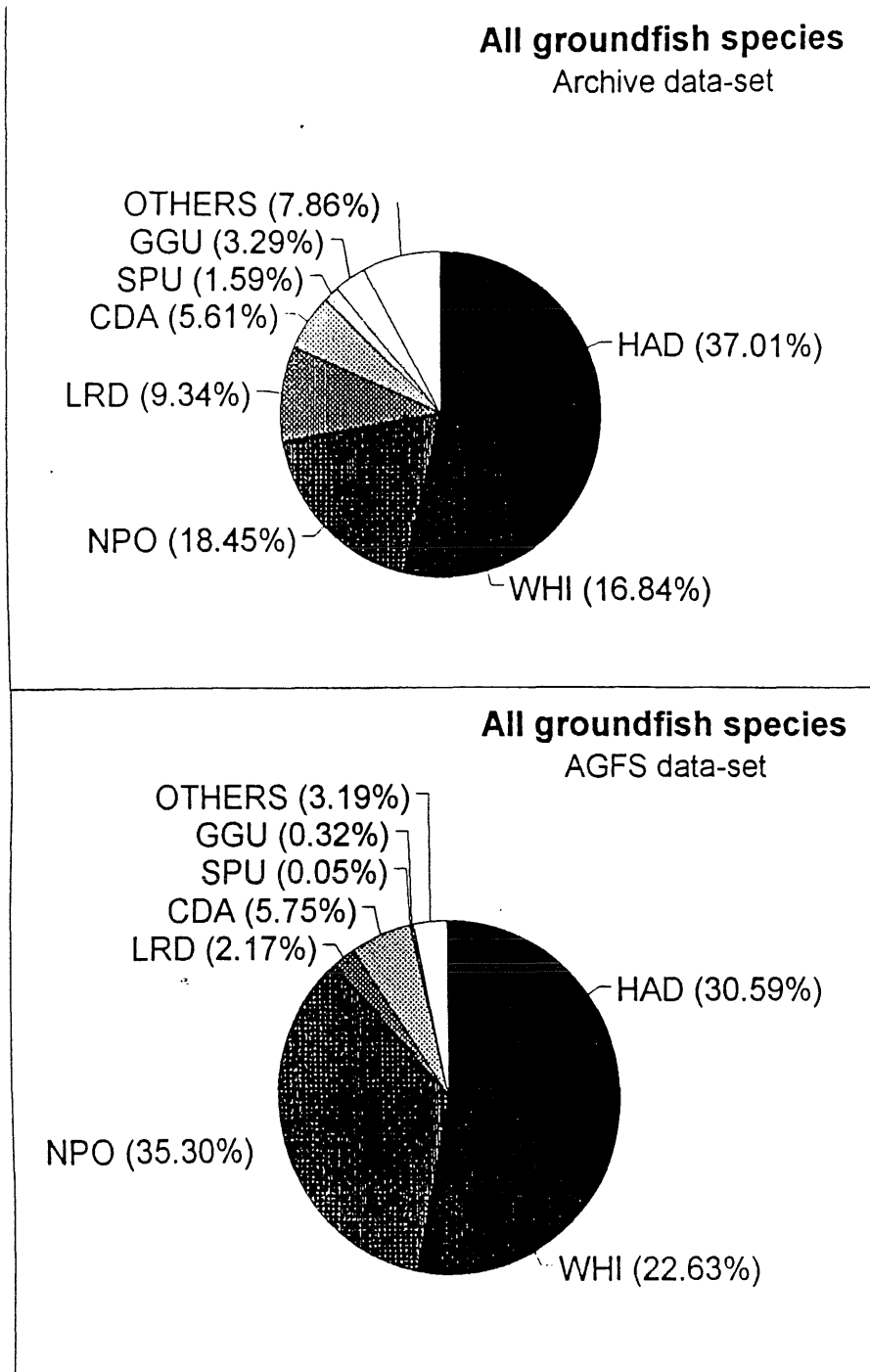
Figure 3.3.3.3.b. Pie chart of percentage occurrence of species for the non-target fraction from Scottish Survey data. Archive data set refers to the period from 1929–1953; AGFS data set refers to the period from 1980–1993. NHA= Norway haddock; PCO = poor cod; LSO = lemon sole; GGU = grey gurnard; LRD = long rough dab; CDA = common dab.

Non-target groundfish species
Archive data-set

OTHERS (4.59%)
NHA (0.45%)
PCO (6.71%)
LSO (5.18%)
GGU (15.66%)
LRD (44.65%)
CDA (22.77%)

Non-target groundfish species
AGFS data-set

OTHERS (3.83%)
NHA (2.40%)
PCO (2.81%)
LSO (10.55%)
GGU (3.72%)
LRD (32.88%)
CDA (43.80%)
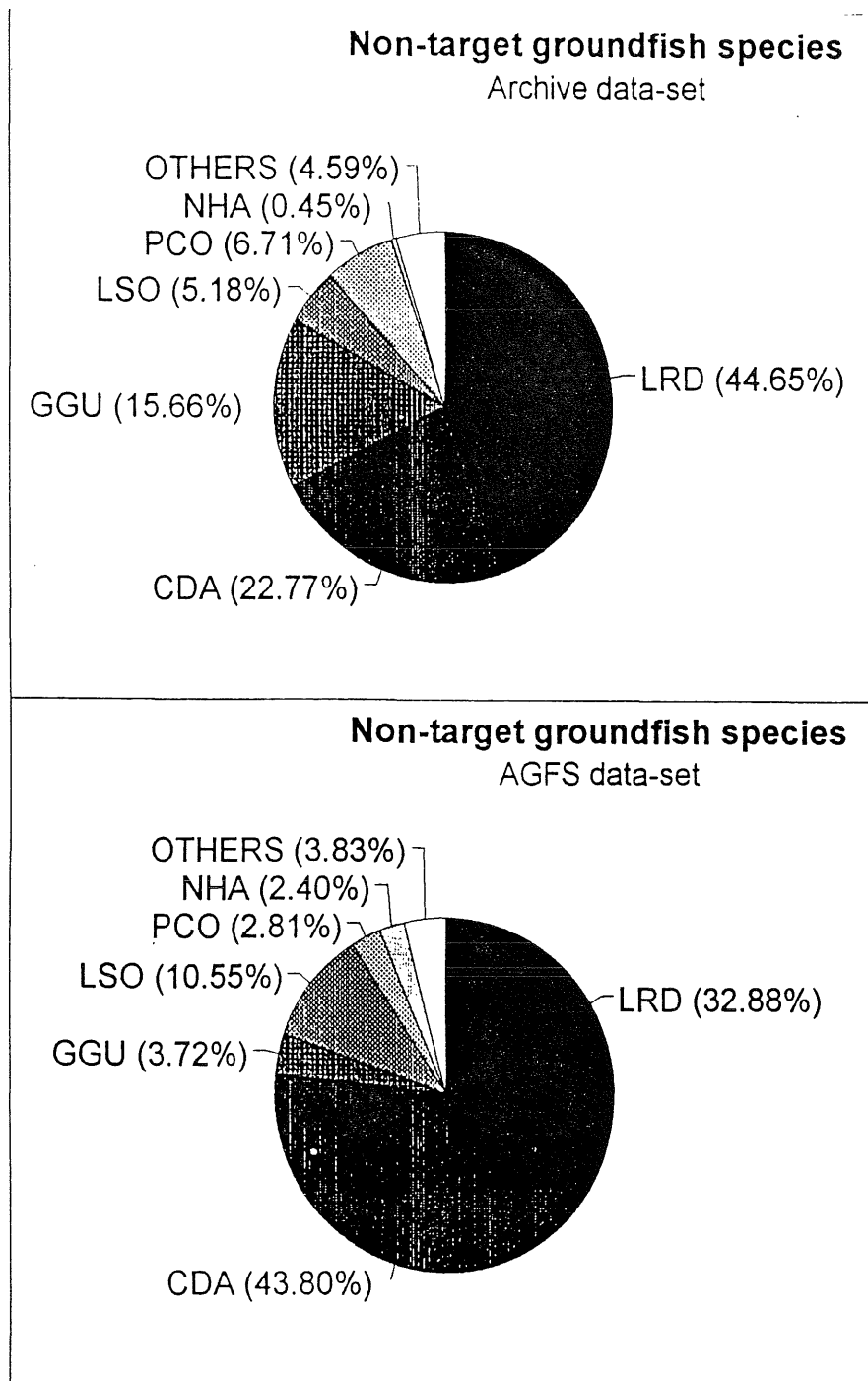
Figure 3.4.6.1. Patterns of slopes (upper) and intercepts (lower) over years from annual regressions on ln (numbers) on ln (length).

EGFS
Slope of size spectrum versus time

$y = -0.0873x + 168.15$
$R^2 = 0.4901$

EGFS
Intercept of size spectrum versus time

$y = 0.1424x - 263.84$
$R^2 = 0.3867$
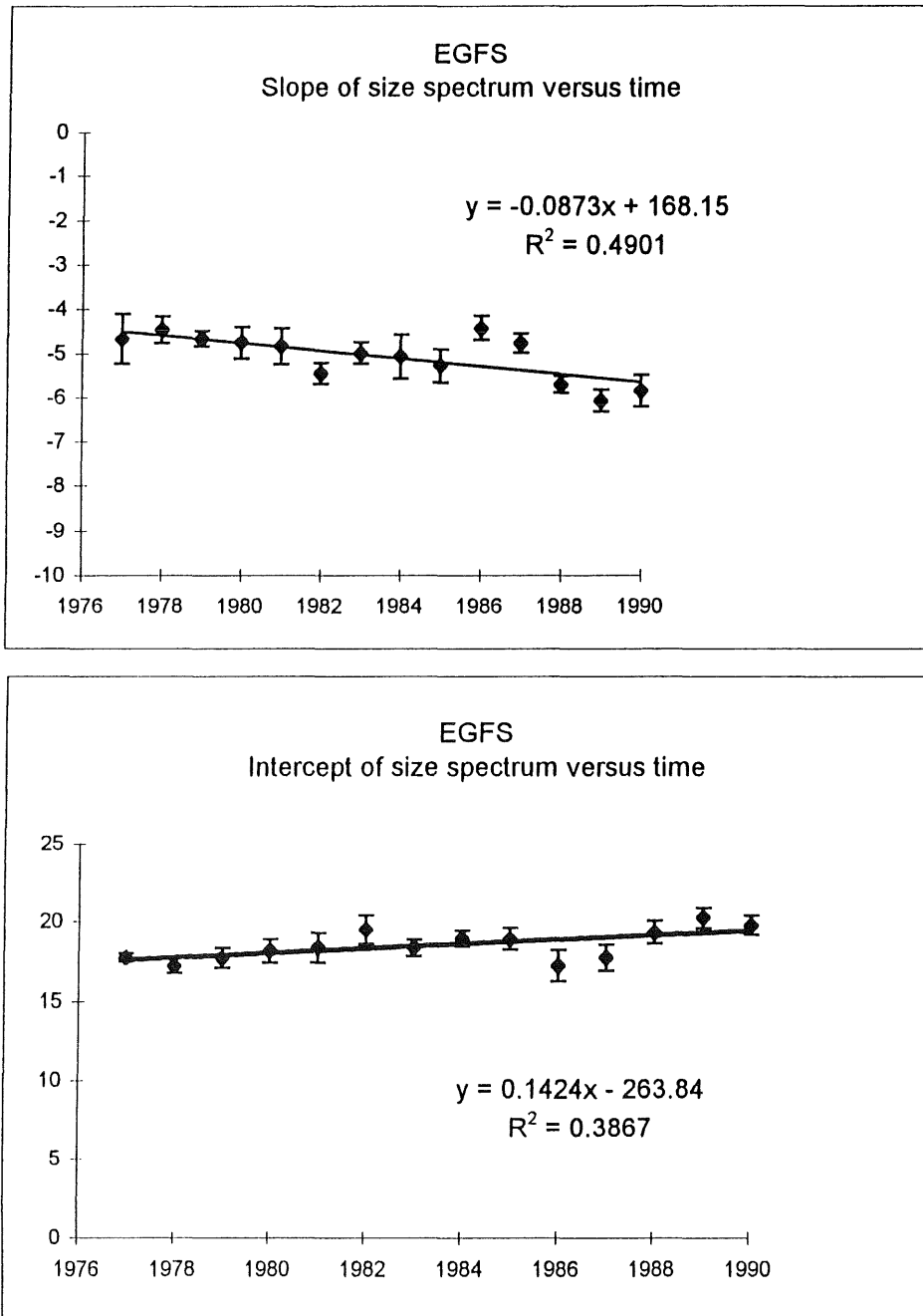
Figure 3.4.6.2. Patterns of slopes (upper) and intercepts (lower) over years from annual regressions on ln (numbers) on ln (length).

IBTS
Slope of size spectrum versus time

$y = -0.0771x + 147.74$
$R^2 = 0.4287$

IBTS
Intercept of size spectrum versus time

$y = 0.1768x - 333.59$
$R^2 = 0.4821$
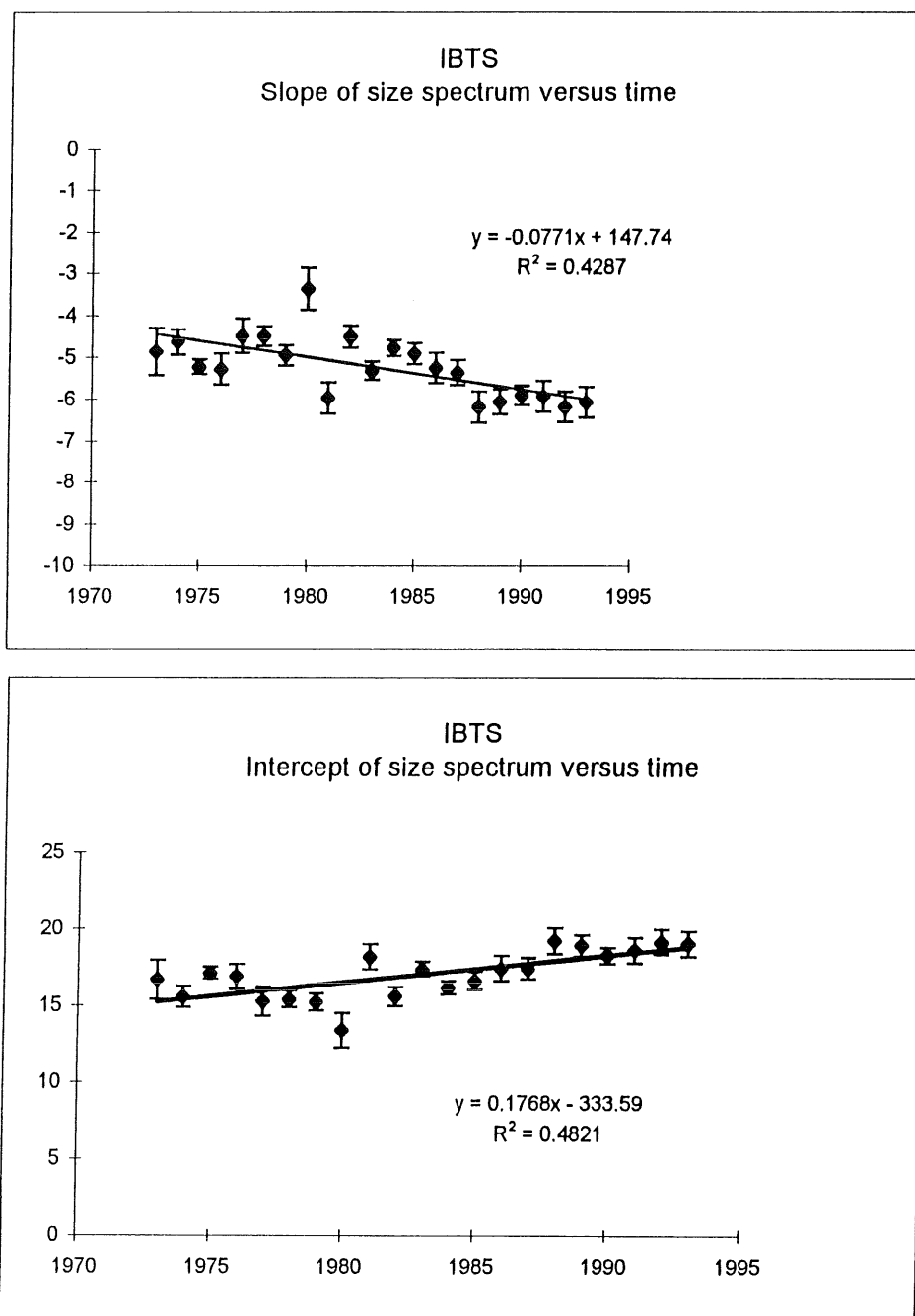
Figure 3.4.6.3. Contour plot of ln (numbers) as functions of length and year. Data from EGFS.

English Groundfish Survey
Ln(numbers) by year and length
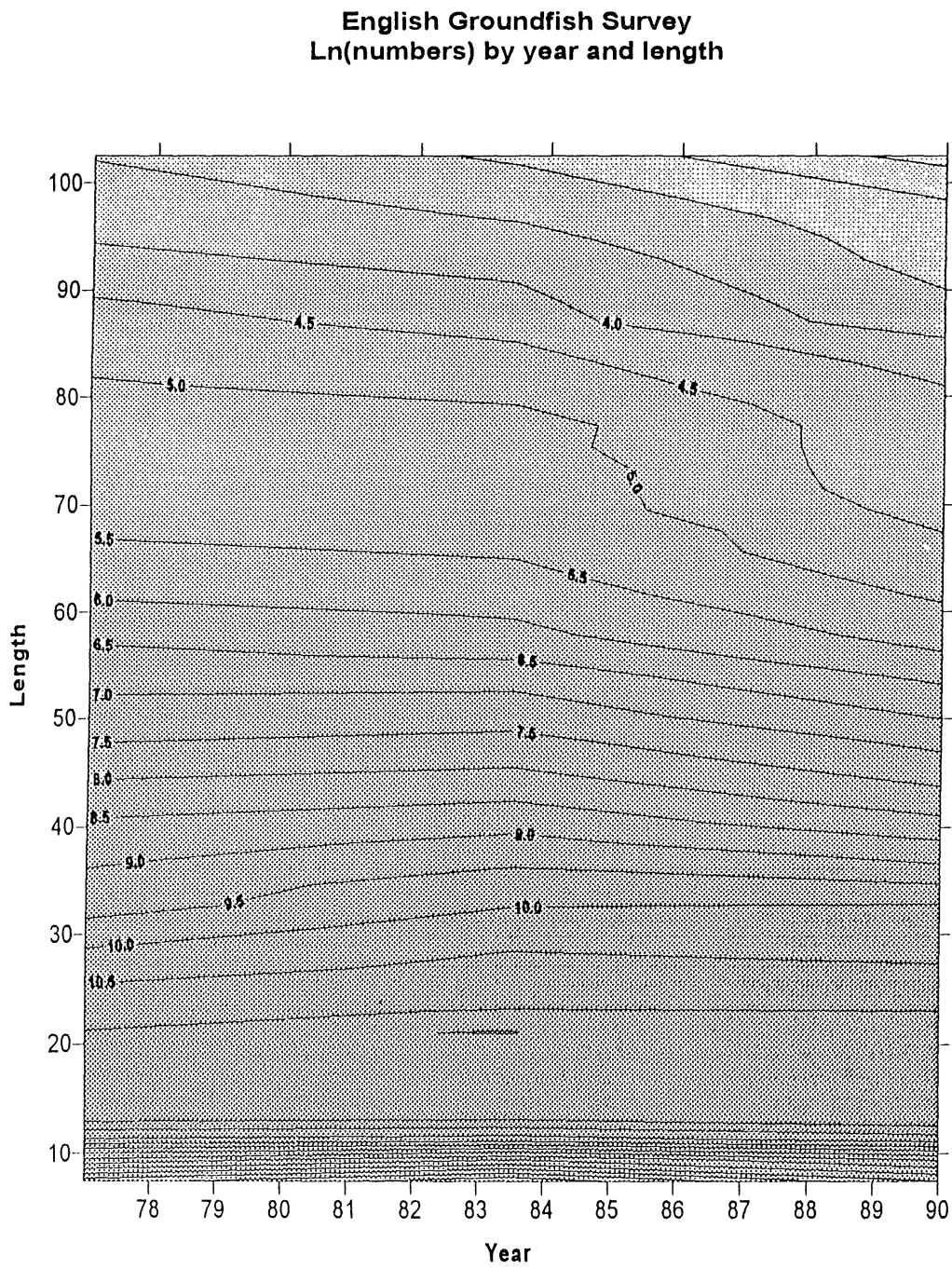
Figure 3.4.6.4. Contour plot of ln (numbers) as functions of length and year. Data from IBTS.

Int. Bottom Trawl Survey
Ln(numbers) by year and length
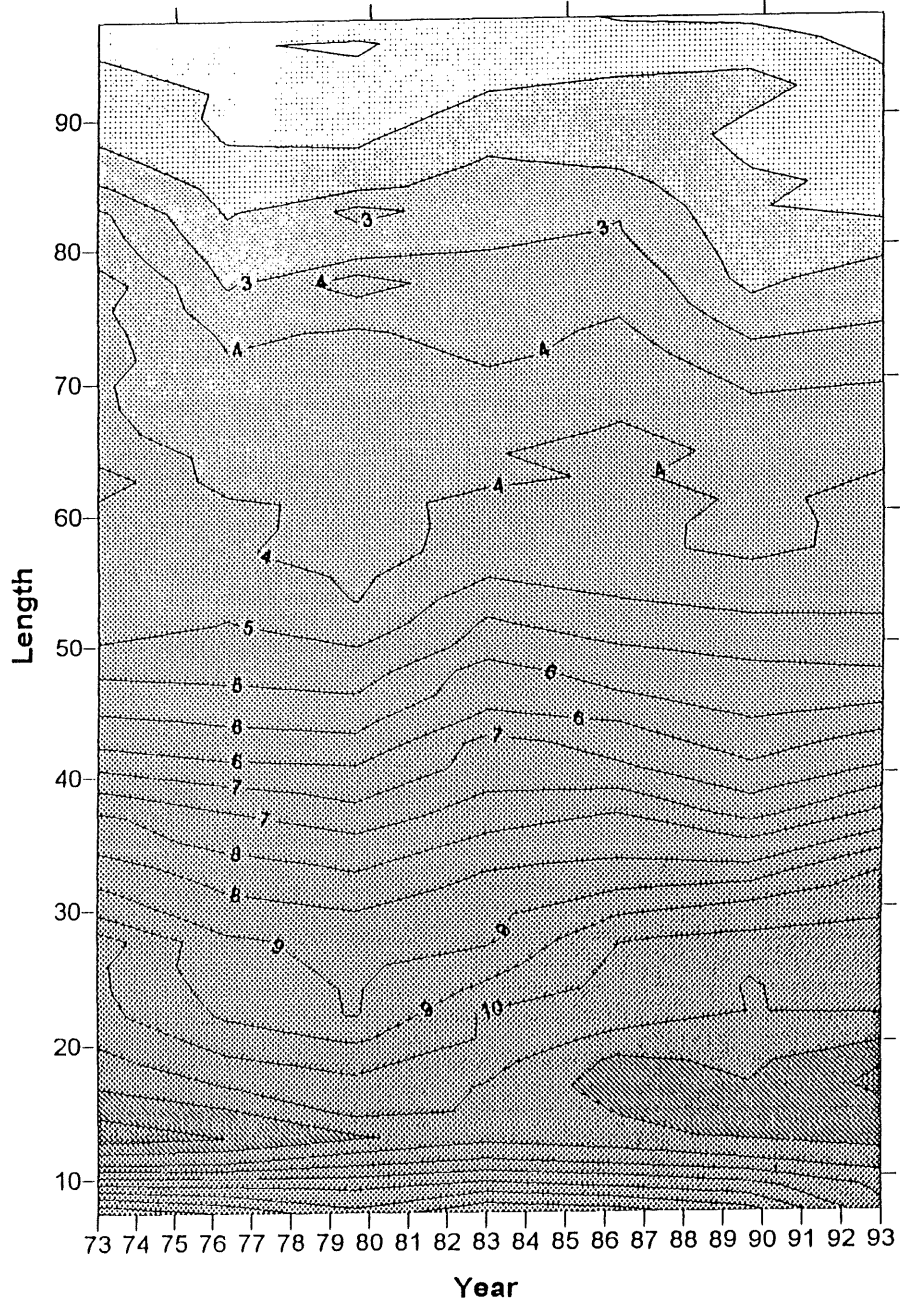
Figure 3.4.6.5. Patterns of slopes (upper) and intercepts (lower) over years from annual regressions of diversity on length class. Data from EGFS.

EGFS
Slope (+/- std. err.) of diversity spectrum

$y = -0.4514x + 889.51$
$R^2 = 0.246$

EGFS
Intercept (+/- std. err) of diversity spectrum

$y = 0.049x - 94.759$
$R^2 = 0.143$
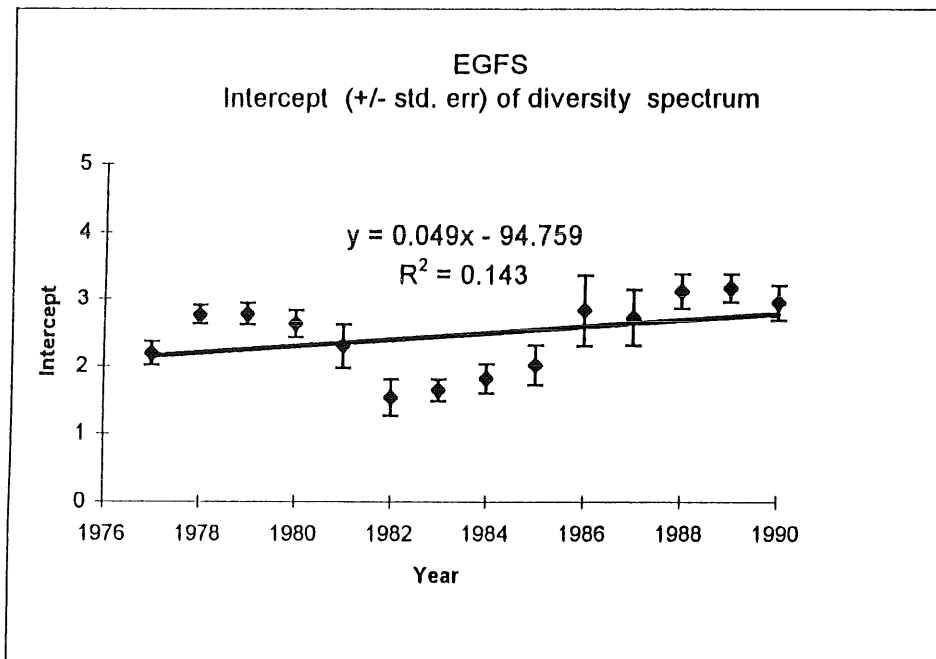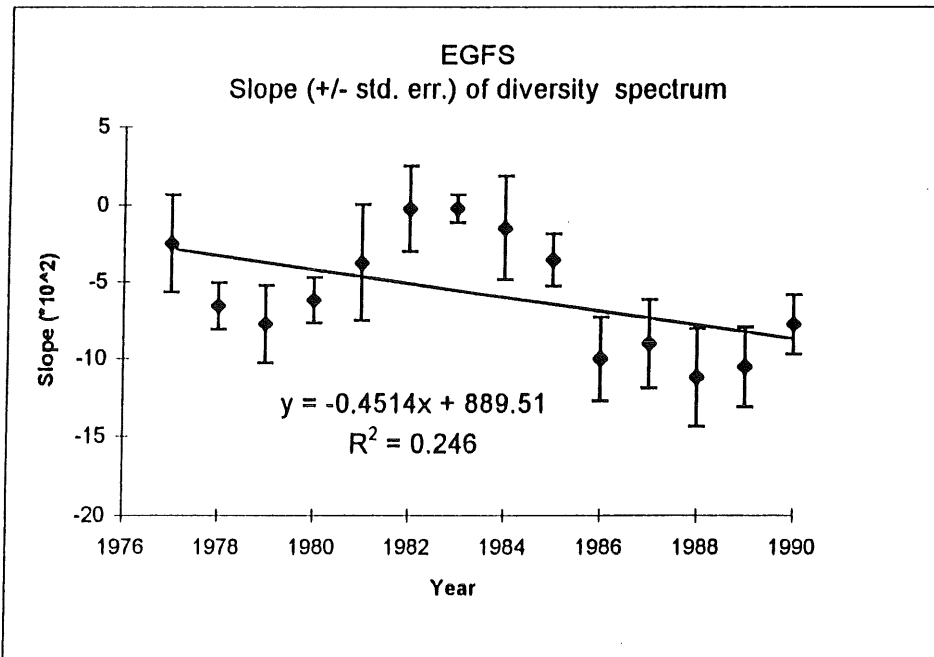
Figure 3.4.6.6. Patterns of slopes (upper) and intercepts (lower) over years from annual regressions of diversity on length class. Data from IBTS.

IBTS
Slope (+/- std. err.) of diversity spectrum

$y = -0.4068x + 799.95$
$R^2 = 0.4131$

IBTS
Intercept (+/- std. err) of diversity spectrum

$y = 0.1051x - 206.07$
$R^2 = 0.7546$
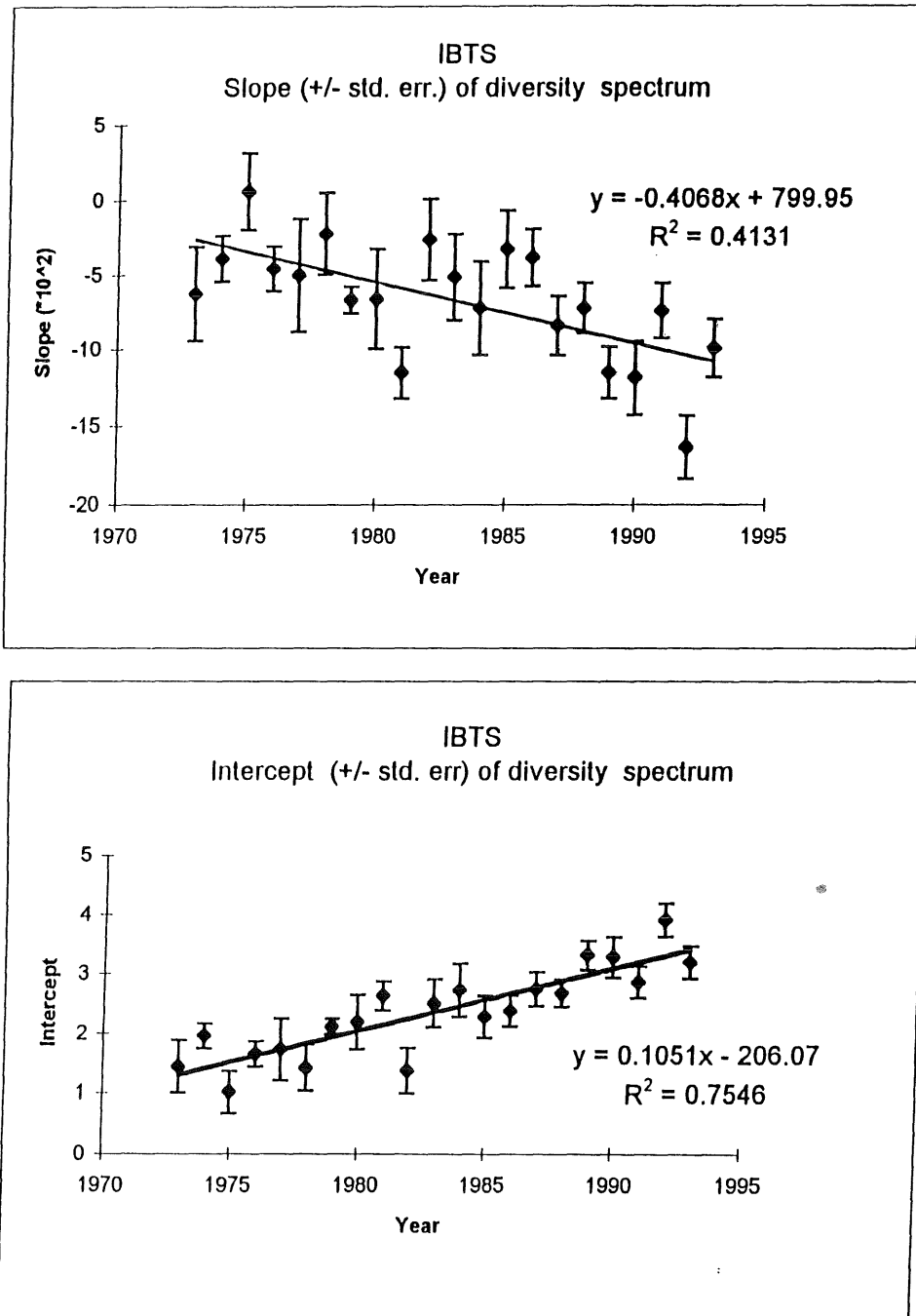
Figure 3.4.6.7. Contour plot of diversity as a function of length and year. Data from EGFS.

English Groundfish Survey
Diversity by year and length
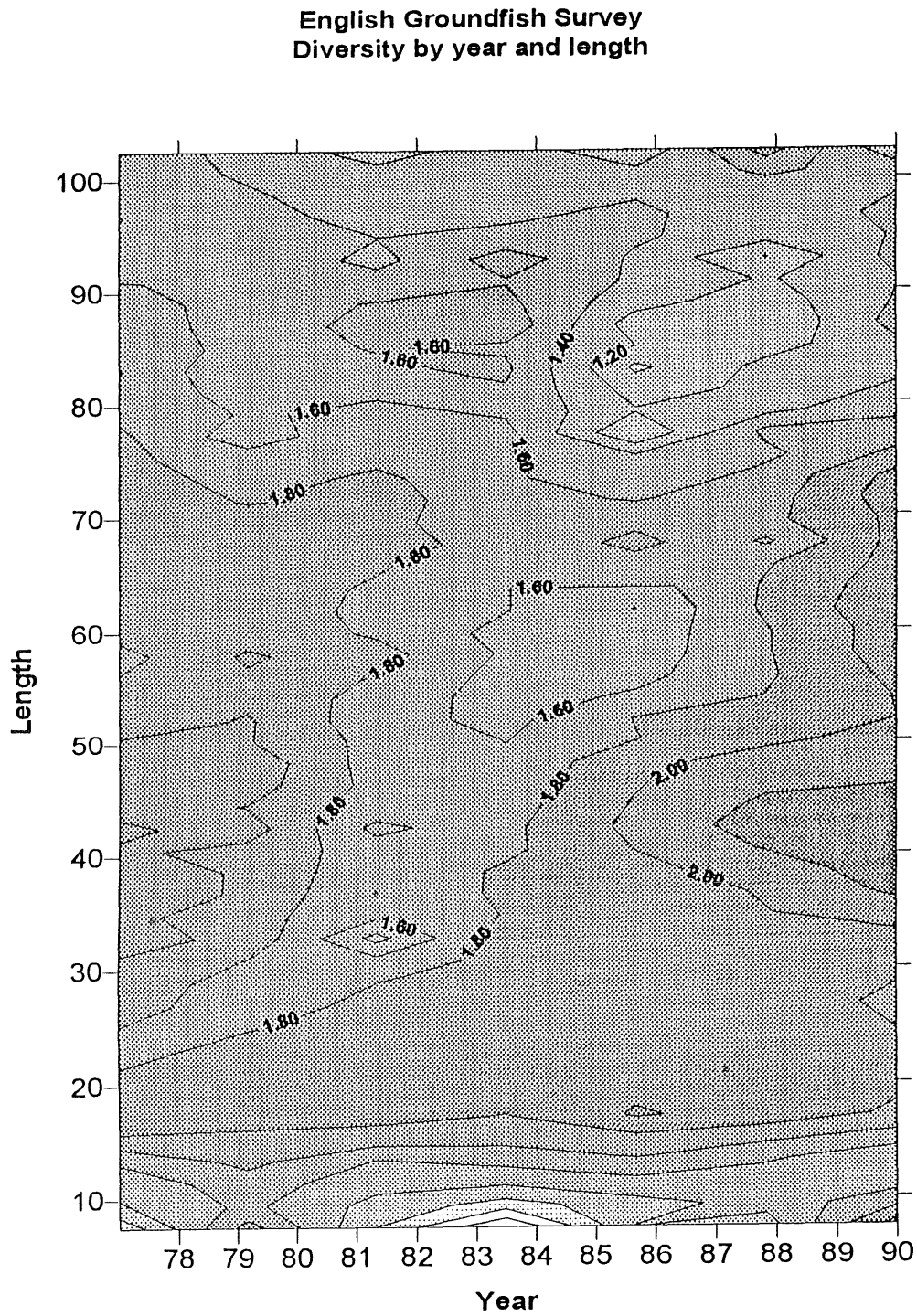
Figure 3.4.6.8. Contour plot of diversity as a function of length and year. Data from IBTS.

Int. Bottom Trawl Survey
Diversity by year and length
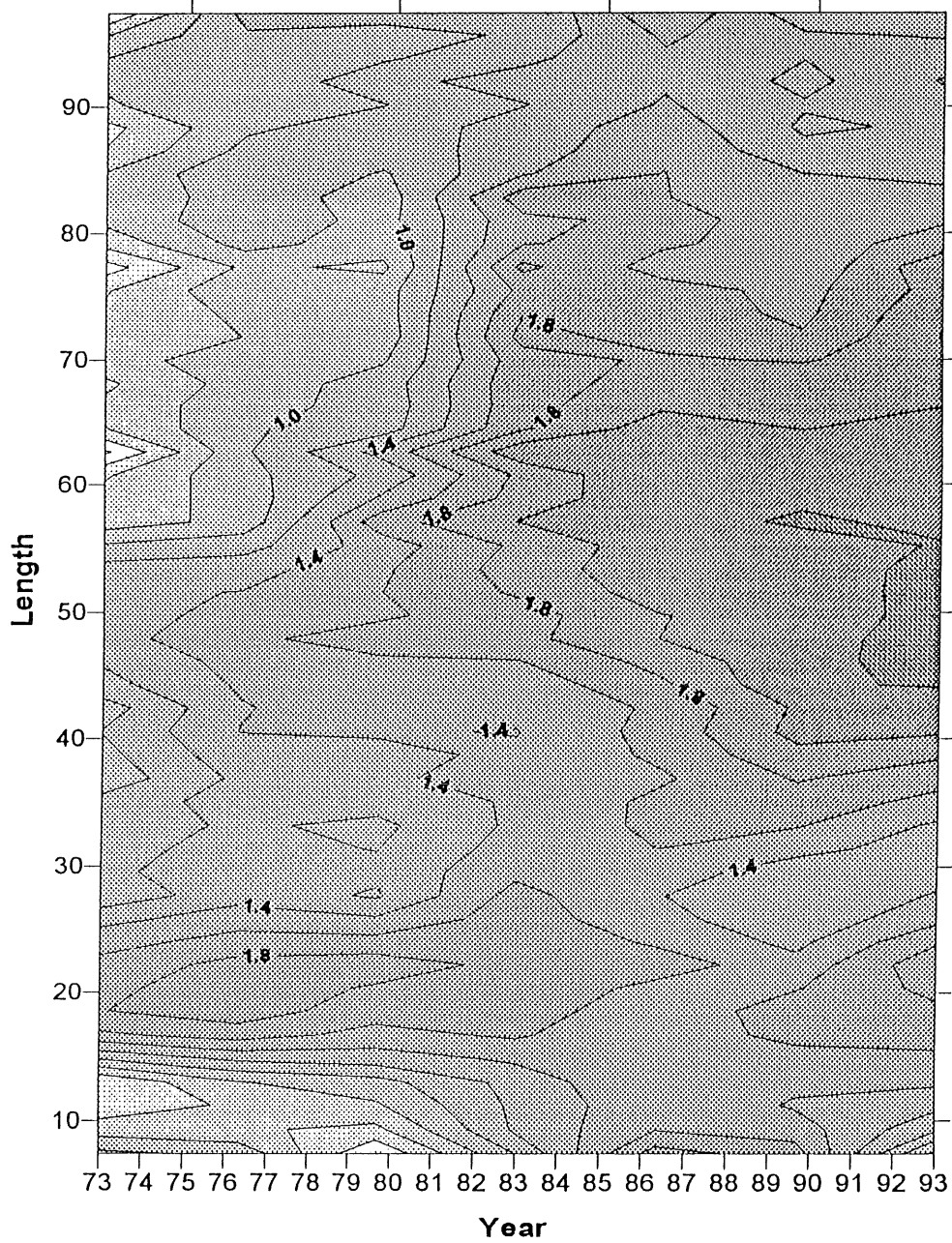
Figure 3.4.7.1. Plots of number of species (y-axis) *versus* year (x-axis) for each of ten categories of abundance for the IBTS (upper) and EGFS (lower) data. For each set of 10 panels the lowest abundance category is in the upper left (representing the number of species each year with mean abundance of zero in a given year in all tows), the next category of abundance (number of species each year with a mean abundance <1.0 individuals per haul) is second from left on top, etc., to the largest category (number of species each year with mean abundance <2256 individuals per haul) in the lower right panel. Each panel has a lowess smooth fit through annual counts of species per category of abundance. [Note: Axis labelling for each panel has been removed to reduce clutter. The x-axis is identical for all panels, the range on the y-axis varies between panels. Data from which these figures were constructed are given in Table 3.4.7.1.]

IBTS Abundance
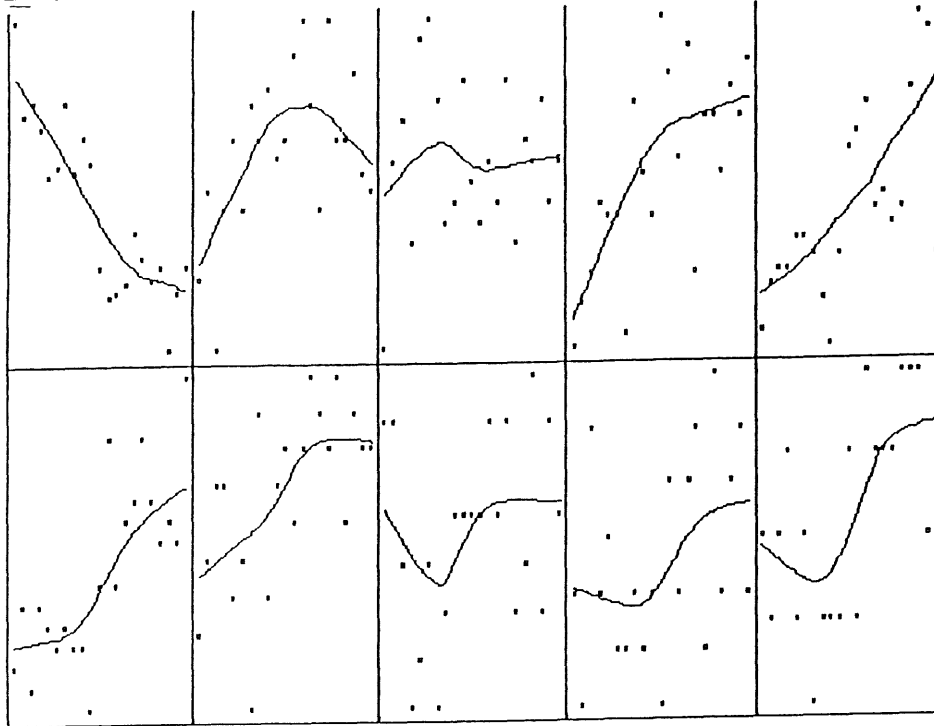
EGFS Abundance
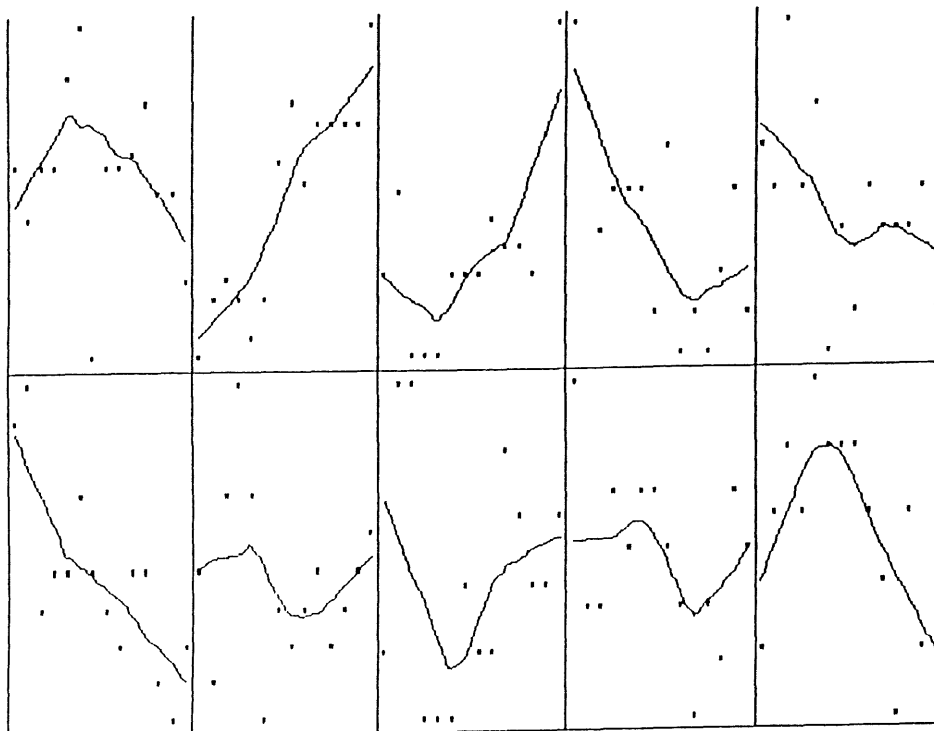
Figure 3.4.7.2. Plots of number of species (y-axis) *versus* year (x-axis) for each of ten categories of biomass for the IBTS (upper) and EGFS (lower) data. Panels for biomass category run from the lowest biomass (mean of zero per haul) category in the upper left to the highest biomass category (mean <32 kg per haul) in the lower right panel. Each panel has a lowess smooth fit through annual counts of species per biomass category. [Note: Axis labelling for each panel has been removed to reduce clutter. The x-axis is identical for all panels, the range on the y-axis varies between panels. Data from which these figures were constructed are given in Table 3.4.7.2.]

IBTS Biomass
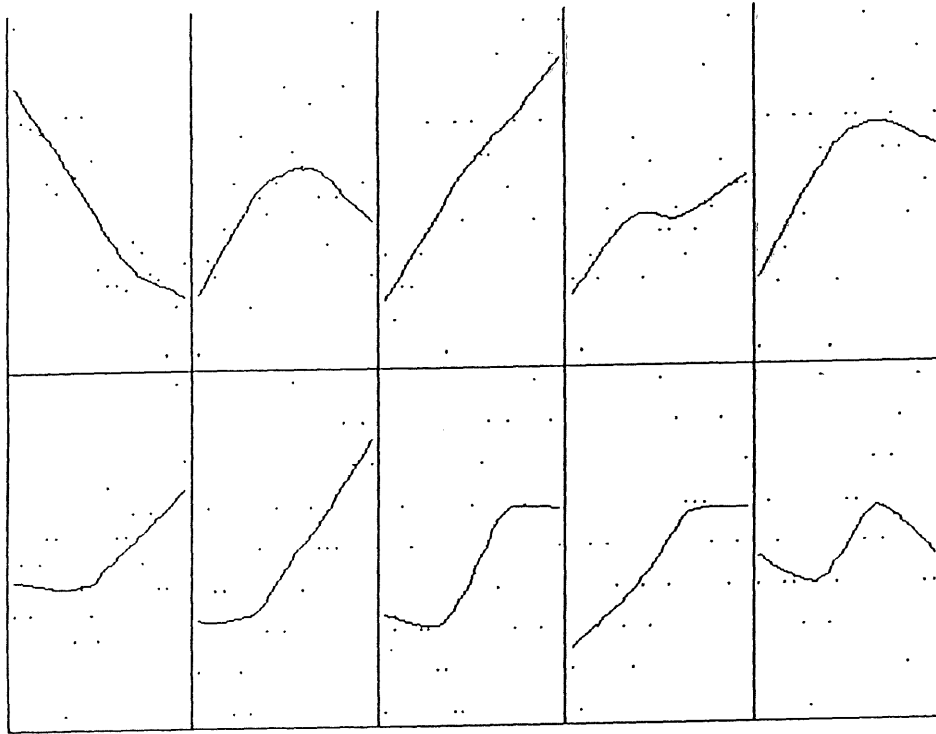
EGFS Biomass
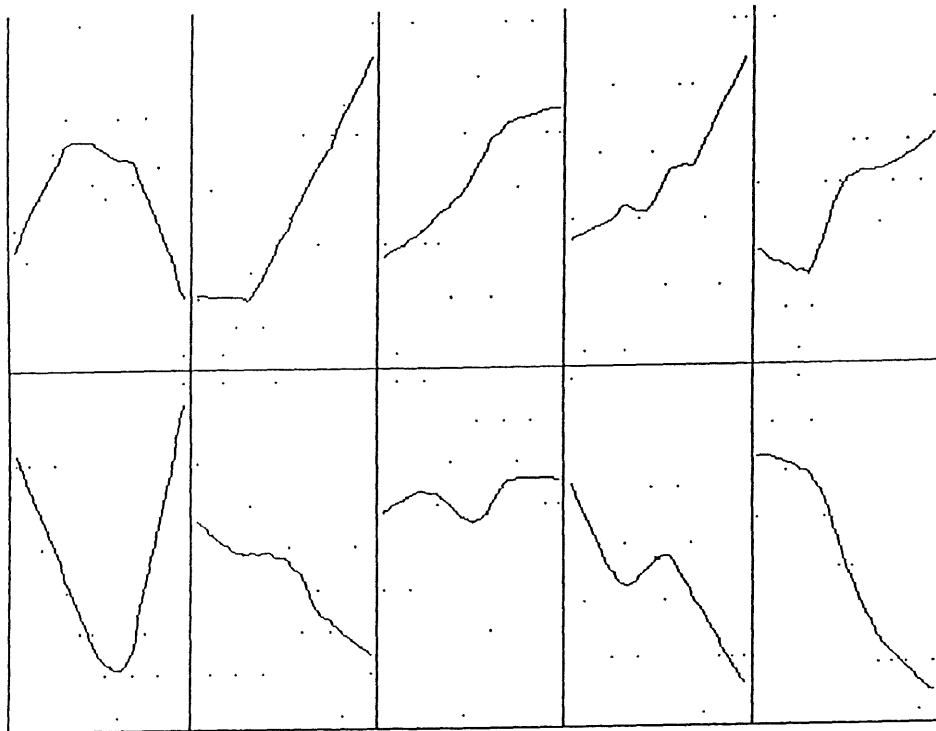
Figure 3.5.1. Plots of slopes and intercepts of the numbers and diversity size spectra for the Scottish Survey data from the three North Sea regions shown in Figures 3.3.1.1.
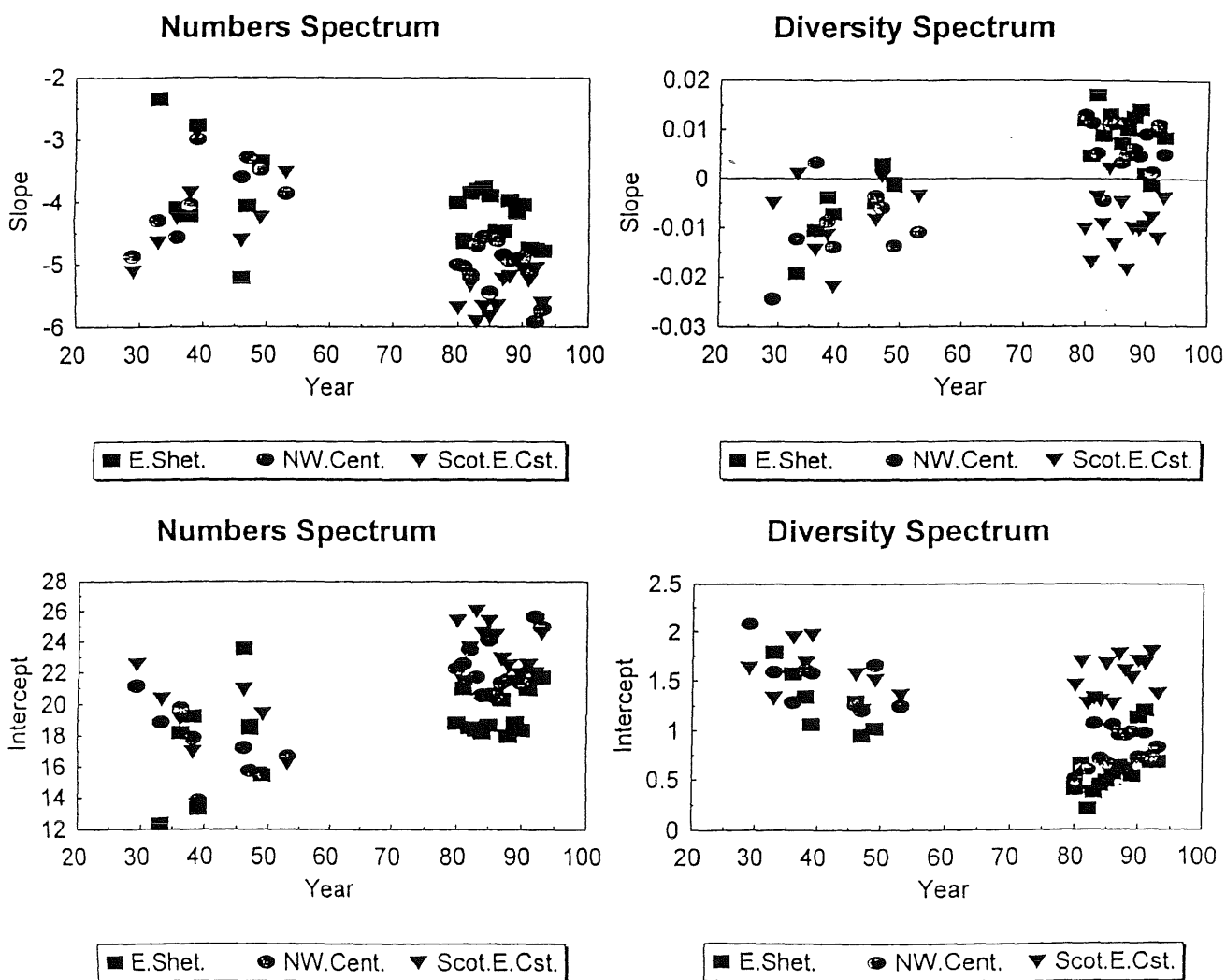
Figure 3.5.2. Diversity size spectra for the Scottish Survey data from 1929–1953 (open symbols, dashed line) and 1980–1993 (filled symbols, solid line) Lines are lowess smooths through each data set.
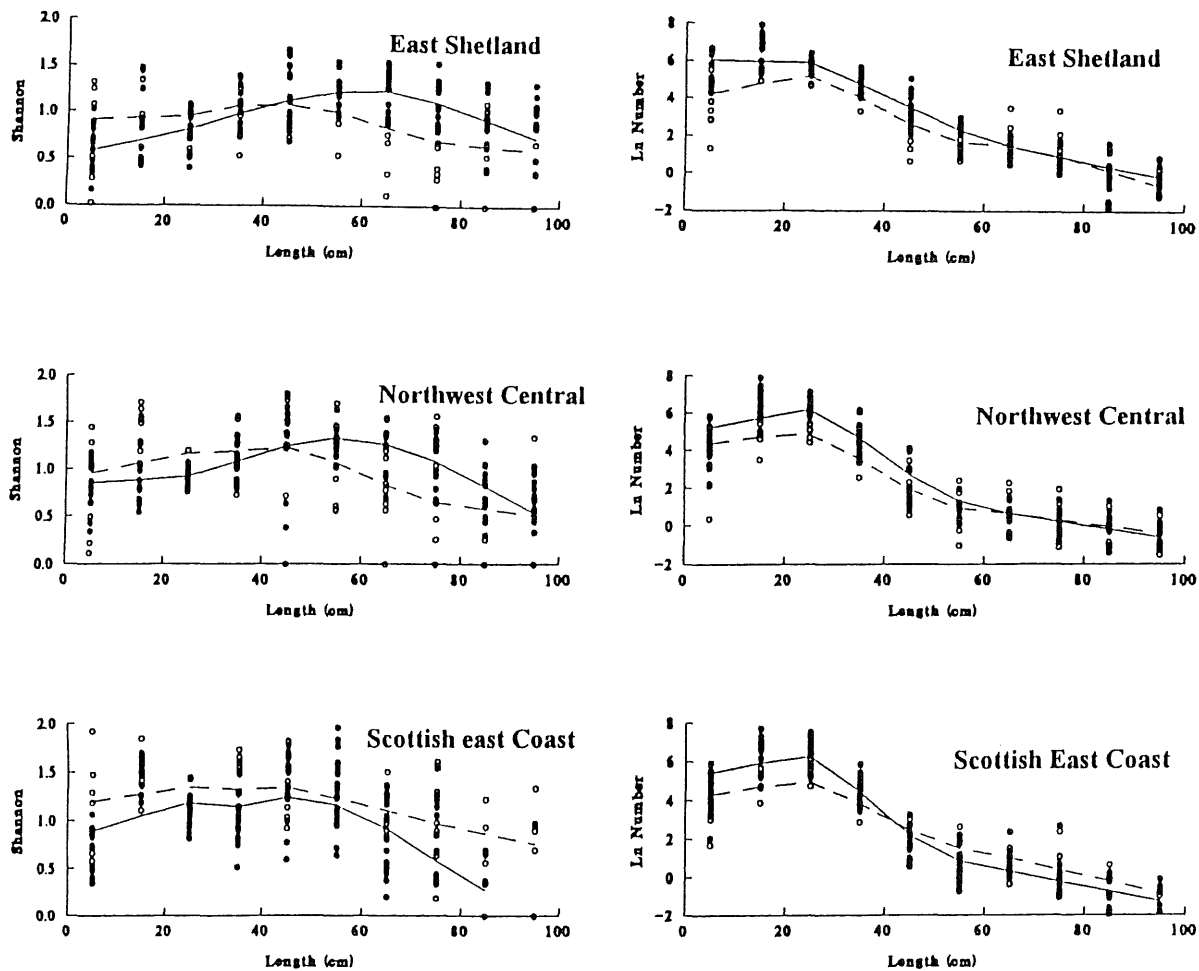
Figure 3.5.3. Species richness *versus* length for Scottish Survey data from 1929–1953 and 1980–1993..

Species Richness
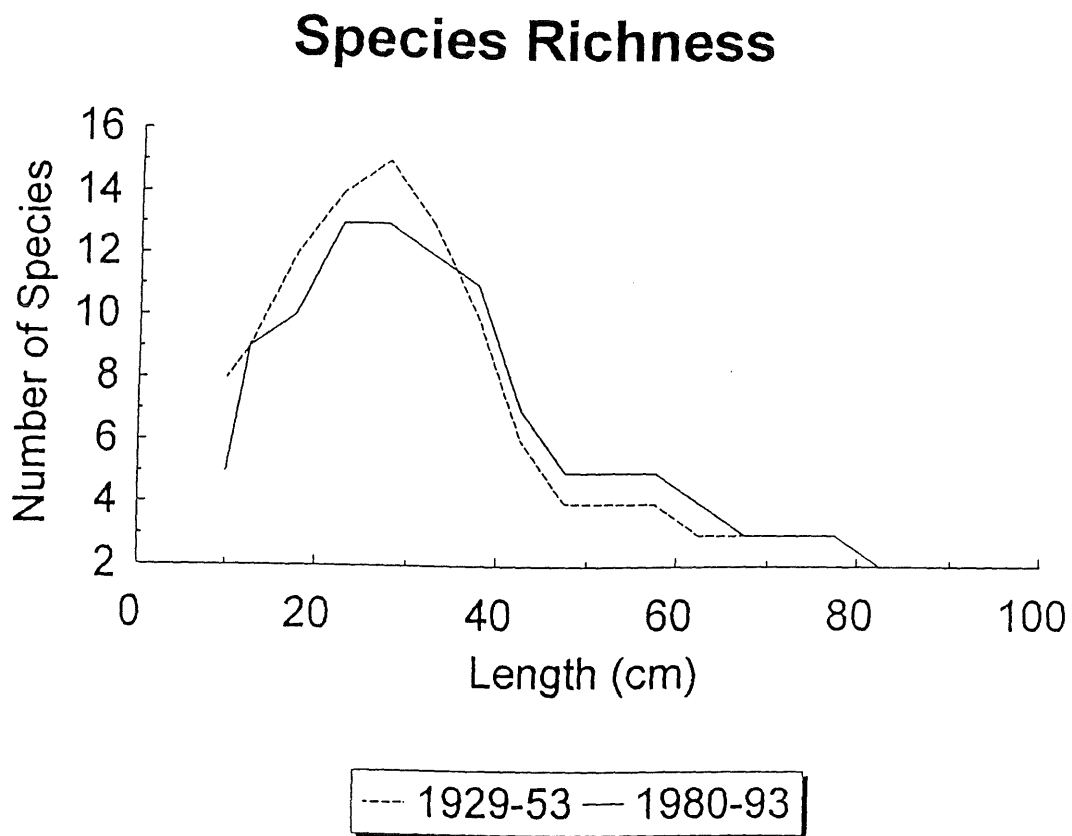
Figure 3.6.3.1. International Beam Trawl Surveys from 1990 to 1995 (Corystes, Isis, Belgica, Solea). Total number of species (above) and total number of hauls (below).
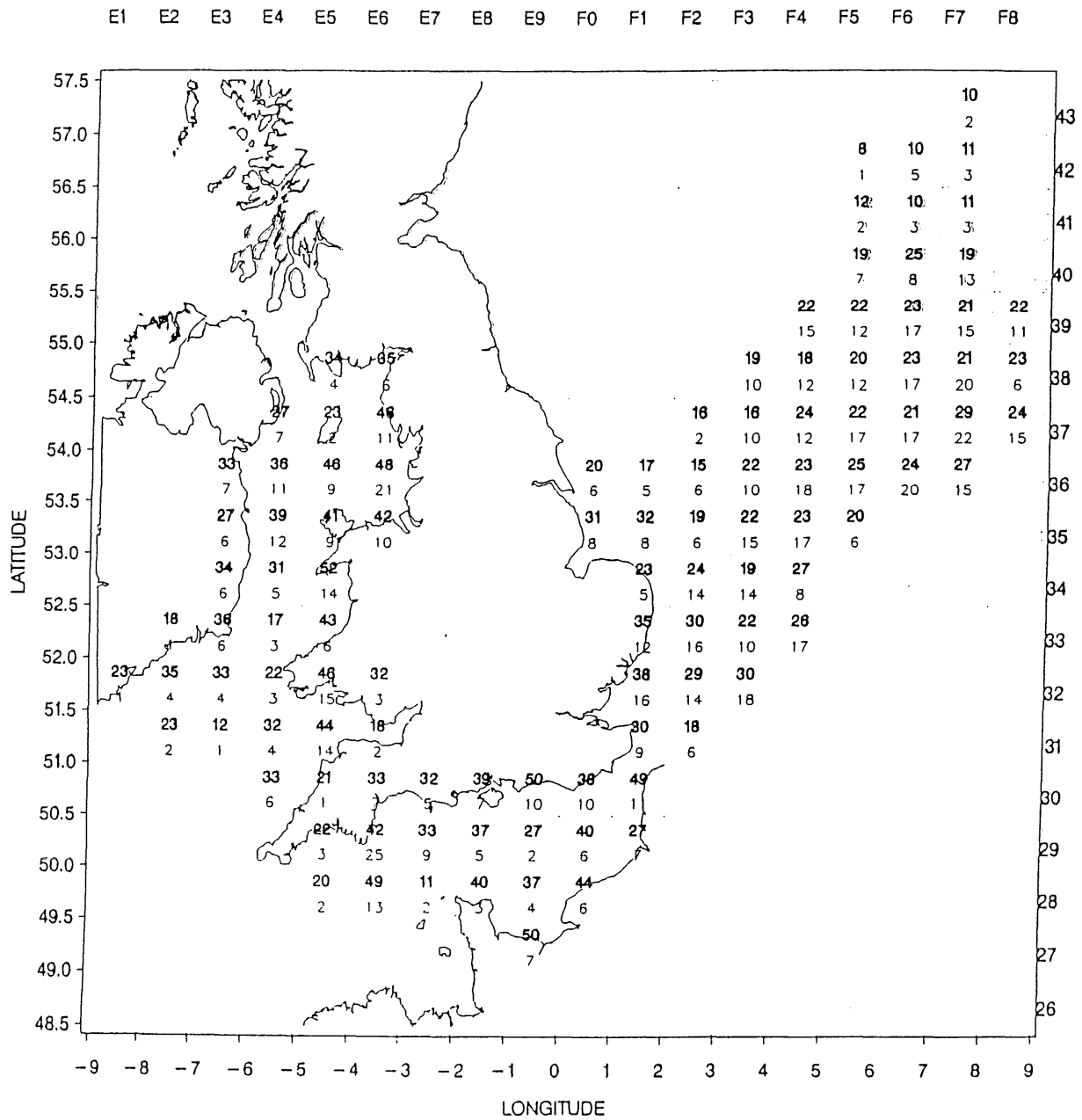
Figure 3.6.3.2. International Beam Trawl Surveys from 1990 to 1995 (Corystes, Isis, Belgica, Solea). Shannon Weiner diversity index (demersal commercial and non-target).
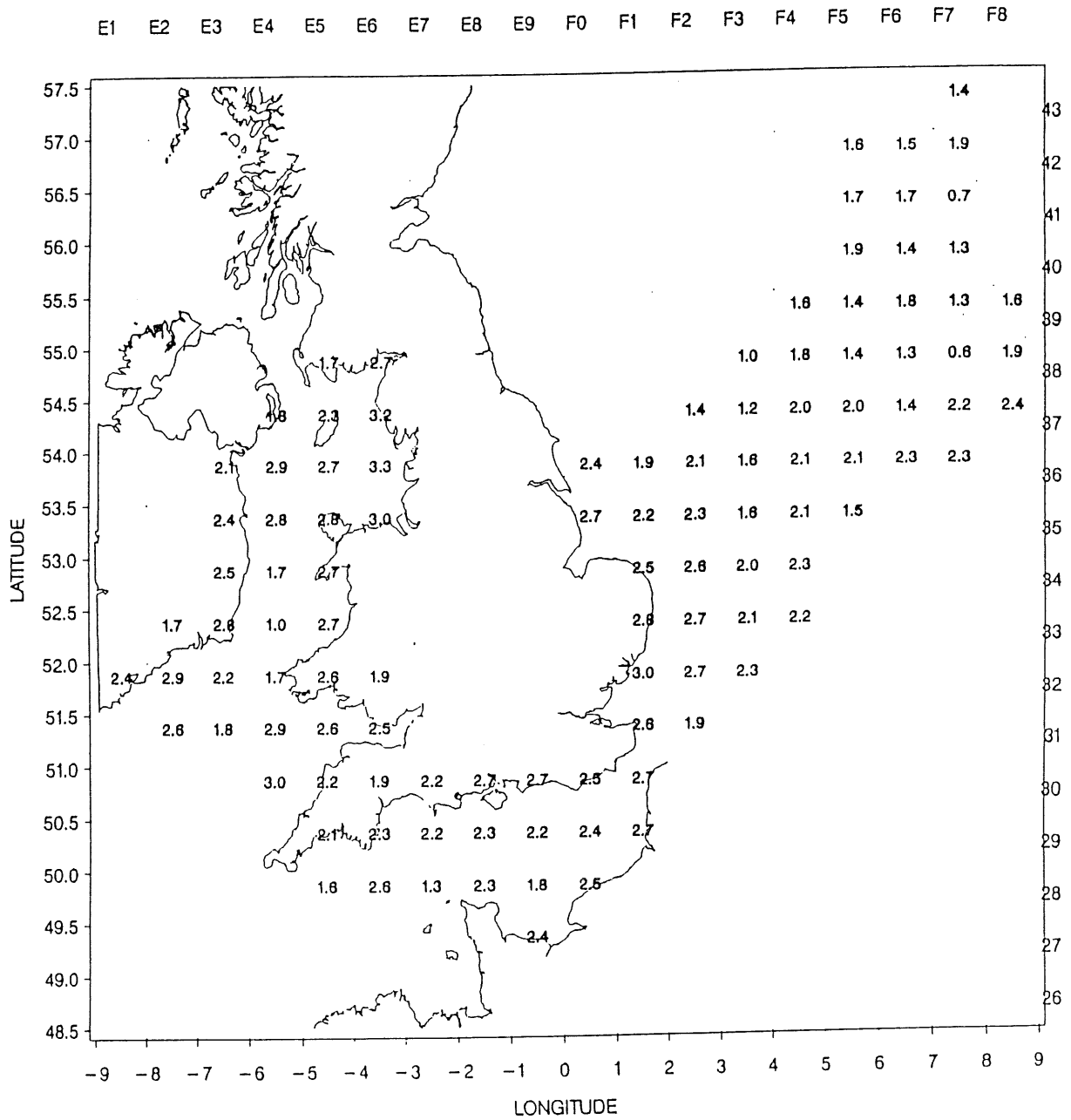
Figure 3.6.3.3.a. Length frequency distribution of beam trawl catches in Q3 in North Sea Task Force Areas.

5 cm length groups — (zero = 0 — 4.9cm)

NST   3—3—3- 3b   4·-4·-4· 4   5·-5·-5· 5   6—6—6· 6   7—7—7- 7a   8—8—8- 7b
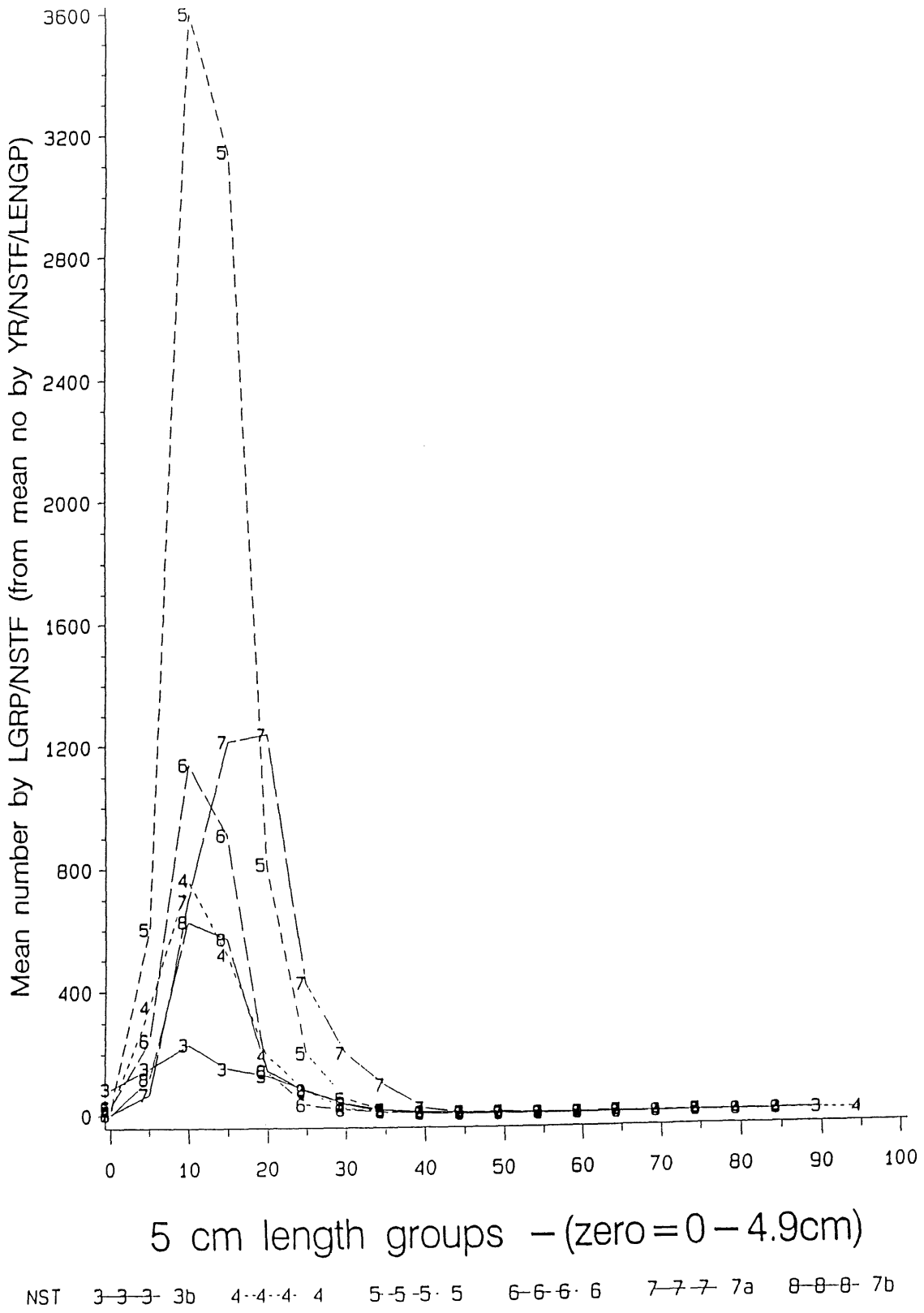
Figure 3.6.3.3.b. Length frequency distribution of beam trawl catches in Q3 (west and south UK, by Division).

5 cm length groups −(zero = 0 − 4.9cm)

DIVN   1—1—1- 7A   2·2·2· 7D   3-3-3· 7E   4-4-4· 7F   5-5-5- 7G
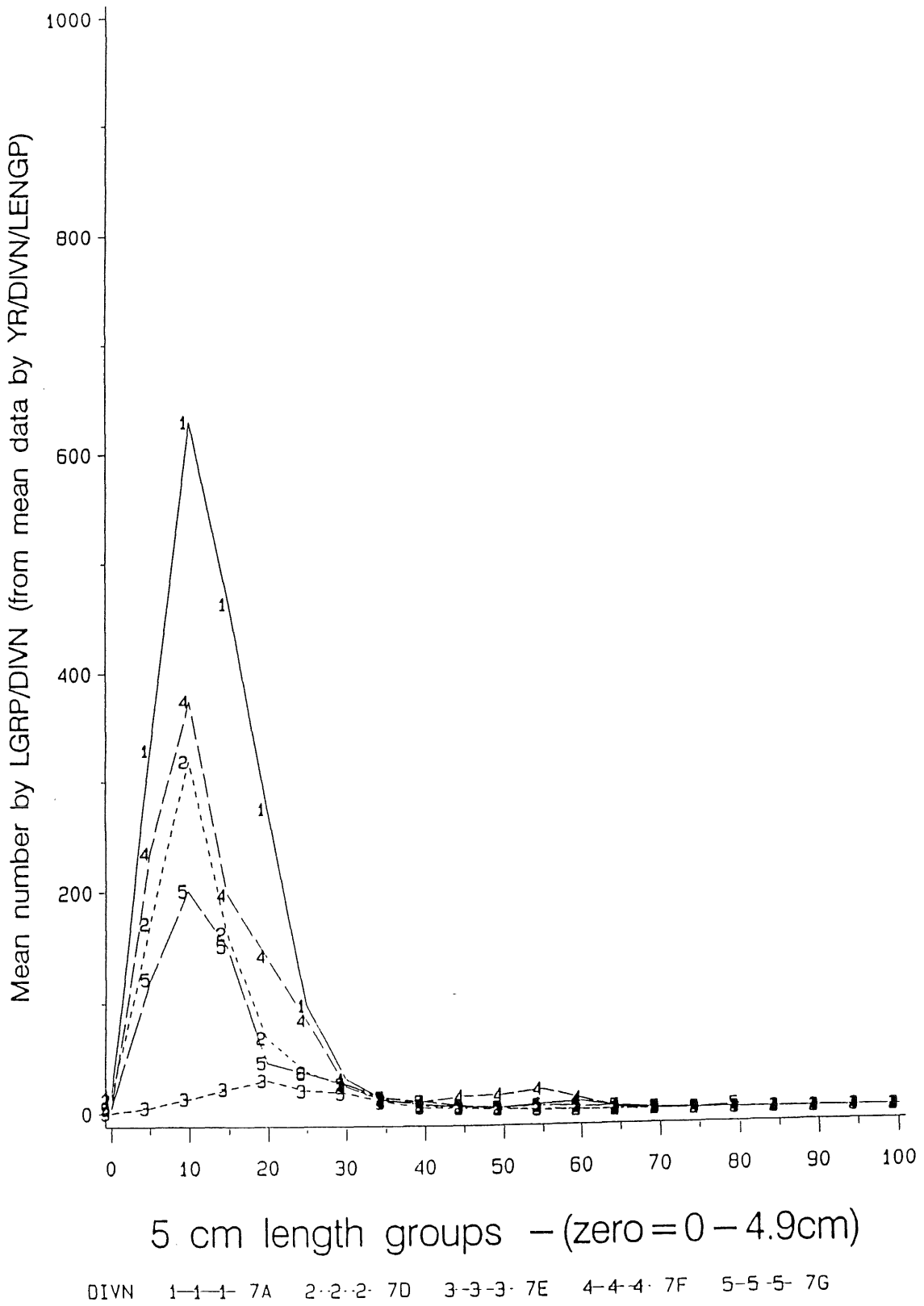
Figure 3.6.3.4.a. Shannon Weiner diversity index for International Beam Trawl data. North Sea Task Force Areas (3b = 3; 4 = 4; 5 = 5; 6 = 6; 7a = 7; 7b = 8).
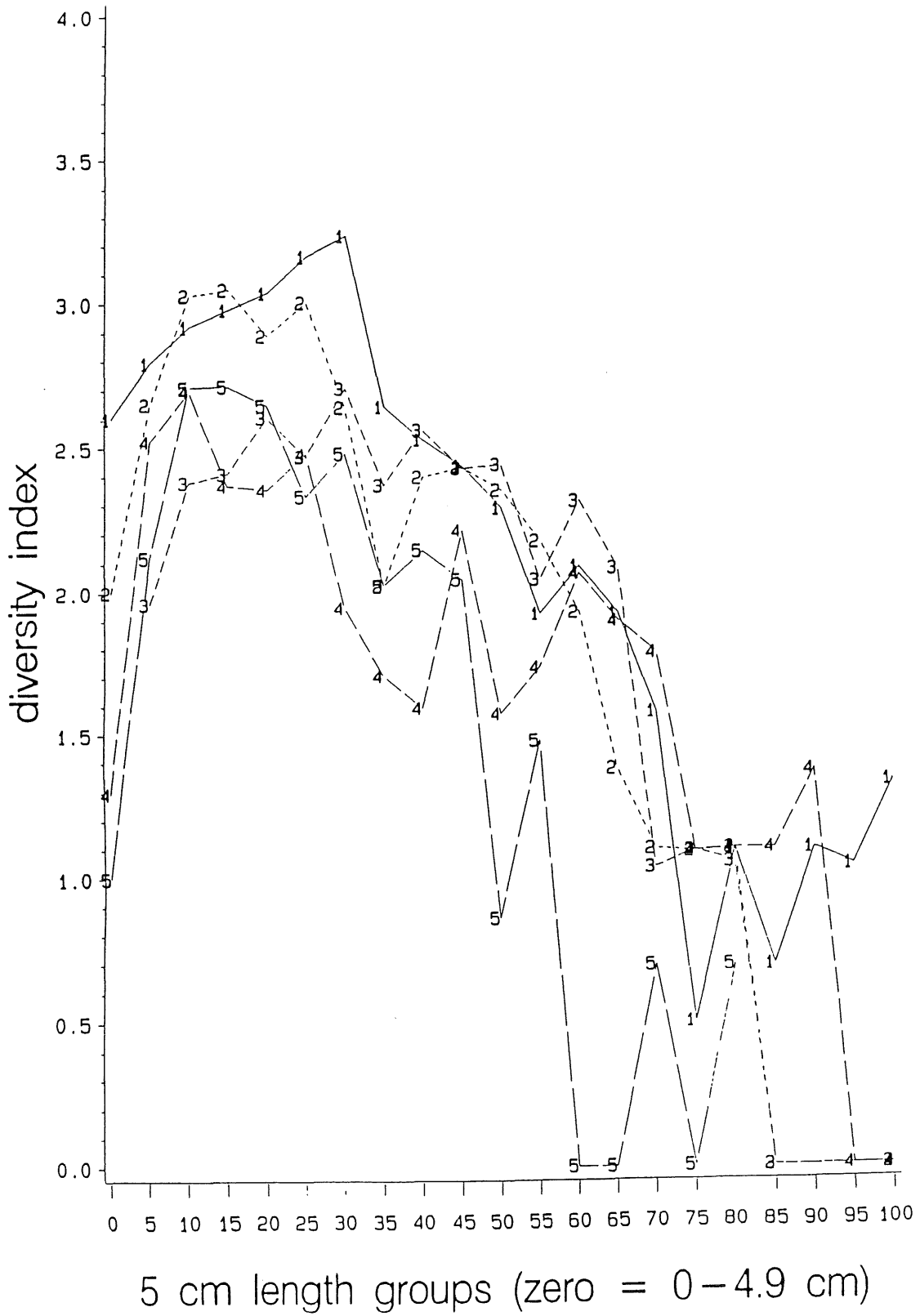
Figure 4.2.2.1. Abundance of cod in three regions of the northwestern North Sea during 1929–1953 and 1980–1993.

Cod: East Shetland

Cod: NW Central

Cod: Scottish east coast
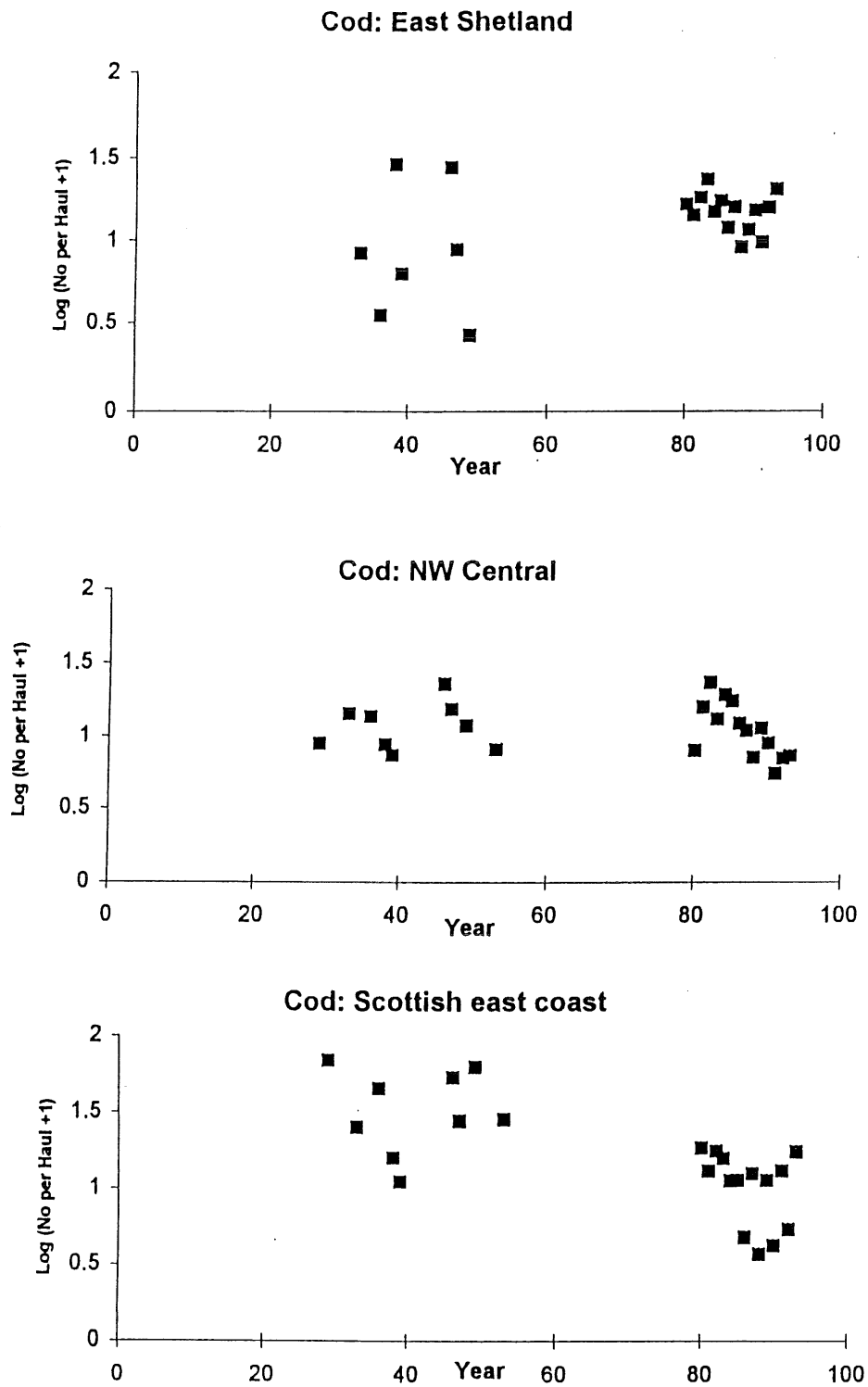
Figure 4.2.2.2. Abundance of grey gurnard in three regions of the northwestern North Sea during 1929–1953 and 1980–1993.
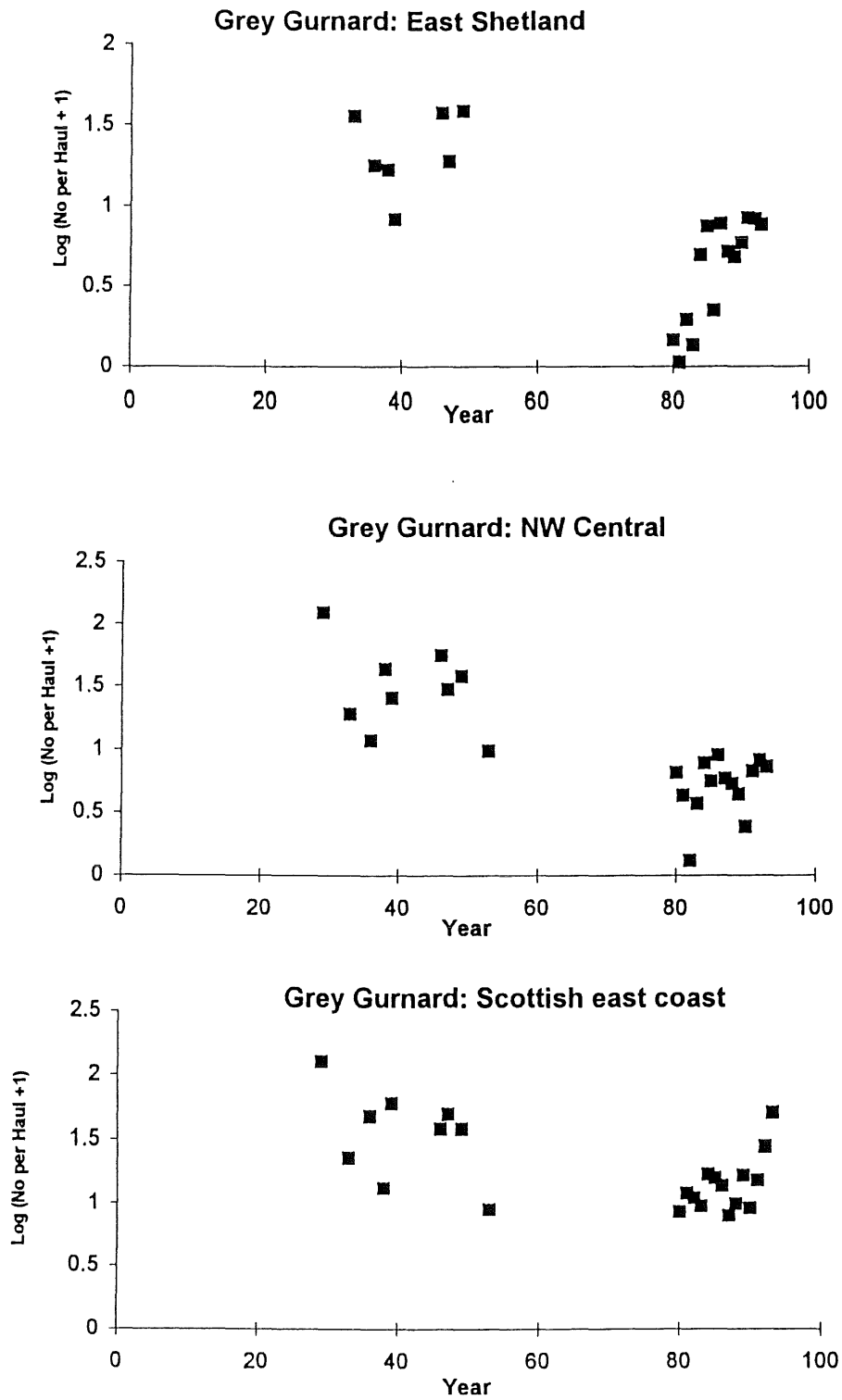

Grey Gurnard: East Shetland

Grey Gurnard: NW Central

Grey Gurnard: Scottish east coast

Figure 4.2.2.3. Abundance of skate in three regions of the northwestern North Sea during 1929–1953 and 1980–1993.

Skate: East Shetland

Skate: NW Central

Skate: Scottish east coast
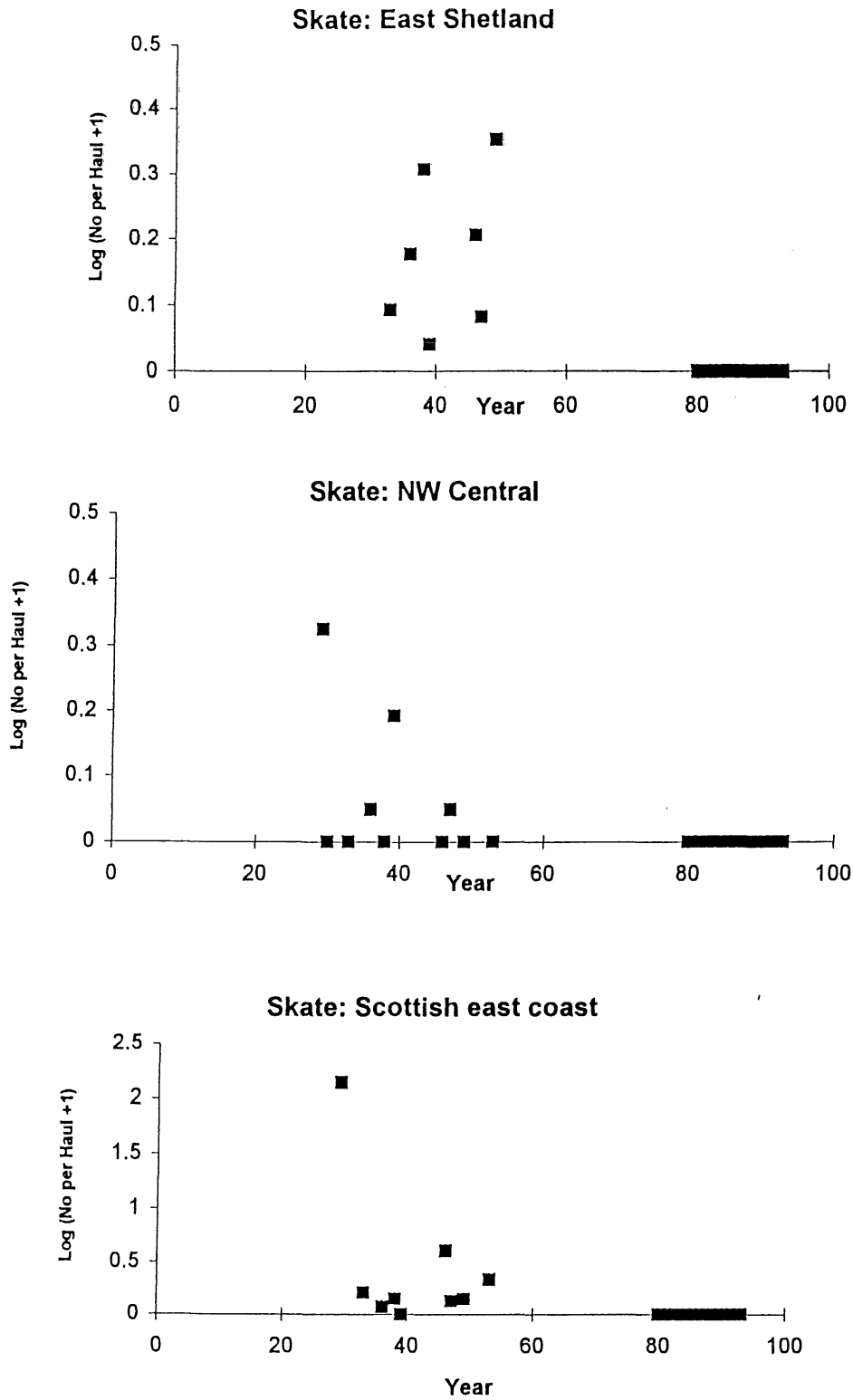
Figure 4.2.2.4. Abundance of spurdog in three regions of the northwestern North Sea during 1929–1953 and 1980–1993.

Spurdog: East Shetland

Spurdog: NW Central
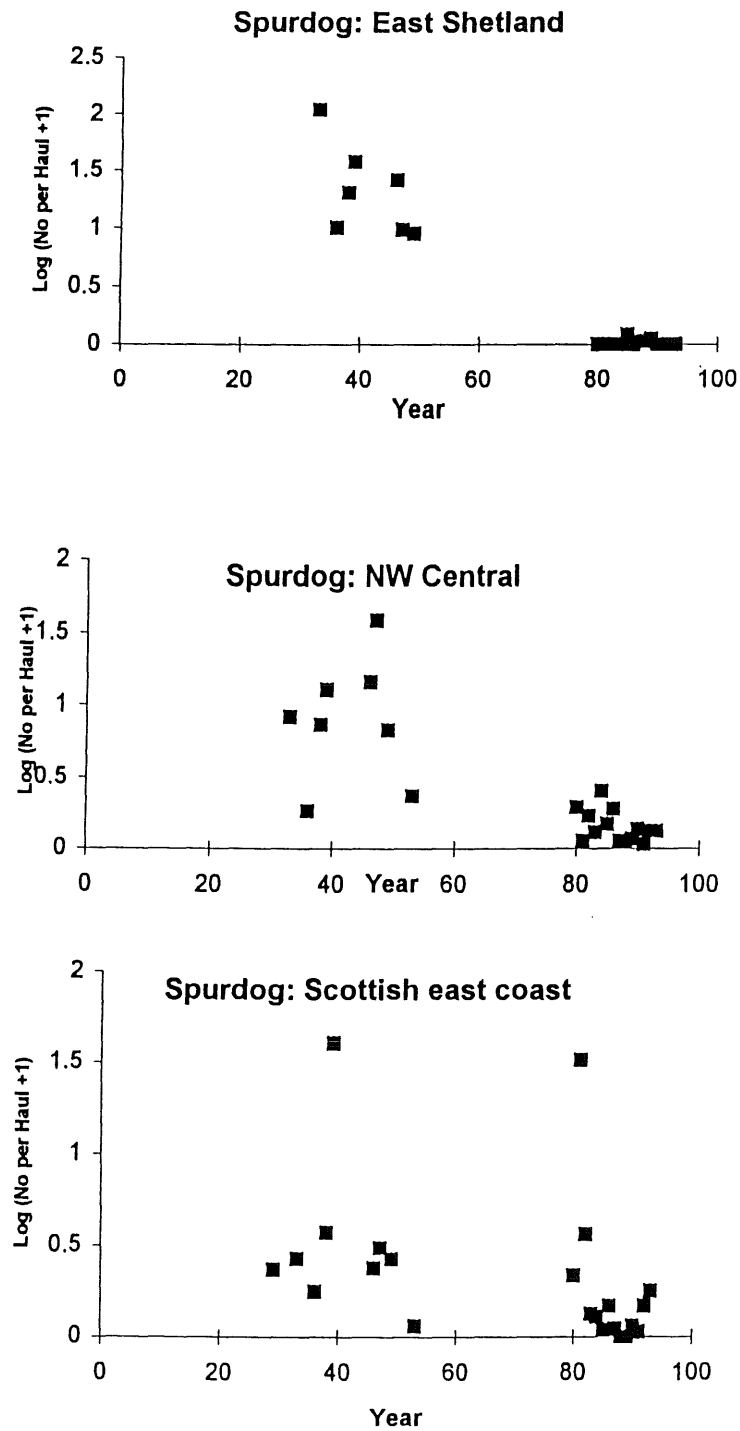

Spurdog: Scottish east coast

Figure 4.3.1.1. Temporal changes in the abundance of macrofauna in samples taken in September at the inshore station of the Northumberland benthic time series and fishing effort in the surrounding ICES statistical rectangle 39E8 (N.B. Benthic data from 1987 missing).
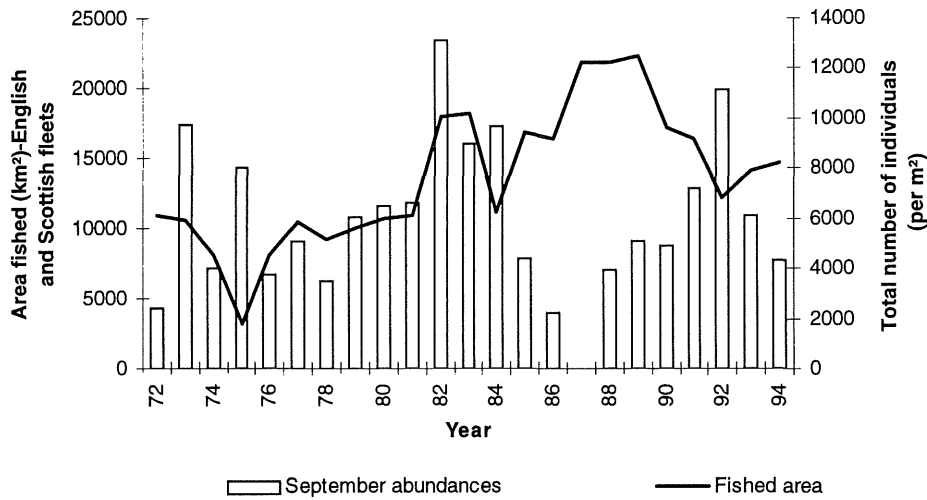

Table 5.3.4.1. Fishing mortality (F) on infaunal species by beam trawls under different assumptions on the proportion of animals dying due to the passage of the gear.

Species	Beam trawls < 300 bhp				Beam trawls > 300 bhp			
	% Dead	F	% Dead	F	% Dead	F	% Dead	F
Amphiura filiformis	100	0.008	10	0.001	100	0.535	10	0.054
Echinocardium cordatum	100	0.111	85	0.094	100	0.579	85	0.492
Tellimya ferruginosa	100	0.107	80	0.085	100	0.652	80	0.522

Figure 5.4.1.1. Parameters of spatially explicit population model (see text for full explanation).
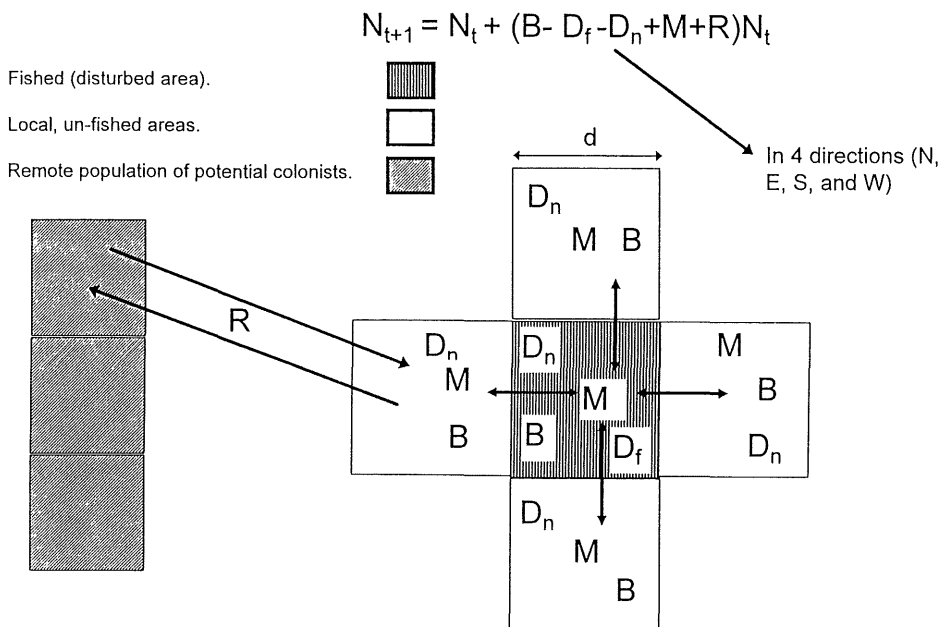
Figure 6.5.4.1 Map of the plaice box (hatched area) and reference areas N and K studied during the 8 m beam trawl survey since 1985.
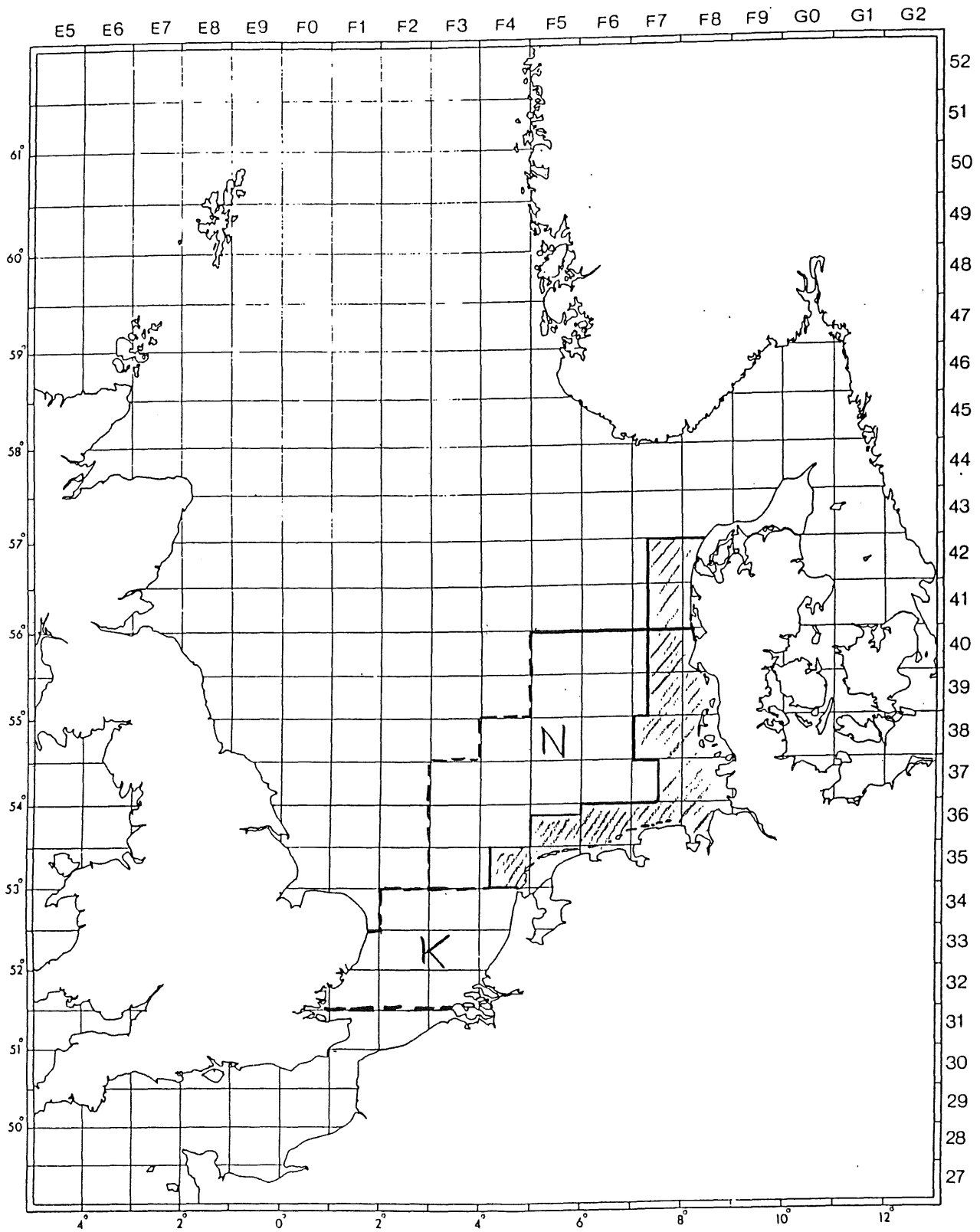
Figure 6.5.4.2. Size distributions of all fish species (left) and non-target species (right) from 8 metre beam trawl surveys before (1985–1988) and after (1989–1994) establishment of a closed area (plaice box) in the southeastern North Sea. The top panel refers to the plaice box area itself, the middle and bottom panels refer to reference areas N and K, respectively.
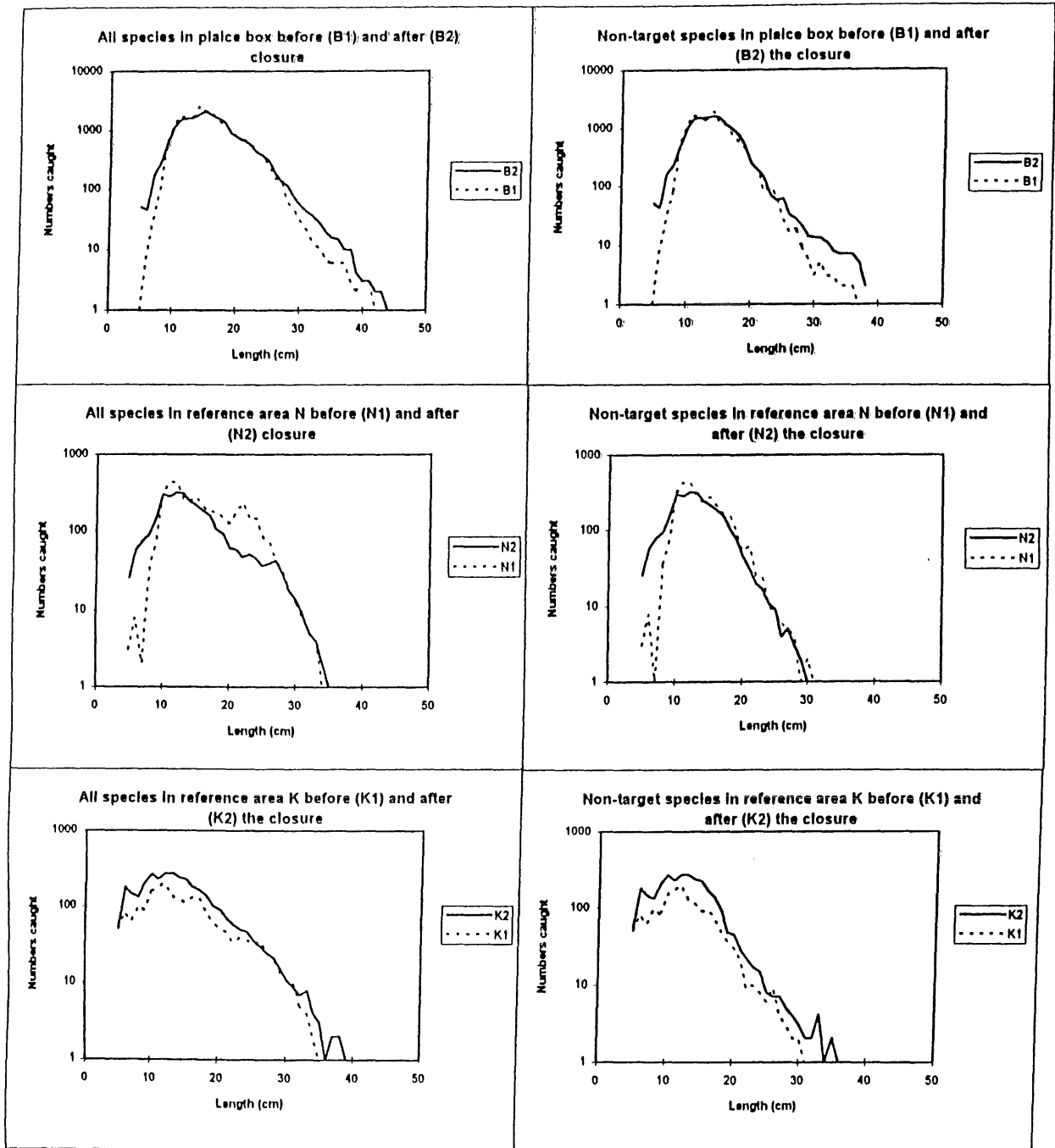
Figure 6.5.4.3. Time trends in number of species (top), Shannon Weiner diversity (middle), and evenness (bottom) in the plaice box and reference areas N and K.
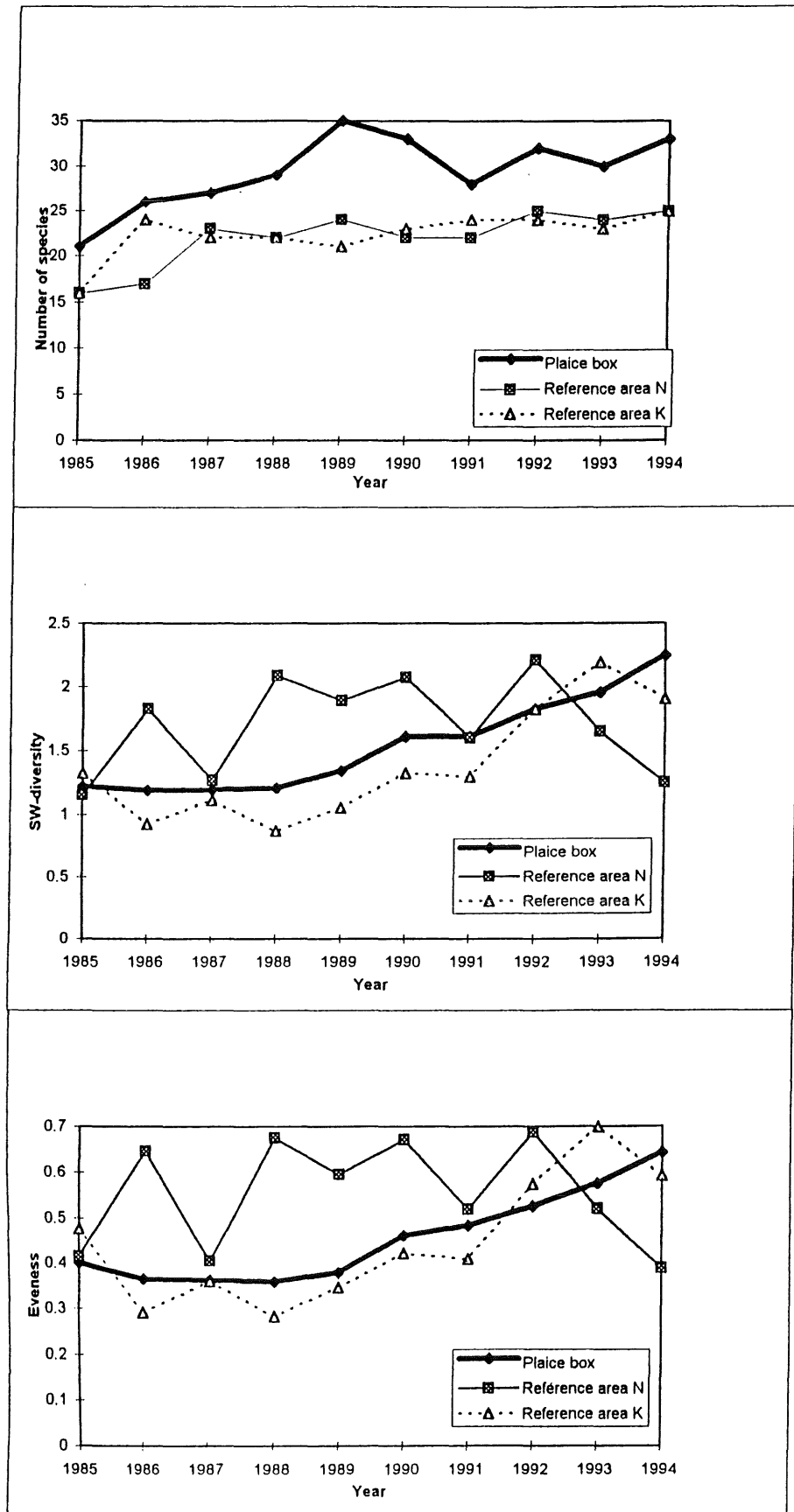
Figure 6.6.1. Regions determined, for the reasons described above, to be less suited for the establishment of undisturbed areas for scientific studies of the response of the marine ecosystem to a cessation of fishing activities. [N.B. B, C, D, and E are areas for which long time series of data are available.]
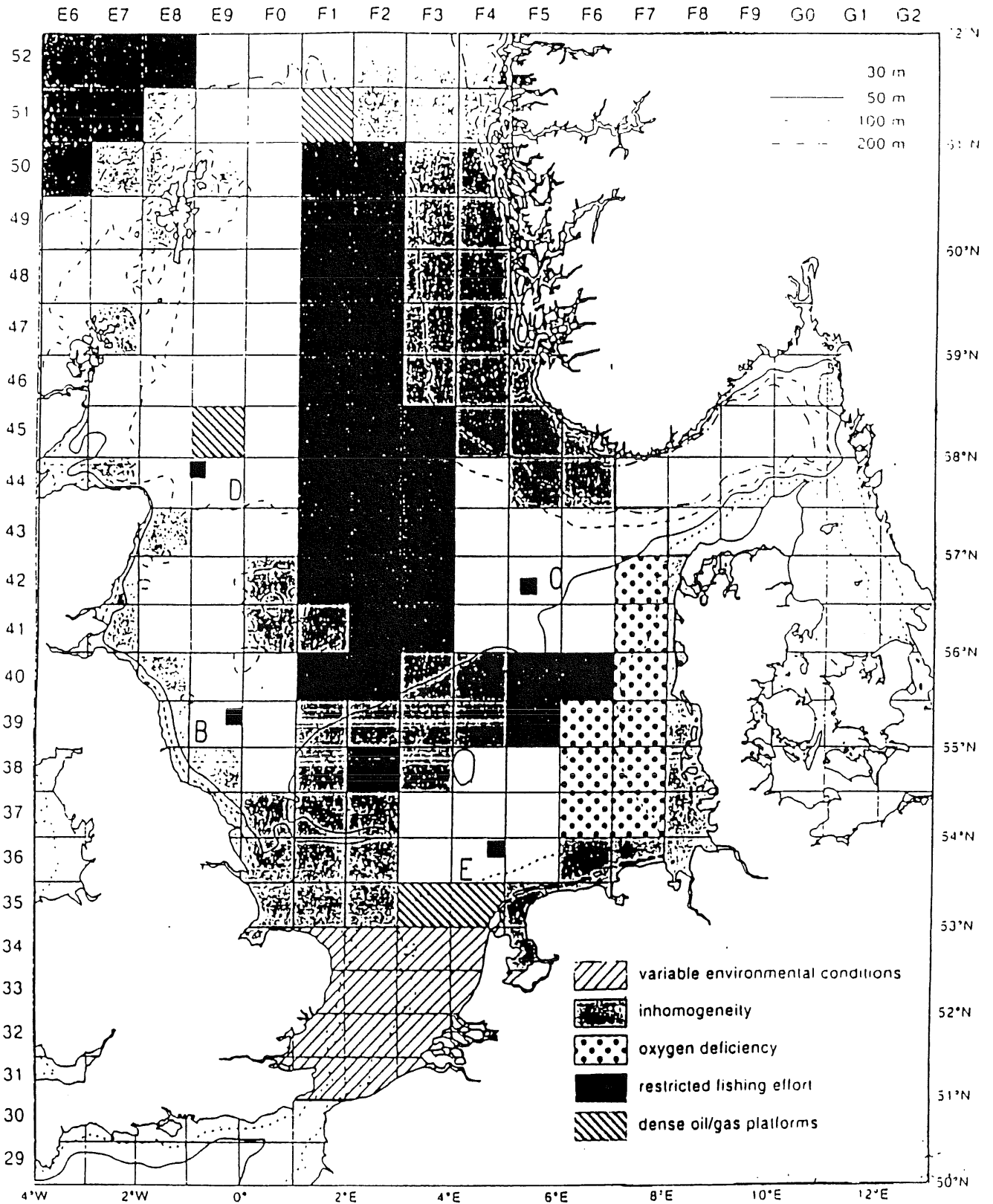
Table 7.3.1.1. Species caught in Scottish groundfish surveys and their status in Scottish commercial fisheries and landings (from Greenstreet and Hall, 1996; and Anon., 1995).

Species	Target	Landings (Tonnes)
Hagfish *Myxine glutinosa*	Non-target	
Lesser spotted dogfish *Scyliorhinus canicula*	Non-target	
Nurse hound *Scyliorhinus stellaris*	Non-target	
Tope *Galeorhinus galeus*	Non-target	
"Sharks"		50
Spurdog *Squalus acanthias*	Target	3,434
Blond ray *Raja brachyura*	Non-target	
Thornback ray *Raja clavata*	Non-target	
Spotted ray *Raja montagui*	Non-target	
Starry ray *Raja radiata*	Non-target	
Skate *Raja batis*	Target	2,985
Shagreen ray *Raja fullonica*	Non-target	
Sandy ray *Raja circularis*	Non-target	
Cuckoo ray *Raja naevus*	Non-target	
Sting ray *Dasyatis pastinaca*	Non-target	
Conger eel *Conger conger*	Non-target	94
Herring *Clupea harengus*	Target	78,434
Sprat *Sprattus sprattus*	Target	1,588
Lesser argentine *Argentina sphyraena*	Non-target	80
Greater argentine *Argentina silus*	Non-target	
Angler *Lophius piscatorius*	Target	11,849
Blackbellied angler *Lophius budegassa*	Target	
Cod *Gadus morhua*	Target	29,522
Silvery pout *Gadiculus argenteus*	Non-target	
Haddock *Melanogrammus aeglefinus*	Target	72,327
Whiting *Merlangius merlangus*	Target	29,671
Blue whiting *Micromesistius poutassou*	Non-target	1,936
Poor cod *Trisopterus minutus*	Non-target	
Norway pout *Trisopterus esmarkii*	Target	0
Bib *Trisopterus luscus*	Non-target	
Pollack *Pollachius pollachius*	Non-target	770
Saithe *Pollachius virens*	Target	8,171
Torsk *Brosme brosme*	Non-target	228
Five-bearded rockling *Ciliata mustela*	Non-target	
Four-bearded rockling *Enchelyopus cimbrius*	Non-target	
Three-bearded rockling *Gaidropsarus vulgaris*	Non-target	
Ling *Molva molva*	Non-target	5,094
Greater forkbeard *Phycis blennoides*	Non-target	7
Hake *Merluccius merluccius*	Target	1,214
John dory *Zeus faber*	Non-target	2
Boarfish *Capros aper*	Non-target	
Bluemouth *Helicolenus dactylopterus*	Non-target	2
Norway haddock *Sebastes viviparus*	Non-target	
Piper gurnard *Trigla lyra*	Non-target	
Red gurnard *Aspitrigla cuculus*	Non-target	4
Grey gurnard *Eutrigla gurnardus*	Non-target	34
Moustache sculpin *Triglops murrayi*	Non-target	
Bullrout *Myoxocephalus scorpius*	Non-target	
Sea scorpion *Taurulus bubalis*	Non-target	

Table 7.3.1.1. Continued.

Species	Target	Landings (Tonnes)
Hooknose *Agonus cataphractus*	Non-target	
Lumpsucker *Cyclopterus lumpus*	Non-target	
Comber *Serranus cabrilla*	Non-target	
Horse mackerel *Trachurus trachurus*	Non-target	4,631
Lesser weever *Echiichthys vipera*	Non-target	
Catfish *Anarhichas lupus*	Non-target	827
Snake blenny *Lumpenus lampretaefornis*	Non-target	
Butterfish *Pholis gunnellus*	Non-target	
Raitt's sandeel *Ammodytes marinus*	Target	8,690
Smooth sandeel *Gymnammodytes semisquamatus*	Non-target	
Greater sandeel *Hyperoplus lanceolatus*	Non-target	
Dragonet *Callionymus lyra*	Non-target	
Spotted dragonet *Callionymus maculatus*	Non-target	
Unidentified gobies *Gobius* spp.	Non-target	
Mackerel *Scomber scombrus*	Target	137,521
Brill *Scophthalamus rhombus*	Non-target	13
Turbot *Scophthalamus maximus*	Target	202
Megrim *Lepidorhombus whiffiagonis*	Target	3,019
Norwegian topknot *Phrynorhombus norvegicus*	Non-target	
Topknot *Zeugopterus punctatus*	Non-target	
Plaice *Pleuronectes platessa*	Target	5,375
Flounder *Platichthys flesus*	Non-target	2
Common dab *Limanda limanda*	Non-target	373
Lemon sole *Microstomus kitt*	Non-target	2,378
Witch *Glyptocephalus cynoglossus*	Non-target	1,820
Long rough dab *Hippoglossoides platessoides*	Non-target	
Halibut *Hippoglossus hippoglossus*	Non-target	138
Solenette *Buglossidium luteum*	Non-target	
Others	Non-target	921

Table 7.3.2.1. Species composition in landings by industrial fisheries in the North Sea (tonnes) in 1994 and means for 1974–1993. For the species marked with an asterisk, mean catches are for fewer years (usually 1984–1993), see ICES (1996a) for further details.

Species	1994	Mean 1974–1993	
Sandeel *Ammodytes* spp.	769,000	690,000	
Sprat *Sprattus sprattus*	281,000	214,000	
Herring *Clupea harengus*	40,000	72,000	
Norway pout *Trisopterus esmarkii*	172,000	303,000	
Blue whiting *Micromesistius poutassou*	11,000	55,000	
Haddock *Melanogrammus aeglefinus*	5,000	16,000	
Whiting *Merlangius merlangus*	10,000	52,000	
Saithe *Pollachius virens*	0	10,000	
Cod *Gadus morhua*	876	1,650	*
Mackerel *Scomber scombrus*	3,576	4,060	*
Horse mackerel *Trachurus trachurus*	4,886	13,500	*
Gurnards *Trigla* spp.	1,139	7,720	*
Common dab *Limanda limanda*	528	3,880	*
Argentines *Argentina* spp.	2,209	3,780	*
Long rough dab *Hippoglossoides platessoides*	529	1,100	*
Plaice *Pleuronectes platessa*	143	340	*
Hake *Merluccius merluccius*	10	280	*
Poor cod *Trisopterus minutus*	0	46	*
Ling *Molva molva*	0	80	*
Witch *Glyptocephalus cynoglossus*	40	340	*
Silvery pout *Gadiculus argenteus*	0	1,220	*
Others	5,141	9,200	*

Table 7.3.2.2. Distribution of landings and associated by-catches of selected species ('000 tonnes) from industrial fisheries by Denmark and Norway by landing categories to the north and south of 57 °N, respectively, in 1994 (from ICES, 1996a). + denotes less than 1000 tonnes.

Northern area
Species composition

Target species	Norway pout	Sandeel	Sprat	Herring	Haddock	Whiting	Saithe	Blue whiting	Other	Total
Norway pout	170	0	+	5						193
Sandeel	0	442	16	4						465
Sprat	10	+	63	14						95
Other	2	+	+	+						6
Sum	182	442	79	23						759

Southern area
Species composition

Target species	Norway pout	Sandeel	Sprat	Herring	Haddock	Whiting	Saithe	Blue whiting	Other	Total
Norway pout	0	0	0	0	0	0	0	0	0	0
Sandeel	+	209	10	2	+	1	0	0	4	315
Sprat	0	13	185	16	+	2	0	0	3	219
Other	0	7	7	+	0	+	0	0	+	14
Sum	0	318	202	18	+	3		0	7	548

Table 7.3.3.1. Estimates of harbour porpoise population size and by-catch in gill nets in northwestern European waters (from ICES, 1996b).

Fishery	Population estimate	By-catch	Source
North Sea Danish cod and turbot gill net	268,500	4,449	Vinther, 1996
Celtic shelf gill net	36,280	1,937	Berrow et al., 1994

Table 7.4.2.1. Summary of species (tonnes) discarded by Spanish fleets in ICES divisions (from Peréz et al., in press). * = from ICES Sub-divisions VIIIb and VIIIc (east); ** = VIIIc (west). The ranges of estimates reflect the exclusion or inclusion of slipping by the fleet.

	VI and VII	VIIIa, b	VIIIc	IXa	Total
Trawlers	20,486	11,155	13,971	28,006	73,600
Long-liners	1,131	503	679		2,300
Gill net			321		300
Purse seine		5,030–10,914*	215**	146	5,400–11,300
Total	21,600	16,700–22,600	15,200	28,200	81,600–87,500

Table 7.4.2.2. Numbers of species discarded by Spanish fleets in ICES divisions (from Peréz et al., in press).

	VI	VII	VIIIa, b	VIIIc	IXa
Trawlers	85	244	155	243	190
Long-liners	21	75	39	57	
Gill net				93	

Table 7.4.2.3. Estimated quantities (rounded to nearest 100 tonnes) of discards and offal in six subregions of the North Sea in 1990 (from Garthe *et al.*, in press). See Figure 7.4.2.1 for areas.

Area	Roundfish	Flatfish	Elasmobranchs	Benthic invertebrates	Offal	Total
NW	54,900	13,100	3,400	7,800	11,700	90,900
NE	53,300	14,300	3,300	8,300	11,400	90,600
CW	26,800	15,000	1,600	7,900	6,000	57,300
C	48,000	61,400	2,700	30,600	11,700	154,400
CE	48,500	68,200	2,700	33,800	12,000	165,200
S	30,700	127,200	1,300	61,400	10,000	230,600
Total	262,200	299,200	15,000	149,800	62,800	789,000

Table 7.4.2.4. Total catch and discards from three beam-trawl trips (August 1994–October 1995) in the German Bight (from Weber, in prep.). "+" denotes less than 1%.

	Total (kg)	%	% of invertebrates	By-catch (kg) per kg of sole landed
Total catch	115790	100		36.7
Discards				
Fish	43923	38		10.6
Total Invertebrates	71867	62	100	22.8
comprising:				
Asterias rubens	29457	25	41	9.3
Ophiura spp.	16985	15	24	5.4
Echinocardium cordatum	9912	9	14	3.1
Psammechinus miliaris	430	+	1	0.1
Corystes cassivelaunus	4388	4	6	1.4
Cancer pagurus	2390	2	3	0.8
Eupagurus berhardus	4626	4	6	1.5
Buccinum undatum	2632	2	4	0.8
Apphrodite aculeata	18	+	+	+
Tunicata spp.	130	+	+	+

Table 7.5.2.1. Proportion of experimental discards and offal consumed by seabirds (%) in six sub-regions of the North Sea (from Garthe *et al.*, in press). The sample size of elasmobranchs was very low, so the overall consumption rate in the North Sea is given. The areas are shown in Figure 7.4.2.1.

Area	Roundfish	Flatfish	Elasmobranchs	Benthic invertebrates	Offal	Sample size
NW	90	28		9	99	9,132
NE	89	41		3	98	3,281
CW	84	32	12	1	92	5,316
C	75	14		1	90	8,519
CE	63	10		3	54	3,396
S	71	8		4	100	1,200
Sample size	21,848	2,345	34	902	5,715	30,844

Figure 7.4.2.1 Map of the six sub-regions of the North Sea used to describe amounts of offal and fish discarded (from Garthe *et al.* in press).
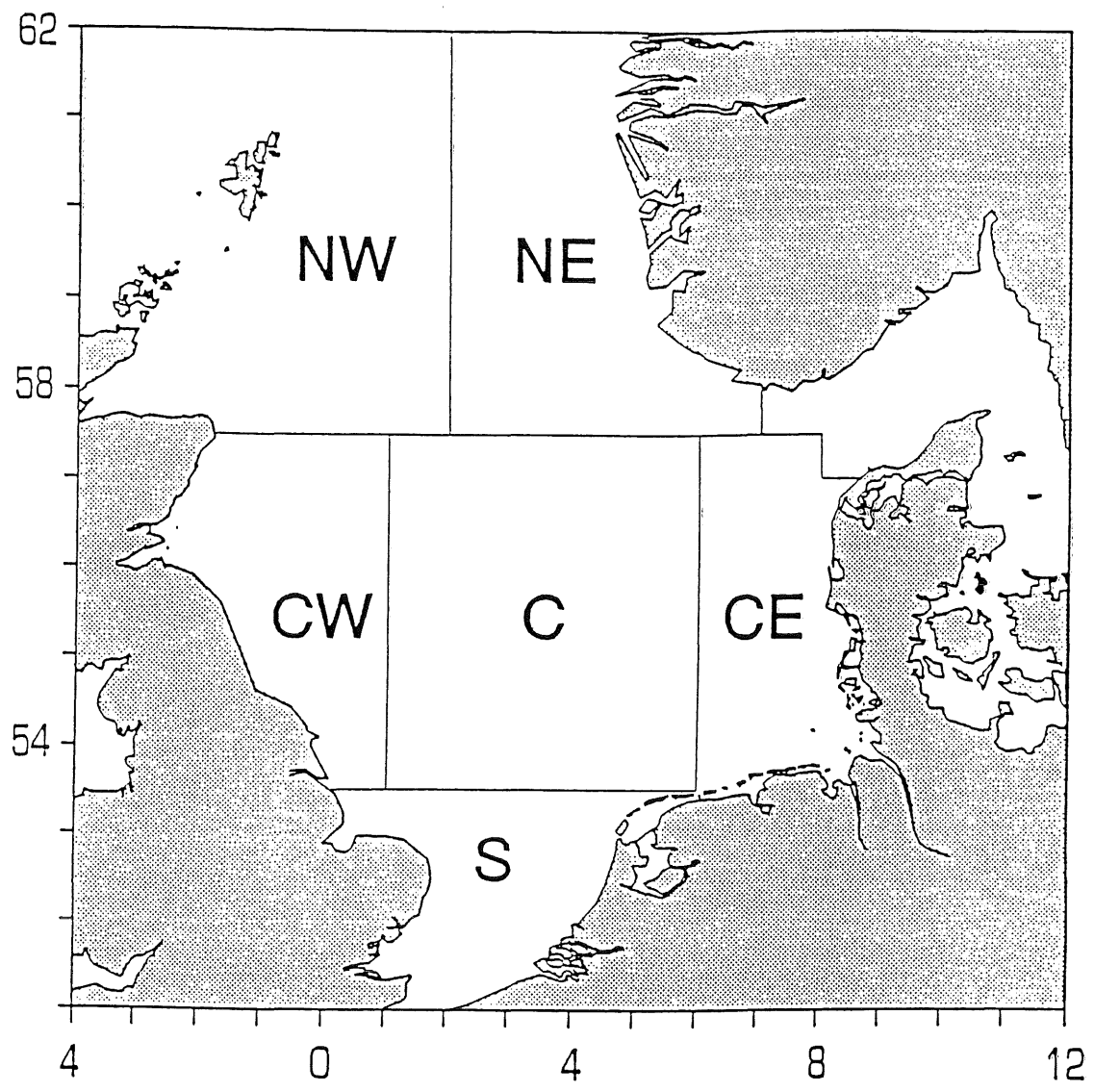
Table 9.1. Known relevant studies on the ecosystem effects of fishing activities in the ICES area. Divisions are shown in Figure 9.1. 1 = study has occurred, 0 = no study known of; Y = study in progress, N = no study in progress. Blank space in table indicates that no known study has occurred or is taking place.

Topics	North Sea	Arctic Atlantic	Celtic Seas	Bay of Biscay	Central Atlantic	NW Atlantic	Baltic
Benthic communities							
description	1, Y	1, Y	1, Y	1, Y	1, Y	1, Y	1, Y
signal species	1, Y		1, Y	1, Y			
influence on abundance	1, Y		1, Y			1, Y	1, Y
community structure	1, Y		1, Y			0, Y	1, Y
mortality	1, Y		1, Y				1, Y
habitat alteration	1, Y		1, Y			1, Y	1, Y
Birds							
description	1, Y	1, Y	1, Y			1, N	1, Y
influence on mortality	1, Y	1, Y	1, Y	1, N		1, Y	1, Y
reduction in food	1, N					1, Y	
increased food	1, Y		1, N				1,
populations	1, Y		1, Y			1, N	1,
Mammals							
description	1, Y	1, Y	1, Y	1, Y		1, Y	1, Y
incidental mortality	1, Y		1, Y	1, Y		1, Y	1, Y
reduction in food	0, Y						
increased food							
influence on population							
Fish and Fishing							
description of stocks	1, Y	1, Y	1, Y	1, Y	1, Y	1, Y	1, Y
description of fisheries	1, Y	1, Y	1, Y	1, Y	1, Y	1, Y	1, Y
landings	1, Y	1, Y	1, Y	1, Y	1, Y	1, Y	1, Y
discards	1, Y		1, Y	1, Y			
time trends by gear	1, Y	1, Y	1, Y	1, Y	1, Y	1, Y	1, Y
assessed species							
impact on recruitment	1, Y		1, Y			1, Y	
age distribution	1, Y		1, Y	1, Y			1, Y
growth	1, Y		1, Y	1, Y			1, Y
adult mortality	1, Y		1, Y	1, Y			1, Y
non-assessed stocks	1, Y		1, Y				1, Y
non-targetstocks	1, Y		1, Y				1, Y
changes in abundance	1, Y		1, Y				
Whole system							
change in size structure	1, Y		1, Y			1, Y	
stability	1, Y		1, Y			1, Y	
trophic structure	1, Y		1, Y			1, Y	
biodiversity	1, Y		1, Y				1, Y
Other impacts							
genetic changes				1, Y			
litter	1, Y						

Figure 9.1. Regions of the maritime area covered by the Oslo and Paris Commissions (OSPAR, 1995).
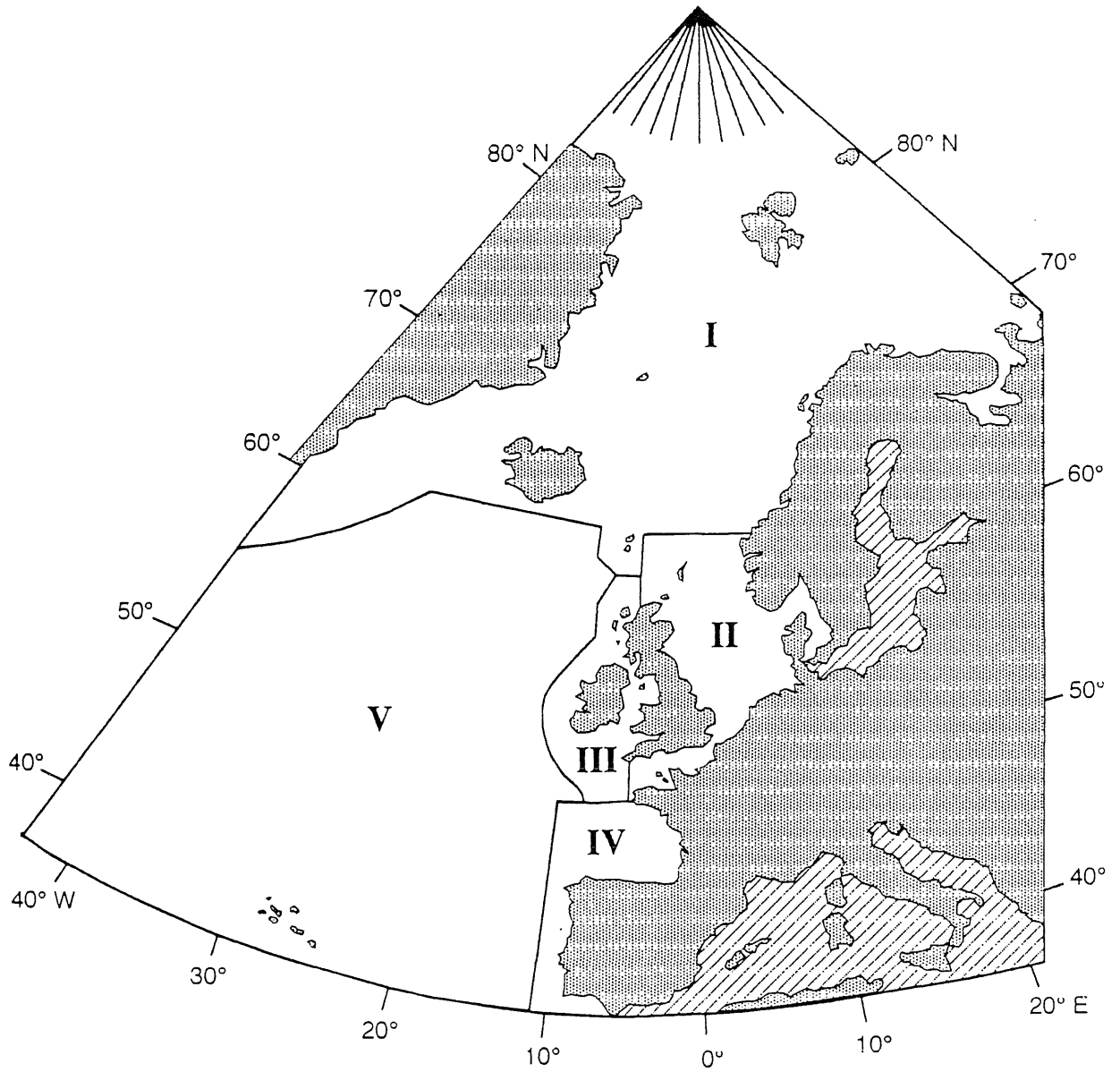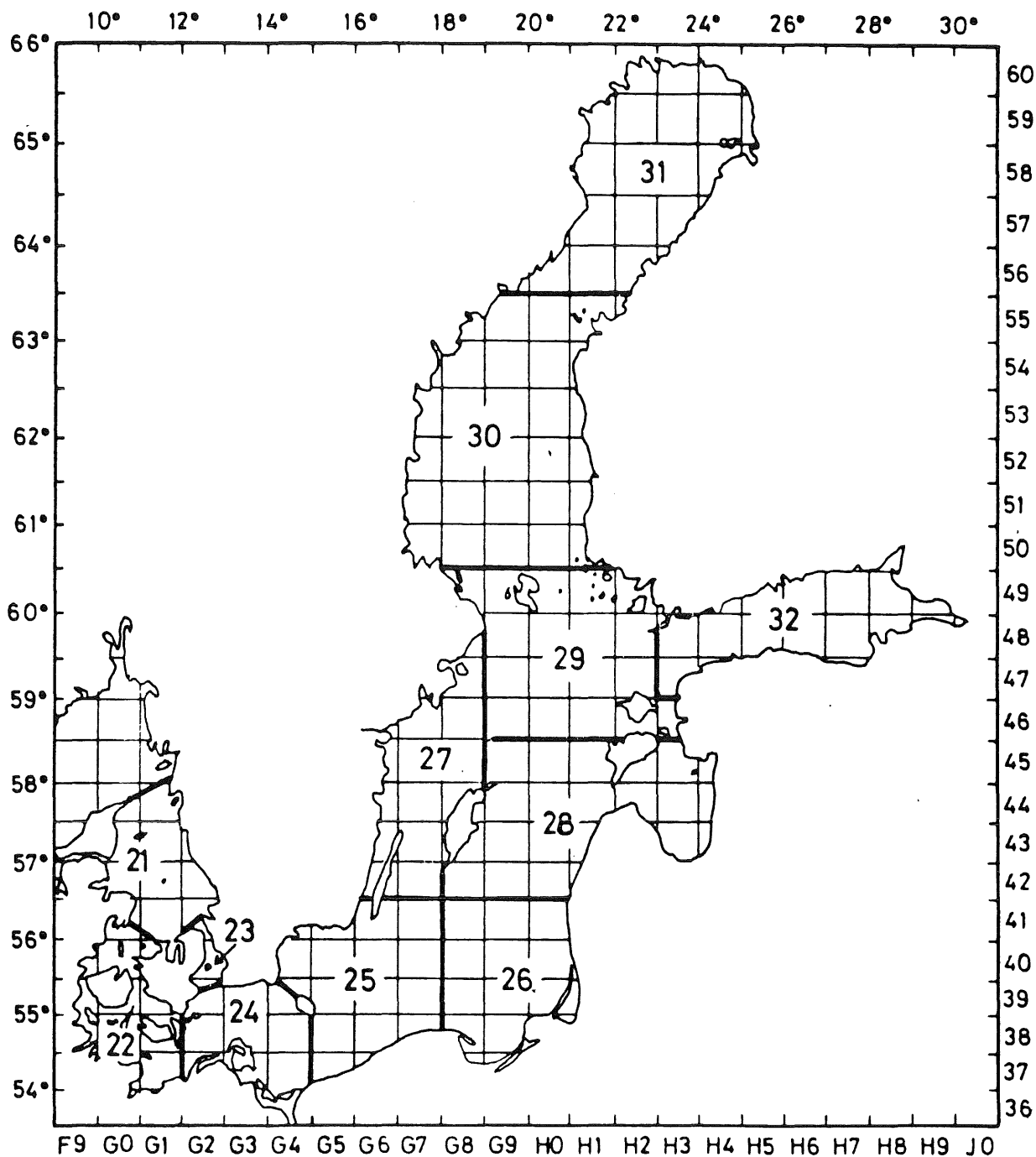
Figure 9.2. ICES Sub-Divisions in the Baltic Sea (ICES, 1987).

ICES   27.3.03.00   (Baltic)

Division IIIa includes Sub-division 21 (plus Skagerrak)
Division IIIb is equivalent to Sub-division 23
Division IIIc is equivalent to Sub-division 22
Division IIId is divided into Sub-divisions 24-32

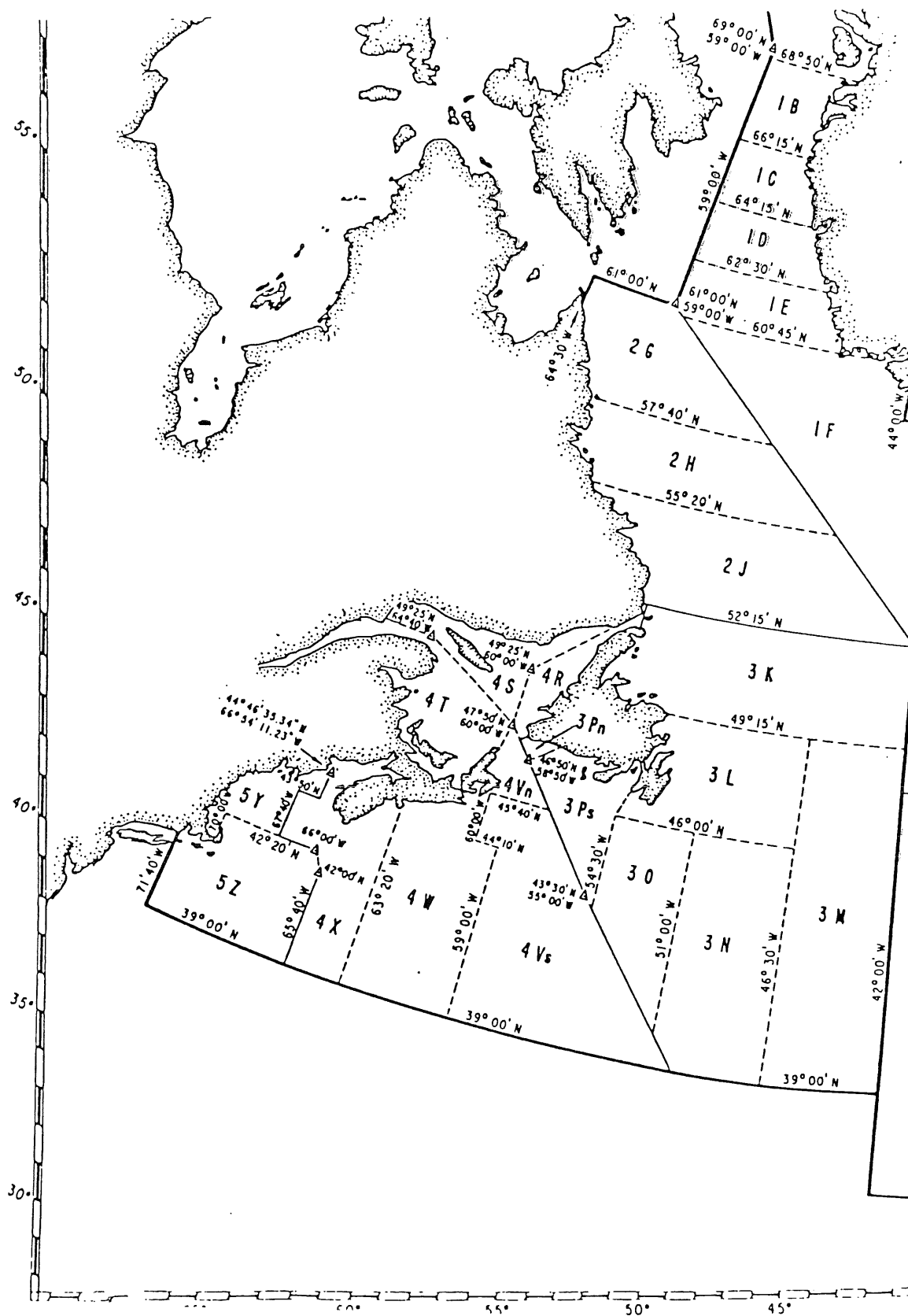**Figure 9.3.a.** ICNAF Divisions of Canadian waters (ICES, 1980).

Figure 9.3.b. Sampling areas of the Northwest Atlantic. Dotted lines indicate NAFO area boundaries (ICES, 1980).
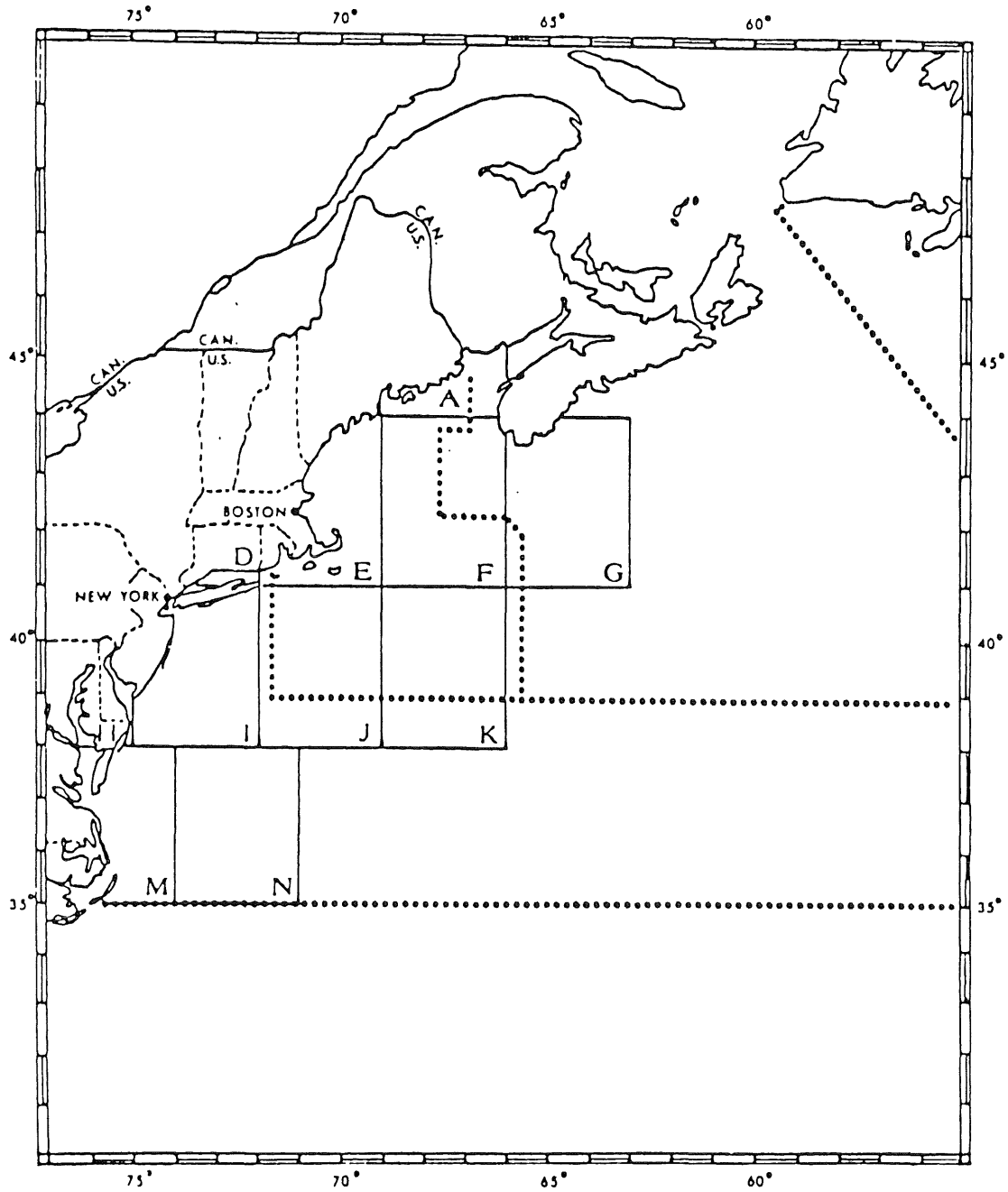
Figure 10.1.1. Huston's (1979) dynamic equilibrium model.
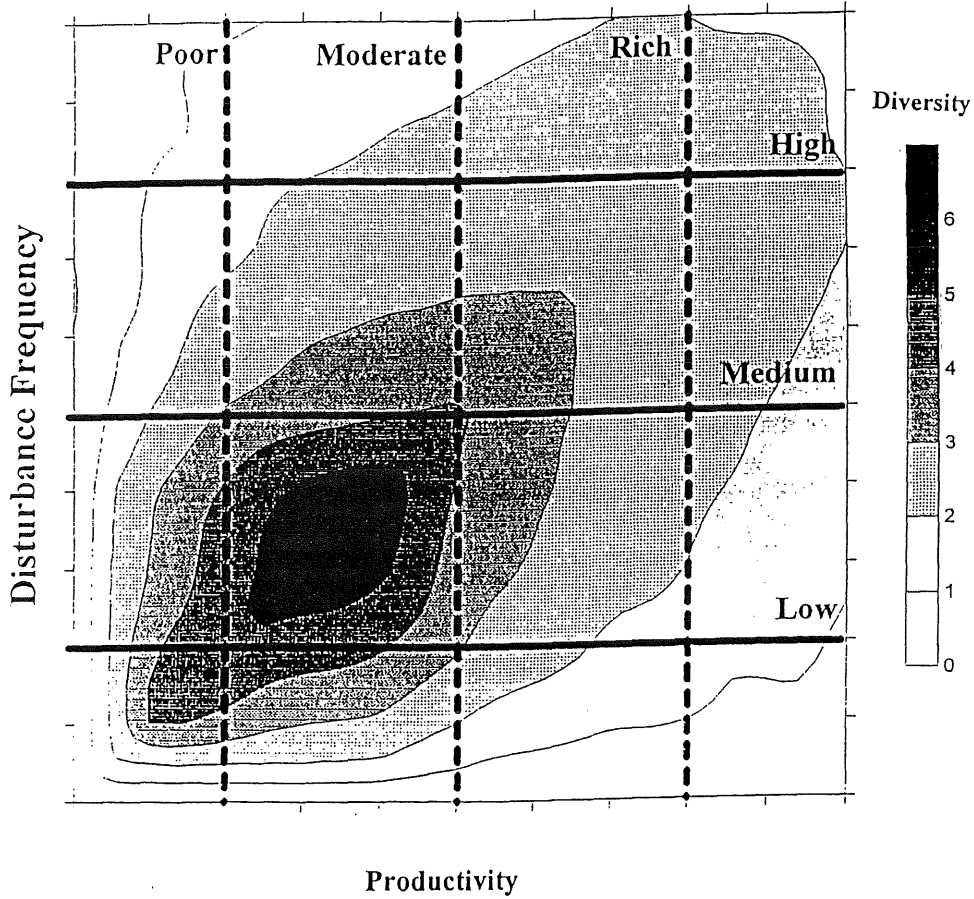

Productivity

Figure 10.1.2. Relationship between species diversity and environmental productivity at three fishing disturbance intensities predicted by the dynamic equilibria model.
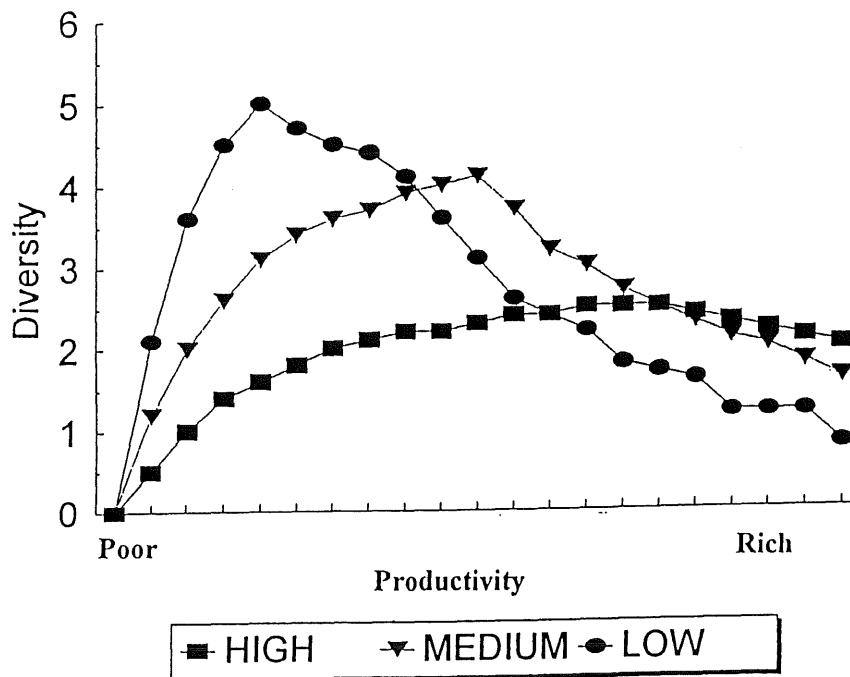
Figure 10.1.3. Relationship between species diversity and fishing disturbance at three levels of environmental productivity predicted by the dynamic equilibria model.
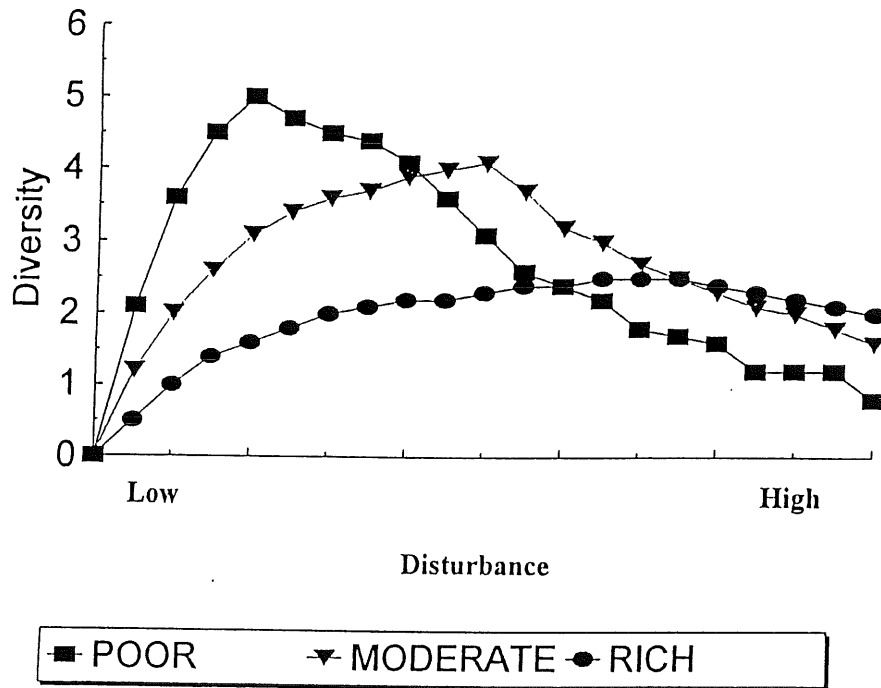

Figure 10.1.4. Effect of disturbance on the mean diversity and the standard deviation estimated over the full productivity range. Low, medium, and high disturbance ranges shown in Figure 10.1.1 and plotted in Figure 10.1.2 are indicated.