REPORT OF THE

WORKING GROUP ON ECOSYSTEM EFFECTS OF FISHING ACTIVITIES

ICES Headquarters, Copenhagen, Denmark
24 November–2 December 1997

International Council for the Exploration of the Sea
Conseil International pour l'Exploration de la Mer

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1 INTRODUCTION

The Working Group on Ecosystem Effects of Fishing Activities (WGECO) met from 24 November through 2 December 1997 at ICES Headquarters in Copenhagen. The meeting was opened at 10.00 hrs by the Chairman, Dr Jake Rice, and was welcomed by the ICES Environment Adviser. Members attending the meeting were:

Fatima Borges	Portugal
Jean Boucher	France
Siegfried Ehrich	Germany
Chris Frid	UK (England)
Henrik Gislason	Denmark
Simon Greenstreet	UK (Scotland)
Sture Hansson	Sweden
Astrid Jarre-Teichmann	Denmark
Knut Korsbrekke	Norway
Ronald Lanters	The Netherlands
Sigbjorn Mehl	Norway
Henn Ojaveer	Estonia
Gerjan Piet	The Netherlands
Stefan Aki Ragnarsson	Iceland
Jake Rice (Chairman)	Canada
Adriaan Rijnsdorp	The Netherlands
Stuart Rogers	UK (England)
Sigmar Arnar Steingrimsson	Iceland
Mark Tasker	UK (Scotland)

The list of participants including contact information is attached as Annex 1.

2 ADOPTION OF AGENDA/TERMS OF REFERENCE

The Terms of Reference for the meeting are listed below.

ICES C.Res.1996/2:15:6 and ICES C.Res.1997/2:12:1

The **Working Group on Ecosystem Effects of Fishing Activities** [WGECO] (Chairman: Dr J. Rice, Canada) will meet at ICES Headquarters from 24 November to 2 December 1997 to:

a) continue to develop the underlying theory on the behaviour of community metrics in relation to changes in fishing activities by:

   i. integrating information on fish assemblages sampled by different North Sea surveys;

   ii. carrying out comparative analyses on fauna assemblages from different ecosystems,

   iii. investigating spatial differences in relation to long-term trends in fishing impact by area and gear;

b) collaborate with MAWG to estimate changes in levels of predation on benthos by fish in relation to changes in exploited North Sea fish species;

c) collate and provide information on the impact of fishing activities on the size distribution/age composition and spatial distribution of the target fish populations of commercially exploited stocks of fish and shellfish (cod, herring, sole, mackerel and hake) in the five OSPAR regions [OSPAR 1998/4.1];

d) collate information on quantities of discards by gear type and OSPAR regions for commercially exploited stocks of fish and shellfish provided by AFWG, WGNPBW, HAWG, NWWG, WGNSSK, WGMHSA, WGNSDS, WGSSDS, WGNAS, WGPAND, WGNEPH, SGDEEP, and SGASSO [OSPAR 1998/4.3];

e) collate and provide information on changes in abundance of individual species of non-target fish owing to fishing activities in the OSPAR regions [OSPAR 1998/4.4];

f) develop and examine potential reference points which might be used for including ecosystem considerations in relation to the precautionary approach;

g) identify and define any requirements to protect local aggregations of sandeels in sensitive areas close to important wildlife assemblages such as seabird colonies;

h) continue development of 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;

i) update the information available to evaluate the effect of sandeel fisheries on local aggregations of sandeels in areas close to important wildlife assemblages such as seabird colonies, and the effects of seasonal and localised catch regulations.

WGECO will report to ACFM and ACME prior to their meetings in May/June 1998 and to the Marine Habitat Committee at the 1998 Annual Science Conference.

WGECO reviewed its Terms of Reference, acknowledging that they were diverse and demanding. It was noted that Terms of Reference c, d, and e were to provide the basis for ICES responses to specific requests from the Oslo and Paris Commissions (OSPAR). These were to be given extra priority to ensure that the response from ICES to OSPAR was as complete and relevant as possible. The ICES Environment Adviser provided useful explanations to a number of questions regarding the intent of these Terms of Reference as well as the fate of the information to be compiled by WGECO. WGECO further noted that although all Terms of Reference were important, d and f were particularly sensitive in that those sections of our report would be scrutinized by a number of different groups and, therefore, it was vital that our presentation be particularly clear and fully qualified, in order to minimize the possibility of misinterpretation. In its deliberations, WGECO concluded that Term of Reference h, regarding the theory of size and diversity spectra, was simply a special case of Term of Reference a, addressing the theory of community metrics in general. Correspondingly, Section 3.4 presents WGECO's response on Term of Reference h, integrated with the rest of the work on this general theme.

The size of the workload referred to WGECO was noted in last year's report (ICES CM 1996/Assess/Env:1). This meeting faced a similar situation, but with fewer participants and shorter meeting time. WGECO agrees that it is the appropriate group to address the issues which have been referred to it, but stresses the need for national institutions to support the work of the group. This support must include active participation of relevant experts at WGECO meetings as well as necessary intersessional work in annual workplans. WGECO is aware that the infrastructure to support the work of even long-established Working Groups has been impacted by changes in funding for national laboratories, making this perhaps a difficult time to support new types of activities. Nonetheless, interest in ecosystem effects of fishing continues to grow, and if credible scientific advice on these matters is to be framed by ICES, new types of work must be supported. This report, like its predecessors, gives some indications of the types of work which are needed. WGECO also notes that a reasonable balance must be struck by including some Terms of Reference with noteworthy scientific content as well as those to compile and provide information if WGECO is to continue to attract top flight scientists to its meetings.

The diversity of workload compounded the problems presented by the size of the workload. The Chairman was blessed with a creative and hard-working membership, covering a variety of disciplines. Nonetheless, many individuals had to work extensively in areas somewhat different from their usual focus of activity. We are confident that this did not reduce the quality of the work in any way, but did increase the time required to complete tasks. Problems posed by the diversity of workload were aggravated by some Terms of Reference whose intent was not completely clear. The issues associated with ecosystem effects of fishing are complex. Good science and good support for advice from WGECO requires that diligence be used in making clear the intent of Terms of Reference. This may, in turn, require more in-depth dialogue with the clients requesting the advice.

The Term of Reference on discards was particularly problematic, and the information provided by the assessment working groups was often unhelpful. The *Nephrops* Assessment Working Group did a thorough job and provided very useful information. Data from other working groups was usually incomplete, ignored published sources as well as programmes coordinated by national laboratories represented at the meetings, or collapsed material in ways which discarded information needed to answer the request from OSPAR. Part of the problem stemmed from the inadequate phrasing of the Term of Reference to the assessment working groups. However, informally it has been made clear to members of WGECO that at least some working groups, as currently functioning, are unlikely to provide improved information on discards in the near future. The assessment working groups must receive much clearer guidance on this topic, including clarification of the importance ICES attaches to the issue. Moreover, individuals from outside the core stock assessment community who are working actively on discard and by-catch issues must be brought into the working groups that address this task. These points and some proactive suggestions for remedying a situation which could be very embarassing to ICES are covered in more depth in Section 6.

As with past WGECO reports, despite the best efforts of all members of the working group, there are undoubtedly studies of relevance to some questions which we did not use in our work. It was particularly frustrating that for some Terms of Reference, important information was not available to the working group, despite efforts by ICES and working group members prior to the meeting, to ensure necessary data sets would be forwarded to ICES. Almost all topics could receive a more detailed treatment, if more people and more time were available before and during the meeting. At the same time, the body of knowledge is in its infancy on many aspects of this large field of ecosystem effects of fishing. Clients and interested readers are assured that this report reflects the consensus scientific judgment of all participants and can, therefore, be used with appropriate confidence. Equally, they are warned that the caveats which are encountered frequently in the report reflect serious concerns by WGECO, with regard to the incompleteness of our sources in some cases, and the state of knowledge in others. These warnings should be considered fully by any users of our work.

The Working Group would like to thank the members of the ICES Secretariat for their usual high level of support and for the friendly environment we experienced. Henrik Sparholt, Mette Bertelsen, and most particularly Melodie Karlson provided many invaluable acts of assistance. The Working Group thanks Henrik Gislason for superior work in arranging our activities outside the meeting. Finally, the Chairman thanks all the members for their exceptionally hard work, with special acknowledgment to the willingness of individuals to take on tasks far outside their normal domain of work, in order to complete all the tasks needed at this meeting.

3 THEORY ON THE BEHAVIOUR OF COMMUNITY METRICS: INTRODUCTORY DISCUSSION

3.1 Integrating Information on North Sea Assemblages from Different Surveys

Long time-series of marine fisheries surveys are one of the most important datasets that we have available, and these can provide invaluable insights into the temporal changes in fish populations. In general, regular surveys become more valuable the longer they are continued, and it is important, therefore, that established surveys are continued. Recently it has become possible to extend some of these series backwards in time using archived data from earlier generations of scientists who did not have access to electronic data storage. Some of these datasets extend back almost to the start of the 20th century, and relate to a period which experienced less extensive fishing impact than the present day.

The purpose of this Term of Reference is to identify all such time-series of data which are collected within the North Sea ecosystem, and which relate specifically to fisheries assemblages. These datasets fall into two distinct categories, those collected during research vessel surveys undertaken by European Research Laboratories, and those collected by other methods, unrelated to fishing surveys or routine market sampling of landings at ports.

3.1.1 Research vessel surveys

The earliest Research Vessel (RV) survey data from the North Sea that have been computerised relate to a series by Dutch and English vessels from 1906 to 1909. These are described by Rijnsdorp *et al.*, 1996, and summary data shown in Table 3.1.1.1. For each haul the numbers of the larger fish species caught are available for 10 cm groups, and some information on the bottom fauna was also recorded. Smaller fish species not considered 'food fish' were not recorded systematically but were sometimes recorded under the heading 'bottom fauna'. The distribution of the fishing stations in the North Sea was fairly uneven, but the southeastern North Sea was well covered by most surveys.

The Scottish August Groundfish Survey (AGFS) has taken place every year since 1980. This survey was undertaken by the RV Explorer until 1982 and then by the RV Scotia. The Scotia is approximately twice as powerful as Explorer and its trawl speed was approximately 1.21 times that of the older vessel (Greenstreet *et al.*, in press), consequently the distance covered in hours' trawling was greater. The 48 foot Aberdeen trawl has been used throughout this survey. This gear is identical to that used by the Marine Laboratory Aberdeen in groundfish survey work extending back to the early 1920s. The data for the entire AGFS, as well as for the months of July to September, are available in electronic format back to 1925.

One of the longest North Sea time-series for demersal species is provided by the first quarter International Bottom Trawl Survey (IBTS), which began in 1960/1961 and has been carried out annually in February since 1965. Initially the target species was herring (hence the initial name of the International Young Herring Survey), and the survey coverage was restricted to the southern and central North Sea, but the coverage was extended when it was realised that the surveys could also provide recruitment indices for cod, haddock, whiting and Norway pout (hence the change to the International Young Fish Survey). Since 1969 the Skagerrak and Kattegat have been sampled and from 1974 the entire North Sea has been included in the survey area. The survey has evolved into a highly standardised, internationally coordinated trawl survey, in which nine countries have been participating (ICES, 1992). Although commercially important species have been the principle target, length data of all the by-catch species have been collected by most participants. The otter trawl gears employed during the series have varied, and over a period of several years up to 1982 have now become standardised on the French designed GOV trawl which has a high vertical net opening. Before this time there was also some inconsistency in the survey area covered annually, which has also been resolved. Data are stored on the ICES IBTS database, but only data collected since 1983 are completely computerised. For the period 1970–1982 the records are incomplete and many data are still in paper format, stored in different laboratories. The contributions of the different countries to the first quarter IBTS database are shown in Table 3.1.1.2.

The need to monitor flatfish stocks in the heavy beam trawl fishery in the shallow coastal waters of the southern North Sea led to the introduction of fisheries independent surveys using beam trawls. By 1988 a number of countries which border the North Sea had developed these surveys, and these targeted different age ranges of flatfish and used beam trawls and vessels of different size and specification. Collation and analysis of some of the data derived from these surveys was initially focused on the North Sea and eastern Channel, but during the early 1990s all surveys in subareas IV and VII were included (ICES, 1991). During the 1980s, five countries which border the North Sea and western waters of the UK had developed a range of beam trawl surveys (Table 3.1.1.3). Some of these surveys were designed to sample pre-recruit (0- and 1-group) sole and plaice on nursery grounds with light gears, while others used beam trawls of commercial design to catch juveniles and adults. Six of these surveys were modified following recommendations of the Beam Trawl Study Group to develop a more standardised sampling protocol (ICES, 1994).

In addition to these surveys already included in the activities of the Beam Trawl Study Group, there are others which have only recently been transferred to electronic format. The Sole Net Survey (SNS) was initiated in 1969 to obtain pre-recruit indices for 1- and 2-group plaice and sole. The survey consists of 10 transects parallel or perpendicular to the continental North Sea coast between the Dutch/Belgian border and Esbjerg in Denmark, and a number of fixed stations is fished on each transect. The DFS was initiated in 1970 to obtain pre-recruit indices of brown shrimps and 0- and 1-group plaice and sole. For this survey two types of gear are used, a single 3 m beam trawl in the Wadden Sea and Scheldt estuaries, and a pair of 6 m beam trawls along the Dutch coast. Three areas were distinguished: DFS1 the Scheldt estuary, DFS2 along the Dutch coast and DFS3 the Wadden Sea estuary.

A coastal nursery ground survey has also been operating in the coastal waters (<20 m) on the east and south coast of England since the 1970s, and this survey uses the 2m beam trawl and 1.5m push net (Rogers and Millner, 1996). These two gears were specifically designed to have similar efficiency and selectivity so that the catches could be directly compared.

3.1.2 Other sampling methods

A variety of other sampling techniques have been used to collect time-series data on the abundance of North Sea fish. Examples in this category mainly include the use of fixed stations to collect fish on a regular basis. One example of this technique has been described by Phillipart *et al.*, (1996), in which the fish of the Dutch Wadden Sea have been collected from a tidal inlet using a kom-fyke trap. This gear, operating since 1960, is emptied every day from Monday to Friday, and operates throughout the summer period. A similar series of data have been provided by the catches of fish which impinge on the filter screens of power station cooling water intakes (Henderson, 1989). For both these examples of fixed station recording, there is no information on the relationship of the catches with the total population abundance, but for power stations at least, the fish catch is thought to be highly effective at sampling a wide range of demersal and pelagic species.

3.1.3 Problems with combining gear catches

For a number of reasons, different fishing gears vary in catch efficiency for different sizes of fish, and this is the main problem encountered when comparing catch data collected between one survey and another. In those cases where catches have been combined, the swept area of each gear has often been used (Rijnsdorp *et al.*, 1996). For beam trawls this is the fixed width of the trawl opening, but for otter trawls this parameter varies with the water depth and speed of towing. As all fishing gears are selective and the catchabilities of fish at size vary, this standardisation to the swept area of the gear does not resolve all the problems, and relative catchabilities can only be obtained when all gears are fished simultaneously on the same ground.

The selectivity and catchability of a demersal trawl is influenced by the way that the net is rigged, the type of ground gear, the length of the towing warp and otter trawl sweeps, the mesh size in the cod end and the speed at which the gear is towed. In addition, the ground over which the gear is towed and the tidal conditions during towing will also influence catch rates of fish. Gear parameters are most variable for otter trawls, where for the same gear, headline height varies with tow speed and depth of fishing, and catch rates are influenced by the length of the sweeps (ICES, 1996). The best way to ensure that fish catch rates from different surveys can be combined is to use identical gears operated in precisely the same manner. This situation rarely occurs, however, and a number of studies in the North Sea have attempted to get conversion factors between vessels and gear by undertaking comparative fishing trials. The gear used during the IBTS first quarter surveys, the GOV otter trawl, is recommended to have sweep lengths of 60 m for fishing in shallow areas and 110 m at stations deeper than 70 m to avoid possible changes in gear parameters due to depth and to the length of the warp. Comparative gear trials conducted in 1994 using warps of two different lengths concluded that catch rates of cod, haddock, whiting and herring were different, and that for some species, particularly herring, the catch rate at size also varied.

It is possible that beam trawls of the same design but of different widths may not show a linear relationship in their catch rates of all demersal species, and that the use of different attachments (chain mat, flip-up ropes, etc.) will also affect the gear efficiency. Comparative fishing exercises (Groeneveld and Rijnsdorp, 1990) compared the 4 m beam trawl with chain mat and flip-up ropes, and the 8 m beam trawl with tickler chains and flip-up ropes. During surveys in 1990 and 1991, catch ratios of dab, sole and plaice between the two gears were consistently different (ICES, 1993), suggesting that it was not possible to derive raising factors to convert the catch numbers of one gear into that of another gear.

When considering long time series, particularly those covering periods of 20 years or more, it is important that the fishing characteristics of the different RVs that may have been used in the collection of the data are taken account of. Analyses that depend on absolute numbers of each species sampled may well be affected by differences in the areas swept by the fishing gear as a result of vessels of differing horse power towing the gear at different speeds. Some species diversity indices, for example, are particularly sensitive to variation in sampling effort. In the 70-year time series (Greenstreet and Hall, 1996), four different RVs were involved. The area swept by the standard fishing gear used varied by a factor of approximately 1.89 from the most to the least powerful vessel.

A further related issue involves trawl tow duration. During the 1970s and 1980s many groundfish surveys used standard one hour trawl durations. Recently some Institutes have reduced this to half hour tows causing potential problems with the analysis of longterm trends in species diversity. Furthermore, other Institutes have continued to trawl for one hour making contemporary comparisons difficult. The issue of sample size dependence of some community metrics is particularly relevant when it comes to considering the effect of variation in trawl duration.

A final consideration in comparing different datasets, again related to the problems of sampling effort dependence, are the possible consequences of the protocols used for handing catches once they are brought aboard the vessel. It is frequently impossible to sort and handle every single fish in a large catch. Subsampling is necessary. Straight forward proportional division of the catch, sorting one fraction and discarding the rest effectively reduces sampling effort at that station, and it reduces the probability of finding rare fish. In biodiversity studies it is important that not only is the haul duration standardised, but that the entire catch is sorted in such a way as to obtain a reasonably accurate estimate of even the rarest species

These examples of comparative gear trials suggest that the levels of standardisation currently used in the IBTS Database are important to ensure that catch data are collected in a similar way, and that catch comparisons between gears are important. They also illustrate how difficult it is to combine catches from similar gears. The relative catchability of different species by different gears is an important consideration in deciding which species to include in the species-suite in a particular analysis. To combine the catch rates of fish between, for example, the otter trawl catches of the IBTS, and the beam trawl catches of the beam trawl surveys in the North Sea, will require extensive species by species knowledge of the selectivity of each gear. These data are not yet available.

Catch efficiencies of species caught by the International Beam Trawl Surveys, for example, have been assumed to be in direct proportion to the width of the trawl. It has been necessary to make this assumption for these gears, in order to prepare a spatially extensive dataset and develop our undersanding of the spatial dynamics of demersal species in the North east Atlantic. In this example, corrections between surveys are possible for tow duration and gear width, but more complex corrections for area swept are required for otter trawl gears. These decisions, however, are only partly based on scientific evidence and also include an element of judgement which is based on the experience of scientists who are familiar with the operation of the surveys and the properties of the gears. The selectivity of the gears to target species is also an important consideration. It is clear that more research is needed in order to ensure that these judgements are based on more sound scientific arguments.

3.1.4 References

Groeneveld, K., and Rijnsdorp, A.D. 1990 The effect of a flip-up rope on the catch efficiency of an 8-m beam trawl. ICES CM 1990/B:16, p. 13.

Henderson, P. 1989 On the structure of the inshore fish community of England and Wales. Journal of the Marine Biological Association, UK, 69: 145–163.

ICES. 1991 Report of the Study Group on Beam Trawl Surveys in 1990. ICES CM 1991/G:81, p. 57.

ICES 1992 Manual for International Bottom Trawl Surveys. Revision IV. ICES CM 1992/H: 3/Addendum.

ICES. 1993 Report of the Study Group on Beam Trawl Surveys. ICES CM 1991/G:5, p. 92.

ICES. 1994 Report of the Study Group on beam trawl surveys in the North Sea and Eastern Channel. ICES CM 1990/G:59.

ICES. 1996. Report of the International Bottom Trawl Survey Working Group. ICES CM 1996/H:1.

Philippart, C.J.M., Lindeboom, H.J., Meer, J. van der, Veer, H. W. van der., Witte, J.I. 1996 Long-term fluctuations in fish recruit abundance in the western Wadden Sea in relation to variation in the marine environment. ICES Journal of Marine Science, 53: 1120–1129.

Rijnsdorp, A.D., Leeuwen, P.I., Daan, N., and Heessen, H.J.L. 1996. Changes in abundance of demersal fish species in the North Sea between 1906–1909 and 1990–1995. ICES Journal of Marine Science, 53: 1054–1062.

Rogers, S.I., and Millner, R.S. 1996. Factors affecting the annual abundance and regional distribution of English inshore demersal fish populaitons: 1973 to 1995. ICES Journal of Marine Science, 53: 1094–1112.

3.2 Theory of Community Metrics - Multivariate Indices and Analyses of Communities

In this section we review our current knowledge regarding changes in marine fish assemblages which could be attributed to variation in fishing practice. The Terms of Reference ask us to first consider studies carried out within the North Sea, then to take account of studies carried out in other ecosystems, and finally to consider how examination of spatial differences in the behaviour of different community metrics might help to establish fishing pressure as the cause of observed changes. We therefore present our information in three sections, each section dealing with one of these themes.

In each section we first start by summarizing the Working Papers submitted for consideration by the Working Group. The full text of each paper is available from ICES, and so only a summary of the materials and methods used, and the key points from the results and discussion sections of the papers, have been included here. The analytical techniques applied in the various studies differ considerably. In some instances this makes comparisons between studies somewhat problematic, however we try to compare the implications of the various sets of results. We go on to present analysis of data available to the Working Group using a standard suite of community metrics, in an effort to make direct comparisons more simple. A considerable amount of groundfish survey data, collected by different institutes, using different fishing vessels and fishing gear, and covering a wide range of geographic locations and marine habitats, is now available and we believe that a more integrated approach to their analysis would be profitable.

3.2.1 North Sea region

3.2.1.1 Review of current information

3.2.1.1.1 Monitoring of changes in small-scale fish assemblages in the North Sea (S. Ehrich and C. Stransky, Working Paper)

Bottom trawl surveys were carried out on RV Walter Herwig III (using the standard GOV trawl) and RV Solea (with a similar, but smaller otter trawl) in eight boxes in the North Sea during summer-time from 1987–1996 (Figure 3.2.1.1.1.1). The towing time was 30 minutes with a speed of about 4 knots. Trawling positions and directions within the boxes were randomly selected. In general, at least 20 trawl hauls were made in 3 days within each box.

The catch data were standardised for one hour. The standardised abundance data were used to calculate a diversity index for each box and year. An MDS analysis was performed on the species composition similarity matrix (using a Bray-Curtis index on root-root transformed data) to compare species composition within and between the boxes. The boxes sampled with different vessels were analysed separately.

The boxes with the greatest geographical distance (e.g., A and D, F and H) show the least similarity in fish fauna. The boxes situated more closely to each other (e.g., B and C, E and K) were more similar (Figures 3.2.1.1.1.2 and 3.2.1.1.1.3). No obvious trend in species composition similarity over the years within the boxes was apparent, suggesting little in the way of long-term change in species composition in each box. The number of species caught in each box was relatively constant over the study period, however, variability in the Hill's N1 was high but showed no trend.

3.2.1.1.2 Long-term changes in North Sea fish assemblages based on different beam trawl surveys (G. Piet and A.D. Rijnsdorp, Working Paper)

Five datasets describing demersal fish abundance, sampled using beam trawl surveys, have been described. These surveys covered the Dutch coastal zone, the Wadden Sea estuary, the Scheldt estuary, the coastal SNS and BTS survey. The surveys differ in the gear type used and the geographical area covered. Because of these changes the DFS is more suited to studying changes in smaller fish, whereas the SNS and the BTS are relatively more affected by changes in larger fish.

Analysis of the surveys shows considerable changes in both size structure and species composition. All surveys showed, to a greater or lesser degree, a general trend in the size structure together with year-to-year fluctuations. The general trend was a shift in the relative distribution of biomass towards the smaller size classes probably caused by the effect of fisheries exploitation in the region. The year-to year fluctuations in size structure were to a large extent caused by differences in year class strength of the most abundant species, such as dab and plaice.

The species composition was also affected by changes observed in size structure in combination with life history characteristics of the different fish species. For the SNS survey it was shown that the abundance of species with a large size at maturity decreased while that of species with a small size at maturity increased.

3.2.1.2 Analyses carried out by WGECO

3.2.1.2.1 Dutch Beam Trawl Survey data

The SNS was initiated in 1969 and is a national survey carried out only by The Netherlands. It is aimed at obtaining pre-recruit indices for 1- and 2-group plaice and sole. The survey is carried out using a 6 m beam trawl, rigged with 4 tickler chains and a sole net with a mesh size of 40 mm in the cod-end. The standard station grid of the survey consists of 10 transects parallel or perpendicular to the continental North Sea coast between the Dutch/Belgian border and Esbjerg, Denmark. On each transect a number of fixed stations is fished. There is no further stratification. In total, 55 hauls are made each year, with at least 4 hauls in a transect. The gear is fished with a fishing speed of 3.5 knots and the haul duration is 15 minutes. Three areas were distinguished: SNS1 south of Texel, SNS2 between Texel up to the German Bight, and SNS3 north of the German Bight.

The DFS was initiated in 1970. It aims at obtaining pre-recruit indices of brown shrimps and 0- and 1-group plaice and sole. For this survey two types of gear are used. In the Wadden Sea and Scheldt estuaries a single 3 m beam trawl is used. The gear is rigged with a shrimp net of mesh size 20 mm in the cod-end and one light tickler chain. A ground rope with wooden or rubber bobbins is used. Along the Dutch coast, fishing is done with a pair of 6 m beam trawls. The gear is rigged with a shrimp net in a similar way as the 3 m beam trawl. Fishing speed is 2–3 knots, depending on the strength of the current. Three areas were distinguished: DFS1 the Scheldt estuary, DFS2 along the Dutch coast, and DFS3 the Wadden Sea estuary.

Temporal trends in Hill's N1 and N2 are shown for both data sets in Figures 3.2.1.2.1.1 and 3.2.1.2.1.2. Some interesting trends are apparent. Both indices show species diversity to be consistently higher in SNS1 compared with SNS3. Hill's N1 suggests that diversity in SNS2 tracks diversity in SNS3 at the start of the time series, flips in 1976 to track SNS1 until 1989, then tracks SNS3 for the next two years, before reverting to tracking SNS1. Species diversity, as sampled by the DFS, appears to fluctuate widely. No consistent temporal trend is suggested by either index. However, between-area variation is high from 1969 to 1977, whereupon species diversity in the different areas appears to converge for 10 years or so, before once again diverging towards the end of the time series.

3.2.2 Other oceans and seas

3.2.2.1 Review of current information

3.2.2.1.1 Comparing diversity of coastal demersal fish faunas in the North-East Atlantic (S.I. Rogers *et al.*, Working Paper)

International Beam Trawl Survey data were used to analyse the assemblage structure of commercially important and non-target demersal fish species collected from coastal waters of the Northeast Atlantic (Figure 3.2.2.1.1.1). Catches were dominated by a small number of species, which occurred in large numbers and at high biomass. The most abundant species (plaice, dab) were typical of shallow, uniform sandy and muddy seabed which occurs extensively throughout the southern North Sea and to a limited extent in western UK waters. Renyi's diversity index family was used to rank the diversity of coastal sectors throughout the region. The limited access of the southern North Sea to species-rich southern faunas and the uniform nature of the seabed were largely responsible for the lower diversity of North Sea coastal faunas compared to those in the Channel and west of the UK. West of the Dover Strait, the more heterogeneous substrate supported a more diverse fauna of smaller-sized fish with the occurrence of southern species such as red gurnard and thickback sole and an increasing abundance of elasmobranchs. Patterns in community structure over such a wide spatial scale and without historical perspective can be explained by biogeographic factors, seabed structure, and the influence of regional hydrography. Inferring from these patterns an impact by anthropogenic factors (towed gears) is unlikely to be achieved.

3.2.2.1.2 Spatial patterns of groundfish assemblages on the continental shelf of Portugal (M.C. Gomes and E. Serrao, Working Paper)

Five groundfish surveys (four in autumn and one in spring) were conducted off Portugal from 1985–1989 by RV Norugea using the Norwegian Campell Trawl (horizontal opening 14 m, vertical opening 4 m, groundrope with rollers, cod-end mesh size 40 mm). Sampling was performed at randomly selected stations on longitudinally and latitudinally determined 36 depth strata (at least two stations per depth strata) along the coast of Portugal (depth range 20–500m) during the day. Trawl duration was 30 min and the tow speed about 3 knots. Fish were identified, in general, to the species level. Any species that comprised at least 1 % of the total biomass in at least one of the surveys was included in the analysis. The catch data were log-transformed before further analysis. The stations were compared by pairs using the Bray-Curtis dissimilarity index and clustered using group average hierarchical agglomerative cluster analysis.

Based on cluster analysis, the following depth groups of stations were separated (Figure 3.2.2.1.2.1) as follows.

Shallow-Northern Group (20–100 m). The following species dominated:

sardine (*Sardina pilchardus*), mackerel (*Scomber scombrus*), horse mackerel (*Trachurus trachurus*), and European squid (*Loligo vulgaris*). Other commonly found species were: *Merluccius merluccius, Trisopterus* spp., and *Polybius henslwoi*.

Shallow-Southern Group (20–100 m). Horse mackerel and axillary seabream *Pagellus acarne* made usually over 50 % of the total biomass. Other species usually occurring in catches were *M. merluccius, S. pilchardus, L. vulgaris*, and seabreams.

Intermediate Group (80–180 m). *M. merluccius, S. pilchardus, L. vulgaris*, and *T. trachurus* occurred most often whereas *M. merluccius* dominated in the catches.

Deep-Northern Group. Blue whiting (*Micromesistus poutassou*) made up the majority of the catches with *M. merluccius* composing the reminder of catches.

Deep-Southern Group. The biomass was dominated by blue whiting (*M. poutassou*) with the following fishes occurring in significant amounts: boarfish (*Capros aper*), *M. merluccius*, and some crustaceans.

These five clusters of stations allowed the mapping of groundfish assemblage areas on the Portuguese Shelf (Figure 3.2.2.1.2.2), i.e., the areas characterized by a relatively homogenous groundfish composition. Major changes in the composition of the demersal community off the Portuguese Shelf are associated with depth. The change is especially sharp at depths between 100–200 m, where separation of Deep and Shallow groups takes place. *M. poutassou* dominated in the catches from the deep region (150–400 m) whereas *S. pilchardus, T. trachurus*, and *S. scombrus* were the majority in shallow areas (20–120 m). The second major biogeographic transition occurs in near-shore waters (< 120 m depth). The relative proportion of *S. scombrus* and *Trisopterus* spp. decreases and that of *M. merluccius, L. vulgaris*, and sparids in catches increases.

3.2.2.1.3 Spatial distribution of species assemblages in the Celtic sea and the Bay of Biscay (J.C. Poulard and J. Boucher, Working paper)

Two bottom trawl surveys were carried out in the Bay of Biscay and the Celtic Sea shelves and upper slopes in autumn 1990 and spring 1991 by using the GOV 36/47 trawl with 20 mm cod-end mesh size and of estimated headline height and distance between wings of 4 and 18–20 m, respectively. Trawling was carried out during day-time with a speed of 4 knots and duration of 30 minutes. In the Bay of Biscay, the survey area was divided, according to latitude, into 4 blocks and stratified sampling was performed in the following depth ranges: 15–30, 31–80, 81–120, 121–160, 161–200, 201–400 and 401–600). 137 and 142 hauls were made in 1990 and 1991, respectively. In the Celtic Sea, sampling was performed at fixed stations (grid length 25 nautical miles). 56 and 57 hauls were made in 1990 and 1991, respectively.

The total weight of the catch and abundance of fish species in a catch was recorded. In 1990, only selected fish species were measured whereas in 1991, this was performed for all species caught. Log-transformed catch data of fish species were classified by applying a hierarchical ascending classification procedure to their first PCA coordinates and the groups obtained were then clustered by using a moving centres procedure.

Six types of fish assemblages could be identified within the study area (Figures 3.2.2.1.3.1 and 3.2.2.1.3.2), as described below.

<u>Fish assemblage of the central shelf of the Bay of Biscay</u> (mean depth: 100–112 m, muddy bottoms prevail). Characteristic fish species: *Lesueriogobius friesii, Merluccius merluccius, Cepola rubescens, Nephrops norvegicus, Arnoglossus laterna*, and cephalopods from genus *Alloteuthis*.

<u>Fish assemblage of the western shelf</u> (mean depth: 151–152 m, mostly sandy bottoms). Characteristic species are: *Lepidorhombus whiffiagonis, Capros aper, Todaropsis eblanae*, and *M. merluccius*. Seasonally characteristic species are: *Illex coindeti, Argentina sphyraena*, and *Callionymus maculatus* in autumn and *Pollachius virens* in spring.

<u>Fish assemblage of the Continental slope</u> (mean depth: 310–351 m, hard bottoms dominate in some areas). Characteristic species: *Malacocephalus laevis, Chimaera monstrosa, Galeus melastomus, Helicolenus dactylopterus*, and *Lepidorhombus boscii*.

<u>Fish assemblage of the southern Celtic shelf</u> (mean depth: 143–148 m, mostly coarse sand bottom). *Aspitrigla cuculus, Arnoglossus imperialis*, and *Raja naevus*.

<u>Fish assemblage of the northeastern Celtic shelf</u> (mean depth: 115 m, soft bottom type). Characteristic species: *Trisopterus esmarki, Gadus morhua, Merlangius merlangus, melanogrammus aeglefinus, Hippoglossoides platessoides limandoides, Eutrigla gurnardus, Glyptocephalus cynoglossus*, and *Pleuronectes platessa*. In addition, *Squalus acanthias* is typical in autumn and *Clupea harengus* in spring.

<u>Shallow water fish assemblage of the Bay Biscay</u> (mean depth: 39–47 m). The highest number of species has been recorded in this assemblage. Characteristic species: *Sardina pilchardus, Trachurus mediterraneus, Scomber scombrus, Engraulis encrasicolus, Sprattus sprattus, Merlangius merlangus, Ammodytes tobianus, Hyperolpus lanceolatus, Spondyliosoma cantharus, Dicentrarchus labrax, Callionymus lyra, Dicologlossa cuneata, Solea vulgaris, Echiichthys vipera, Trachinus draco, Loligo vulgaris, Sephia officinalis*, and *Crangon crangon*.

3.2.2.1.4 Analysis of the spatial and temporal variability of the size spectrum of the fish community in the Bay of Biscay, 1987–1995 (J.C. Poulard and J. Boucher, Working Paper)

The data used in this analysis were gathered from seven bottom trawl surveys with RV Evehoe using GOV 36/47 trawl from 1987–1995. Until 1989, sampling was performed by the following scheme: 100 hauls were made at fixed stations and 35 hauls at changeable stations. From 1989 onwards, all hauls were performed at fixed locations (ICES, 1991, 1997).

The number of trawlings by depth ranges and years in the Bay of Biscay are presented below.

Stratum	1	2	3	4	5	6	7	
Depth (m)	*< 31 m*	*31–80*	*81–120*	*121–160*	*161–200*	*201–400*	*> 400*	*Total*
1987	14	21	30	33	14	9	9	130
1988	14	23	26	38	15	11	7	134
1989	15	24	27	40	17	13	6	142
1990	16	21	28	38	18	10	6	137
1992	14	22	26	25	11	5	4	107
1994	11	18	22	25	12	8	5	101
1995	14	18	22	25	18	10	7	114

Fish size spectra were constructed for each depth range and year by summing catch numbers over species within 5 cm size classes ranging from 20–24 cm to 75–79 cm. The size spectrum for the whole Bay of Biscay is the mean of depth strata spectra weighted by the numbers of hauls in each respective depth strata. For the long-term data analysis, a set of fish species was selected for analysis over all years. All treatments were performed separately for all species measured during the study period and for the subset of demersal species.

For data analysis, fish abundance data and size-class categories were log-transformed. Analysis on long-term trends of slopes and intercepts of the size spectra were performed as outlined by ICES (1996). The survey data were also disaggregated by depth strata. To study the covariance of fish abundance by size spectra, the following models were applied:

$y = \mu + \beta x + \alpha_i + \beta_i + \gamma_j + \beta_j x + \delta_{ij} + \varepsilon$ (1) and $y = \mu + \beta x + \alpha_i + \gamma_j + \delta_{ij} + \beta_{ij} x + \varepsilon$ (2), where

y - fish abundance
x - fish size-class
β- slope of the size spectra
α_i - depth effect
γ_j - year effect
δ_{ij} - interaction term of the year and the depth effect
β_{ij} - slope of the size spectra by depth strata and years
μ - the general mean term
ε - the error term

When both demersal and pelagic species were treated together, regressions between fish abundance and size-class were significant for each year. Compared to those in the North Sea, the slopes in the Bay of Biscay were lower (ranging from –6.2 to –3.4 and –7.4 to –6.1, respectively) and the intercepts were higher (13.4–19.0 and 25.1–29.8, respectively). No significant long-term trend in slopes or intercepts was found.

For demersal species only, slopes and intercepts were better determined than for all species, mainly due to better fitting of the linear model. The slopes were considerably higher and intercepts lower with significant long-term trends in both parameters (Figure 3.2.2.1.4.1). However, the conclusions of the analysis of disaggregated data did not change when the pelagic species were removed from the analysis.

The analysis showed that all the effects incorporated into the models were highly significant. The size-class term accounted for the largest amount of variance of the fish abundance whereas the class variables (depth and year) accounted for a smaller, but still significant part of the variation (Table 3.2.2.1.4.1).

3.2.2.1.5 Application of experimental trawl data for estimation of fish stock dynamics in the Gulf of Riga (H. Ojaveer, Working Paper)

Species richness and fish abundance dynamics, including currently non-assessed and non-target species, were monitored during 1974–1986 and 1994–1996 by using catch per unit effort data from monthly experimental bottom trawl surveys (Figure 3.2.2.1.5.1). The trawls were carried out in daylight with a mean trawling speed of 2.5 knots, the towing duration was 30 minutes, estimated trawl opening area was 40 m and mesh size in the codend was 8 mm (from May to July 1994; 20 mm). In 1981–1986, surveys were conducted only in autumn (1–3 surveys per September–November). For further analysis, the basin was divided into two regions: shallow coastal area (Pärnu Bay, strongly dominated by freshwater fish) and deeper parts of the basin (> 20 m, dominated by marine species and glacial relicts). Abundance of all species in a catch was determined through direct counts or through sub-sampling if the catch was too large. Except for sticklebacks (*Gasterosteus aculeatus* and *Pungitius pungitius*) and gobies (*Pomatoschistus* spp.), the fish were identified to the species level.

The abundance data were analysed by the following GLM model:

log (mean catch+1) = Year + Month + ε,

where mean catch is the monthly average catch by number of a species;

Year and Month - the year-effect and month-effect, respectively; ε - the error term. The year-effect in the model was used as an index describing dynamics of fish stock abundance. For estimation of species richness, the following GLM model was applied:

log (mean number of fish species) = Year + Month + ε,

where mean number of fish species is monthly average number of fish species in a catch per area.

Abundance dynamics of fish species inhabiting mostly the shallow region

This category includes all the freshwater species living in the basin and also certain euryhaline species. With a single exception (gobies), fish of this category have shown an increase in stock size (e.g., sticklebacks) or no clear tendency in the abundance estimates is evident (Figure 3.2.2.1.5.2).

Abundance dynamics of fish species living in deeper areas

This category includes marine boreal species and glacial relicts. Two general tendencies in the stock abundance of these species could be pointed out:

- Generally higher abundance in the late 1970s, lower values during the 1980s, and recent increase in stock size (e.g., sprat *Sprattus sprattus*, smelt *Osmerus eperlanus*, eelpout, *Zoarces viviparus*) (Figure 3.2.2.1.5.2).
- Obvious decrease in the abundance or extinction of some species, which were rather abundant during the late 1970s, from the community of the Gulf of Riga from the mid 1980s (e.g., cod *Gadus morhua callarias*, common sandeel *Ammodytes tobianus*, fourhorned sculpin *Triglopsis quadricornis*).

Species richness

In the shallow region, the model estimates indicate, with certain exceptions from 1983–1985, a slight increase in the mean number of fish species. Whereas an obvious decline in this characteristic was evident in deeper areas during 1978–1985, followed by an increase in 1994–1996 (Figure 3.2.2.1.5.3). The increase was caused by an elevated frequency in the occurrence of pelagic euryhaline (sprat *Sprattus sprattus* and sticklebacks) and some cold-water species (smelt *Osmerus eperlanus* and eelpout *Zoarces viviparus*) while other demersal and cold-water fishes, (e.g., sea snail *Liparis liparis* and fourhorned sculpin *Triglopsis quadricornis*), found relatively frequently in the 1970s, were absent or only rarely present in the hauls.

These changes probably reflect different responses of fish species from those different groups (freshwater species, marine fish, and glacial relicts) to alterations in the main abiotic and biotic parameters of the basin (due to natural causes and anthropogenic activities), but also affected by stock (over) exploitation and the presence or absence of the only large marine predator in the ecosystem, cod.

3.2.2.2 Analysis carried out by the Working Group

3.2.2.2.1 Gulf of Riga

Dynamics of three community metrics indices (species richness, Hill's N1, and Hill's N2) were investigated in the Gulf of Riga (Baltic Sea) during 1974–1986 and 1994–1996. The survey data and its collection methods are described above.

For estimation of species richness, the following GLM model was applied:

log (mean number of fish species) = Year + Month + ε,

where mean number of fish species is monthly average number of fish species in a catch per area; Year and Month - the year-effect and month-effect, respectively; and ε - the error term. As only the autumn period was sampled during all the study years, for Hill's N1 and Hill's N2 calculations, fish abundance data from this period was used.

In the shallow region the model estimates indicate, with certain exceptions in 1983–985, a slight increase in the mean number of fish species.This is not the case in the deeper areas during 1978–1985 which showed an increase in 1994–1996 (Figure 3.2.2.2.1.1). The last increase was caused by elevated frequency in the occurrence of pelagic euryhaline (sprat *Sprattus sprattus* and sticklebacks) and some cold-water species (smelt *Osmerus eperlanus* and eelpout *Zoarces viviparus*) while other demersal and cold-water fish (e.g., sea snail *Liparis liparis* and fourhorned sculpin *Triglopsis quadricornis*), found relatively frequently in the 1970s, were absent or only rarely present in the hauls. It seems likely that most of these changes were mainly governed by alterations in environmental conditions and predation by cod rather than the direct effect of fishing.

Long-term dynamics of Hill's N1 and Hill's N2 indices suggest the following patterns of the two spatially separated fish communities in the Gulf of Riga:

1) The fish community in the shallow area is, in general, more heterogeneous than that in deeper areas with no clear trend in the indices calculated;
2) The fish community in the deep area exhibits slight increasing tendency in the heterogeneity measures over the years studied.

3.2.2.2.2 Barents Sea bottom trawl survey

The bottom trawl survey data used for the Barents Sea analysis comes from a combined acoustic and bottom trawl survey for demersal fish in the Barents Sea which has been conducted annually since 1981. Only data from 1985–1996 are used. The survey methodology is described in Dalen *et al.* (1982), Hylen *et al.* (1986) and Jakobsen *et al.* (1997). The sampling trawl used is the Campelen 1800 shrimp trawl with 80 mm mesh size in the front. Until 1989 the trawl was equipped with rubber bobbin but in 1989 a rockhopper ground gear was introduced. This improved the catch efficiency of the trawl (especially the smaller gadoids). This change in ground gear is likely to show up in several of the analyses. The survey area was increased in 1993 but only data from the central regions covered in all years are used in the analysis. The survey area with subareas and strata system, together with the trawl stations taken in 1996, are shown in figure 3.2.2.2.2.1. Note that only data from subareas A, B, C, and D are used in the analysis.

Both diversity indices fluctuated throughout the time series, each index tracking the other. A slight negative trend in species diversity was apparent (Figure 3.2.2.2.2.2). This was corroborated by variation in k-dominance curves calculated over three four-year periods, combining data over years in each period. Dominance was greatest in the period 1993–1996, and least in 1985–1988 (Figure 3.2.2.2.2.3). Species evenness showed a decline over the course of the time series (Figure 3.2.2.2.2.4), but little trend in species richness was apparent (Figure 3.2.2.2.2.5). Changes in the relative abundance of the most abundant species accounted for these trends, rather than any change in the number of species in the assemblage.

Size spectra were examined over two size ranges, 20–50 cm (Figure 3.2.2.2.2.6) and 50–100 cm (Figure 3.2.2.2.2.7). Trends in the slope are shown in Figure 3.2.2.2.2.8. The slope for 20–50 cm fish went through two ossilatory cycles, being least negative in 1988 and 1995. The trend in slope for fish 50–100 cm in length followed an opposite cyclical trend, being most negative in these years and least negative in 1992. This analysis was repeated, this time excluding cod and haddock (Figures 3.2.2.2.2.9 and 3.2.2.2.2.10, respectively). This had the effect of damping the second cycle (Figure 3.2.2.2.2.11).

Variations in N1, N2, and species richness were examined in area D separately and compared with trends for areas A, B, and C combined (Figures 3.2.2.2.2.12 to 3.2.2.2.2.14). Species diversity was generally lowest in area D and this could be explained by the presence of fewer species in this area.

3.2.3 Spatial patterns and the relationship with fishing

3.2.3.1 Review of current information

3.2.3.1.1 Changes in the groundfish species assemblage of the northwestern North Sea between 1925 and 1996 (S.P. Greenstreet, F. Spence, and J.A. McMillan, Working Paper)

This study examined long-term changes in the structure and composition of the groundfish species assemblage in four regions of the northwestern North Sea (Figure 3.2.3.1.1.1). Scottish fisheries research vessel data primarily collected during from July to September during 1925 to 1996 were analysed. Trends in the whole groundfish assemblage and in a subset of the assemblage, which is not targeted by commercial fisheries, were described. These trends are then related to variations in the patterns of fishing activity in each of the areas.

Species diversity in the whole groundfish assemblage had declined in the three areas where fishing pressure had been greatest; in the area where fishing pressure had been least historically, no trend in species diversity was detected (Figure 3.2.3.1.1.2). Only in the area where fishing pressure had been the highest, and at high levels for the longest period of time (Figure 3.2.3.1.1.3), was a negative trend in species diversity observed among the non-target species assemblage. Spatial variation in species diversity was clearly defined. Within the whole groundfish assemblage, diversity was greatest in the inshore and southern regions and least in the offshore northern area, while among the non-target species assemblage, the spatial diversity gradient was reversed.

Multivariate analyses indicated long-term changes and between-area differences in the species composition of both the whole groundfish assemblage and the non-target species subset (Figure 3.2.3.1.1.4). However, these changes consisted for the most part of subtle variations in the relative and absolute abundance of a few key species rather than involving major species replacement events. Only one species showed any marked increase in abundance and this was a case of a dominant species becoming even more abundant.

Examination of species-aggregated length frequency distributions suggested a shift over time towards assemblages more dominated by smaller fish. This was mainly apparent, however, in the whole groundfish species assemblage; the length frequency distributions of non-targeted species were much more stable.

3.2.3.2 Analyses carried out by the Working Group

3.2.3.2.1 Monitoring fish assemblages in small defined areas in the North Sea

The survey design, the position of the boxes (Figure 3.2.1.1.1.1) and some results are already described in Section 3.2.1.1.1. The 8 boxes are distributed over the entire North Sea and cover a depth range from 110 m (Box D) to less than 40 m (Box A) in the German Bight. The boxes are situated in areas where the main fishing gear used and the degree of fishing effort (hours fished) differ considerably. Fishing effort distribution in 1991 was used to calculate the mean annual effort within the ICES rectangles which are touched by the boxes. Effort data for the German fishing fleet and of the STCF data set (international effort without German data) were combined to estimate total international effort. The boxes can be separated into 5 categories: Box A represents an area of high fishing effort, mainly by beam trawl; boxes B and D belong to areas of medium fishing effort using otter trawls; boxes E and F represent areas of moderate fishing effort by beam trawlers; box C belongs to an area of low fishing effort conducted mainly by beam trawlers; and box H represents an area of low fishing effort mainly by otter trawlers (Figure 3.2.3.2.1.1).

Pelagic species such as herring, sprat, mackerel, and horse mackerel, can dominate the species composition since, when they occur, they can occur in very high densities. Under such circumstances diversity indices, such as Hill's N1 and N2, decrease to very low values. To make the results comparable with other papers investigating changes in fish assemblages in the North Sea (Greenstreet and Hall, 1996) the calculations were done including and excluding these 4 pelagic species.

The annual changes in the diversity (Hill's N1 and N2 indices) within the boxes are shown in Figures 3.2.3.2.1.2 to 3.2.3.2.1.9. As expected, there was a general shift to a lower range when the four pelagic species were excluded, but no trends in any of the 8 boxes were indicated by either index whether the pelagic species were excluded or not. Even in Box A, situated just outside the plaice box, and where beam and otter trawl effort has increased by a factor of two from 1982 to 1993 (de Groot *et al.*, 1995), no trend in species diversity was noted.

Changes in mean body weight were investigated in the heavily fished Box A and the less fished Box C to look for evidence that the size spectrum of the exploited fish assemblage had shifted towards the smaller sized end of the size spectrum (e.g., Rice *et al.*, 1996). The pelagic species were excluded from the calculation. Length data were not available at the meeting so variation in the mean individual weight was examined instead (Figure 3.2.3.2.1.10). Apart from the high value in 1992 in Box A related to unusually high immigration of one-year old cod into the German Bight, mean individual weight has declined in both boxes with little difference between boxes, suggesting little effect of fishing effort.

Investigations of the fish assemblage in a small box of 100 nm² in an area of high fishing pressure over the last decade revealed no changes in the abundance of species like dab and grey gurnard, related to changes in fishing effort (Draft IMPACT-II Report). The data presented here also indicate the difficulties in detecting fishing-related changes in fish species assemblages. Neither the diversity metrics nor the mean individual weight revealed any temporal trends associated with increasing fishing effort. Nor did comparisons between boxes differing in the intensity to which they were fished show differences.

3.2.4 Concluding comments and discussion on multivariate metrics

There are two main features of the spatial analyses described here which are worthy of further comment. The first is the extensive amount of survey data that are available to research institutes in Europe collected using a wide range of vessels, gears, stratification methods, sampling accuracy, and seasonal and spatial resolution. Within a study these various factors are adjusted and modified to optimise the sampling programme within a region, often to conform with the national requirements for obtaining stock abundance estimates. This degree of national variation does not, however, help us to combine datasets on a larger spatial scale and in a general international perspective on assemblage structure. The varied and serious issues associated with combining datasets have already been described in detail in Section 3.1. The second important feature of the studies described here is the wide range of statistical analyses that have been applied to these datasets. Among the list of routine statistics that have been described here are Principal Components Analysis, k-dominance statistics, similarity and dissimilarity coefficients, and agglomerative clustering. More novel techniques such as size spectrum analysis (Section 3.3.2), phylogentic relatedness analysis (Section 10.1), and the use of diversity profiling (see Section 3.3.2) has also been applied.

In summary, while the analyses of each regional assemblage is informative and valuable, difficulties in combining these datasets, and the large number of different analyses applied to them, suggests that finding and describing a common theme or themes will be a challenge. For this reason, comparative analyses were carried out on a selection of datasets which were considered to be the most comparable, using a range of simple and widely used community metrics. The results of some preliminary analyses have been described. The covariation in both the N1 and N2 metrics was evident in many of the analyses.This is largely because the fish assemblages studied are all strongly dominance oriented, a fact underlined where k-dominance curves were applied. These were often the most informative metric revealing changes in species relative abundance and they are amenable to statistical comparison. In fact, ranked abundance biomass comparison curves (ABC) where both the cumulative biomass and abundance of ranked species are plotted may well be even more informative. This conveys information about shifts in size and/or weight of the dominant species as well as changes in their numerical contribution to the total assemblage. This may be particularly pertinent where fish assemblages are concerned, since we have a sound theoretical basis to expect fishing to shift assemblages towards the lower end of the size spectrum.

As demonstrated here, examination of trends in the value of particular metrics over time are frequently inconclusive. A major problem here is the short time scale of most datasets (one or two decades) compared with the time over which the North Sea and other marine ecosystems have been fished (one or two centuries). It should be remembered that changes in the populations of exploited species (abundance, age structure) are most apparent when fishing mortality is actually increasing (or decreasing). In periods of relatively constant fishing mortality (albeit high) a stable population structure develops. Over most of the last two decades many exploited species have undergone more or less constant high fishing mortality, giving rise to relatively stable (even if overfished) populations during the period when most groundfish assemblages have been sampled. The most obvious changes will have occured when populations of the exploited species themselves were undergoing the greatest change, that is, when fishing mortality was increasing relatively fast from zero, well before the start of most surveys.

The spatial analyses presented here also suggest a second confounding influence, the powerful effects of abiotic and biotic characteristics of the environment in determining the species composition and structure of fish assemblages. This suggests that temporal variation in environmental parameters may have marked effects on the assemblages under investigation, and these may mask any trends arising from variations in fishing practices. However, careful analysis of spatially referenced fish species abundance data, which includes information regarding spatial variation in abiotic and biotic features of the environment as well as quantitative measures of fishing activity, may well help to unravel the different effects of these interacting influences.

3.2.5 References

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3.3 Theory and Performance of Metrics of Community Properties from Mass-balance Models

3.3.1 Case studies using mass-balance models to compare the trophic structure of ecosystems - pelagic upwelling systems

3.3.1.1 Datasets description

A comparison of mass-balance models of trophic flows in the four large marine upwelling areas (Jarre-Teichmann, 1998) was presented to the group as a working paper in order to illustrate the potential use of this class of models for intersystem comparisons. For this study, the Ecopath II software (Christensen and Pauly, 1992) was used to balance models for different species dominance regimes in the upwelling systems off northern-central Peru (1964–1971 and 1973–1981), California (1965–1972 and 1977–1985), northwest Africa (1972–1979) and Namibia (1971–1977 and 1978–1983). The data for these models had largely been assembled from published literature and/or reports, and the models were built in strictly the same way such as to allow for intersystem comparisons. The study has been expanded since to include the southern Benguela ecosystem as well.

3.3.1.2 Description of the modelling and analysis methodology

3.3.1.2.1 Construction of the models

Assuming mass-balance over an appropriate period of time, the production of each component of an ecosystem (e.g., a sub-population, species or a group of species) is balanced by its predation by other components in the system (predation mortality), its exports from the system (fishing mortality and other exports), and the baseline mortality. Thus,

Production by (i) = All predation on (i) + nonpredatory biomass losses of (i)

+ fishery catches of (i) + other exports of (i)

The terms in this equation may be replaced by

Production by (i) = $B_i \cdot P/B_i$

Predatory losses of (i) = $\Sigma_j (B_j \cdot Q/B_j \cdot DC_{j,i})$

Other losses of (i) = $(1 - EE_i) \cdot B_i \cdot P/B_i$

For any component in the system, this leads to the linear equation

$$B_i \cdot P/B_i \cdot EE_i - \Sigma_j (B_j \cdot Q/B_j \cdot DC_{j,i}) - Ex_i = 0$$

where

- i indicates a component (stock, species, species group) of the model,
- j any of the predators of i,
- B_i the biomass of i,
- P/B_i the production i per unit of its biomass (= total mortality under steady-state conditions),
- Q/B_i the consumption of a component per unit of its biomass,
- $DC_{j,i}$ the average fraction of i in the diet of j (in terms of mass),
- EE_i the ecotrophic efficiency of i (the fraction of the total production consumed by predators or exported from the system),
- Ex_i the export of i from the system (e.g., by emigration, or fishery catch).

The energy balance of each component is given by

Consumption = Production + Respiration + Non-assimilated food

wherein consumption is composed of consumption within the system and consumption of imports (i.e., consumption 'outside the system'), and production may be consumed by predators, exported from the system, or be a contribution to detritus.

This structure defines the necessary parameters for the model. For each component, an estimate of its biomass, P/B and Q/B ratios, diet composition, its exports from the system, and its assimilation and ecotrophic efficiencies are required. However, for each component one of the parameters B, P/B, Q/B or ecotrophic efficiency may be unknown, because it is estimated when solving the system of linear equations, along with the respiration of that component. The model is regarded as balanced when realistic estimates of the missing parameters have been achieved for all components of the ecosystem.

Analysis of the models

After a model has been balanced, it is assured that the various estimates of biomass and turnover rates are mutually compatible, and hence represent a possible and consistent picture of the energy flows in the system. Only after this process has been completed is it meaningful to perform further analyses of the model, e.g., for interactions between its components and/or the role of the fishery. A rich theoretical framework exists for the analysis of energy flows or cycling in ecosystems, notably building on the theories of Odum (1969) and Ulanowicz (1986).

Direct trophic interactions, i.e., predation and fishery, can straightforwardly be assessed by analysing partial mortality coefficients of the prey (or target) groups, and by calculating trophic levels. An additional assessment of indirect trophic interactions, e.g., competition, is possible by mixed trophic impact analysis (Ulanowicz and Puccia, 1990). This approach assesses the relative impact that the change in biomass of a given group would have on the biomass of the other groups in the system. The method is, however, based on the assumption that its trophic structure does not change. Consequently, it is not possible to use it for predictions, but instead as a sensitivity analysis of the cascading effects of changes in an ecosystem's food web.

The partitioning of trophic flows among different consumer groups in an ecosystem can further be illustrative of the role of these consumers in a system, and of their development over time. While fish usually take the largest fraction of fish production (e.g., Bax, 1991; Jarre *et al.*, 1991), the fishery is often the second largest consumer, and often in direct competition with marine mammals.

The fisheries in different ecosystems cannot readily be compared based on their total catch alone, because the species composition of the catch can be rather different. This is in part a result of the specific oceanographic and biological conditions that determine the distribution of a species, but also a result of both fishery management (selection of target species) and fishing practice (selection of fishing gear). Fish are situated at different levels in the food web of an ecosystem, and trophic pathways of different length are therefore required to sustain them. Therefore, the exploitation of fish on lower trophic levels is less expensive in ecological terms than the exploitation of fish on higher trophic levels, and a common currency is needed to compare the ecological cost of fishing among different time periods or systems. Primary production equivalents, as suggested by Pauly and Christensen (1995), are one possibility. Following their approach, a particular end flow in question (e.g., the fishery catch of a species) is traced backwards through the food web, using the ratios of production and consumption of the various components along the path as magnification factors. The sum of the flows leading from the basis of each path (i.e., from the producers" level) to the end flow in question is then the total primary production needed to sustain it.

3.3.1.3 Results

The results of the study indicated that the four upwelling systems ranked rather distinctly after the size parameters primary production, total biomass sustained in the system, fishery catches, and total system throughput (Figure 3.3.1.3.1). They were set apart in geographical rather than in regime-specific (or temporal) order, although considerable changes in energy flows occurred in some of the systems. Mixed trophic impact analysis showed the importance of primary and secondary production, but also the competition of predatory fish with the fishery, and top-down control aspects like the inhibition of semipelagic fish such as hake through the fishery (Figure 3.3.1.3.2). The fishery took 20-30 % of the production of the five dominant species anchovy, sardine, mackerel, horse mackerel and hake in all systems except off California where fishing moratoria applied for part of the period under investigation (Figure 3.3.1.3.3). In this system, a comparatively large fraction of the fish production was consumed by top-predators which are valued more highly by the tourism industry than in the other upwelling areas.

The analysis of primary production required to sustain the fishery (Figure 3.3.1.3.4) reflected changes in the fishing strategy in systems over time. In the Peruvian system, where the magnitude of the catches was reduced by a factor of more than three between the two periods investigated, the primary production required to sustain the fishery decreased only by 10 % as the fishery inceasingly targeted hake, a predatory fish, in the later period. The fishery thus remained just as costly in ecologic terms as it had been during the peak period of anchovy exploitation in the 1960s.

3.3.1.4 Discussion

Fisheries-oriented construction and analysis of trophic models bear the advantage that (i) they are relatively straightforward to construct (ii) a trophic flow diagram allows to put the commercially exploited species (and the fishery) into the entire ecosystem, giving an immediate visual impact of the trophic flows in the system, (iii) a whole toolkit of established methods of network analysis is available to assess, e.g., indirect trophic interactions and top-down control processes, (iv) flows can straightforwardly compared between different periods in the same system, or among similarly structured systems.

The comparison of upwelling systems showed, among other results, that the systems are not only driven by food availability as repeatedly suggested, but a number of top-down control mechanisms exists. The posistion of the small pelagic species inthe food web, the low transfer efficiency betweeen trophic levels, and the mixed trophic impact of the lowe trophic levels appeared to be rather global properties. By identifying similarities between ecosystems, experiences in their fisheries management could consequently become transferable.

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3.3.2 Case studies using mass-balance models to compare the trophic structure of ecosystems - application to the Baltic Sea - 1900 to the Present

3.3.2.1 Description of data

The Baltic Sea is a comparatively young, brackish, boreal ecosystem the coasts of which have been inhabited by people for a long time. Primary productivity increased markedly during the past 90 years due to eutrophication. This has also been the reason for higher fish biomass at present than around 1900, total fish biomass has approximately increased by a factor of three (Thurow, 1997). Catches in the Baltic Sea increased about ten-fold in that period. Marine mammals (three species of seals plus harbour porpoise) were abundant at the beginnng of this century, but have almost vanished now due to excessive hunting.

As a case study, the seasonal mass-balance models of carbon flows in the central Baltic Sea of Jarre-Teichmann (1995), which were based on Elmgren (1984), Wulff and Ulanowicz (1989) and ICES (1996), were re-arranged to give and annual average of trophic flows at the end of the 1980s, when cod biomass in the Baltic was very low. Food composition of mysids was updated based on Hansson *et al.* (1997). This model of recent trophic flows was compared to a model of carbon flows in the Baltic around 1900, which constructed during the working group based on Elmgren (1989) and Thurow (1997). The ECOPATH software, explained in Section 3.3.1 of this report, was used for balancing and analysing the models. The results of the comparisons are given in in Table 3.3.2.1 and in Figure 3.3.2.1.

3.3.2.2 Results and discussion

Odum's (1969) theory divides measures of ecosystem maturity into five groups, community structure and energetics, life history, nurtrient recycling, selection pressure and system homeostasis. Slightly rearranged, measures of (i) community structure, (ii) structure of the food web, (iii) nutrient cycling and (iv) system homeostasis were addressed for this case study (Table 3.3.2.1.). A high community production/respiration ratio indicates a rather immature system sensu Odum. Production per unit biomass is relatively high although it has decreased during the past 90 years. More biomass is supported per unit energy flow at present, but the fraction is still relatively low. The same holds true for the total biomass in the system. These indicators of community structure point at a rather immature system, as should be exected from the relatively young age in the Baltic.

Connectance index and system omnivory index, both metrices of the diversity of trophic flows in the model, indicate that trophic niches became narrower during the past 90 years, a result well in line with Elmgren's (1989) description of lost food chains due to bottom anoxia in the deeper parts of the Baltic Sea. The loss of the marine mammals a top predators (and their exploitation) resulted in the loss of at least one level in the trophic flow pyramid. However, the average path length in the system remained approximately the same, as cod took over the former role of the mammals. The transfer efficiency of flows between trophic levels increased, probably an indicator of stress.

Three metrices of food we structure indicate that the food web stayed approximately the same apart from the loss of some food chains and the mammals as top predators. This is largely due to unchanged flow patterns at the lower trophic levels. To what extent this is an artefact of model construction can at present not be decided.

Cycling, nutrient regeneration and the role of detritus in nutrient regeneration all indicate a loss in maturity sensu Odum, which is explained by increased primary production (less food limitation) on the other hand but on the other hand stress to pollution. The increasing oxygen depletion in the Baltic in periods of stagnation (no inflow of high saline, oxygen-rich water from the North Sea) has been extensively discussed, and this discussion need not be repeated here. However, its effects appear to show in the system summary metrices derived from this relatively simple model.

Summarizing, the seemingly contradictory results from the metrices pertaining to community structure and nutrient recycling, respectively, may be explained in the following way: energy throughput in the Baltic Sea has increased due to eutrophication, making the system 'larger', and, with the loss of homoiotherm predators, the average organism size has increased along with the increase of fish biomass. However, the additional nutrients appear not to be worked up as well as before, leading to increased detritus accumulation (by slightly more than a factor of two) and thus, through increased areas suffering from oxygen depletion, to relatively decreased recycling of nutrients.

The metric that according to our present knowledge becomes closest as an indicator for system stability, system overhead on exports (calculated following Ulanowicz, 1986) is slightly higher that for large upwelling systems, and lies in the lower range of other shelf ecosystems. The increasing factor of mutual information (Ulanowicz, 1986) points at increasing certainty about the destination of a given unit of flow in the system, corresponding to the lower connectance and system omnivory indices. The metrices of system homeostasis thus indicate a shelf-like system which is more vulnerable than typical shelf ecosystems.

The primary production required to sustain the fishery catches increased from 5 % at the turn of the century (this already includes severe hunting for mammals) to 15 % at the end of the 1980s. Although the fishery as a total is probably sustainable in the Baltic, this is only the case because herring and sprat are comparatively lightly fished. Cod, on the other hand, is outside of safe biological limits (ICES, 1997). An assessment of the total ecological cost of the fishery in a system can therefore not replace the assessment of its impact by species, but it can indicated its general compatibility with the flows in the ecosystem.

While mammals consumed about 35 % of the total fish production around 1900, their consumption is now lower than 1 %. The fishery took slightly less than 11 % of the total fish production at the turn of the century, this fraction increased to 36 % at the end of the 1980s. Consumption of fish by fish has been relatively constant, with approximately 44 % of the total fish production. 9 % and 19 % of fish production were directed to other sinks in the two periods, respectively.

Assuming the same diet composition for mammals as used at the beginning of the century, there would at present be enough food to sustain slightly less than half of their biomass at the turn of the century, i.e., 3 mg C m^{-2}. The fish production which would be available to mammals appears at present not to be directly consumed in the system, but to enter the detrital food chain. The observation that seal population in the Baltic are presently increasing at a high rate supports the assumption of availabe food in the system. However, it is without doubt that the present level of fishing overlaps with the food requirements for mammals at their historic population size. Which of the two forms of consumption in the ecosystem is to be preferred is necessarily a choice of society, balancing, e.g., cultural preferences, economic returns and a commitment to sustain biodiversity. Whichever the choice, it must be ensured that habitat requirements beyond food supply are also met.

The results of this case study need to be viewed with caution, as the model around 1900 pertains to the entire Baltic Sea, while the model of the late 1980s was constructed for the central Baltic (ICES SDs 25, 26, 28, 29), excluding the western Baltic, Gulf of Riga, the Bothnian Bay, and the Gulf of Finnland. However, we believe that the trend which emerged here is correct, as it is the more vulnerable areas of the Baltic which were excluded from the more recent model, while the bulk of the fish production has always taken place in the central Baltic Sea. Furthermore, the balancing of the models can only be regarded as preliminary due to time constraints.

The study also showed that the Baltic was far from an unexploited system around 1900, and a considerable further step backwards in time may be required to arrive at a system which was not subject to major anthropogenic influence.

3.3.2.3 Metrices addressing the impact of fishing in this case study

Fishing practices in the Baltic have changed substantially from the beginning of this century. Not only have the catches increased by one order of magnitude, but at the turn of the century a coastal fishery existed which was largely directed towards herring, in combination with seal hunting. At present, the herring fishery continues, but removals of sprat and particularly cod have increased by factors of about 25.

There are three metrices which in the framework of a mass-balance model directly address the impact of a fishery in the ecosystem. The trophic level of the fishery puts the fishery into the ecosystem as a predator, and the models show that the fishery continues to be the top predator in the Baltic Sea. The transfer efficiency between trophic levels, which increased during the past 90 years, reflects the increased productivity (= mortality) of the exploited fish species. The primary production required to sustain the fishery reflects the increased ecological cost of fishing, taking into account the position of the targeted species in the food web (as discussed in connection with comparative modelling of upwelling systems in an earlier section of this report). The increase of the ecological cost of fishing by a factor of three, agree well with the observed removal of total fish production.

Direct trophic interactions, i.e., predation and fishery, can straightforwardly be assessed by analysing partial mortality coefficients of the prey (or target) groups. In addition, mixed trophic impact analysis allows to assess the indirect trophic interactions, taking into account, e.g., competition of predator groups for prey. The mixed trophic impact of the fishery changed markedly during the past 90 years. Whereas at the turn of the century mammals were strongly impacted by hunting, cod, herring and sprat were only inhibited very lightly. The mixed trophic impact of the fishery on herring increased (in the negative way) by a factor of 5 from the turn of the century to present. A slight inhibition of sprat at the turn of the century turned to a slight favoring (by inhibiting its competitor and predator at present), consistent with observed trends of increasing biomasses in the Baltic. Cod are at present strongly inhibited by the fishery, which shows through an increase of the mixed trophic impact index by an order of magnitude in the negative way.

3.3.2.4 References

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Table 3.3.2.1. Results of the comparison of trophic flows in the Baltic Sea around 1900 and in the late 1980s.

Metric/Period	ca. 1900	ca. 1990
Overview		
Primary production (g $cm^{-2}yr^{-1}$)	79	160
Sprat production (g $cm^{-2}yr^{-1}$)	0.013	0.258
Herring production (g $cm^{-2}yr^{-1}$)	0.112	0.351
Cod production (g $cm^{-2}yr^{-1}$)	0.016	0.085
Fishery catches (g $cm^{-2}yr^{-1}$)	0.027	0.295
Mammal consumption (g $Cm^{-2}yr^{-1}$)	0.087	<0.001
Trophic level of the fishery ()	4.30	4.36
Community structure		
Community P/R ()	1.60	1.69
Community P/B (yr^{-1})	37.3	16.8
Biomass supported by unit energy flow (g $cm^{-2}yr^{-1}$)	0.010	0.017
Net community production (g $cm^{-2}yr^{-1}$)	43.5	62.6
Total organic matter sustained (g cm^{-2})	3.54	9.51
Food web structure		
Connectance index	0.209	0.189
System omnivory index	0.137	0.108
Average path length	3.41	
No. of discrete trophic levels	>8	7
Transfer efficiency between trophic levels	9.3 %	12.4 %
Nutrient regeneration		
Finn's cycling index (%)	12.6	12.0
Nutrient regeneration (overhead on exports) (%)	4.2	3.6
Role of detritus in nutrient regeneration (%)	59	50
Residence time (B/(R+Ex)		
System homeostasis		
Stability (% system overhead)	71.0	73.0
Information content of flows ()	1.24	1.89
Primary production required to sustain fishery catches (%)	4.7	14.8
Mixed trophic impact analysis of the fishery		
... impacting sprat	-0.018	0.008
... impacting herring	-0.002	-0.014
... impacting cod	-0.032	-0.275

3.4 Theory on size and diversity spectra

3.4.1 Size spectra

Studies of the amount of biomass in various size categories has suggested that the logarithm of the biomass in log weight intervals should be approximately constant (Sheldon *et al.*, 1972). Recognizing that the amount of biomass in each size category would depend on the width of the size categories, Platt and Denman (1978) standardized the spectrum by using log biomass divided by the width of the interval on the abscissa and found the standardized spectrum to have a slightly negative slope. Models have been developed which explain the slope and intercepts of the biomass spectrum as a function of the energy-transfer between adjacent trophic levels (e.g., Borgman, 1987; Thiebaux and Dickie, 1993). These models have later been used to predict fish biomass in lakes, but with variable success (Cyr and Peters, 1996).

Fisheries biologists have studied abundance-size rather than biomass-size relations. Pope and Knights (1982) compared the size composition of demersal fish caught by bottom trawl surveys in the North Sea and at the Faroe Islands and found that a straight line fitted log numbers per size class *versus* size in both cases. Subsequent comparisons of size spectra from various parts of the world have confirmed that log numbers per size group often are linearly related to the size of the fish (Pope *et al.*, 1987; Murawski and Idoine, 1992; Gobert, 1994). They have also suggested that the slope of this relationship could be related to fishing intensity. Spectra from areas subjected to different fishing intensities have thus shown that the slope of the size spectra is steeper in heavily fished areas than in less fished areas. The slope is more negative in the heavily fished North Sea than it is in Faroe waters and on the Georges Bank (Pope and Knights, 1982;

Murawski and Idoine, 1992). Haedrich and Barnes (1997) linked decreases in biomass and numbers as well as decreases in the mean size of target and non-target species of fish on the northeastern Newfoundland and Labrador shelf to increases in fishing effort. Similar changes in mean size over time was found in analyses of survey data from West Greenland. In the last report of this Working Group, the analysis of data from two independent bottom trawl surveys in the North Sea revealed an increase in the intercept and a decrease of the slope with time.

In the Bay of Biscay, however, Rochet *et al.* (Working Paper) found no significant time trend in neither slope nor intercept of the size spectrum of pelagic and demersal species of fish caught from 1987–1995. When considering only demersal species, a significant increasing trend in the slope and a significant decreasing trend in the intercept was found, but the interpretation was somewhat confounded by significant year/depth strata interactions.

A Working Paper by Piet and Rijnsdorp, in which the species and size composition of the fish catch obtained in five beam trawl surveys in the southern North Sea were analysed, revealed a general increase in the proportion of the smaller size classes of fish in the catch. The authors attributed this increase to a decrease in predation leading to an increase in the number of small fish. Their findings suggest that the fish assemblage has changed from a top-down regulated assemblage towards a bottom-up regulated assemblage in which increased competition could limit the growth rate of the smaller fish. Although alternative explanations are possible, this interpretation is in accordance with the decrease in weight at age of small plaice and the increase in growth of larger plaice (> 35 cm) found by Rijnsdorp and van Leewen (1996).

Section 3 of this report provides additional examples of spatial and temporal changes in size spectra within the North Atlantic.

Rice and Gislason (1996) compared the North Sea size spectrum with a size spectrum derived from the numbers at length estimated from the output of MSVPA (Sparre, 1991; ICES, 1997). The changes in the slopes and intercepts of the two spectra were similar. When single and multispecies fish stock assessment models were used to predict changes in the slope and intercept of the size spectrum of the commercially exploited fish in the North Sea in response to fishing it was found that both variables were approximately linear functions of overall fishing effort (ICES, 1996; Gislason and Rice, 1996). The linearity was a consistent feature irrespective of whether a single species model with constant recruitment was used, a stock recruitment model was added to this model, or a multispecies model (MSFOR) was used to predict the change. In all cases the slope was inversely proportional and the intercept directly proportional to overall fishing mortality. A sensitivity analysis showed that the response of the size spectrum to changes in fishing mortality was virtually unaffected by the level of natural mortality assumed. The response was far more sensitive to changes in growth and stock recruitment dynamics. Changes in growth resulted in major changes in the relationship between fishing mortality and the slope and intercept of the size spectrum. When growth was reduced, the slope and intercept of the size spectrum became much more sensitive to changes in fishing mortality. When growth was increased, sensitivity decreased.

Gislason and Lassen (1997) analysed the mathematical background for the linearity of the change in slope with fishing mortality. Assuming that natural mortality was a function of 1/length:

$$M = a + \frac{b_1}{L} + \frac{b_2}{L^2}$$

where:

M:	natural mortality
L:	length
a, b_1, b_2:	constants

and that growth could be described by the von Bertalanffy growth equation and it was shown that the slope of the size disribution for a single species could be described by:

$$slope_{species} = \frac{\partial \log N(L)}{\partial L} =$$

$$-\frac{b_2}{kL_\infty^2} * \left(\frac{1}{L(t)} + \frac{L_\infty}{L^2(t)} + \frac{1}{L_\infty - L(t)}\right) - \frac{b_1}{kL_\infty}\left(\frac{1}{L(t)} + \frac{1}{L_\infty - L(t)}\right) - \frac{a+f}{k}\left(\frac{1}{L_\infty - L(t)}\right)$$

Where

N(L): numbers at length

L_∞,k; von Bertalanffy growth parameters

Differentiating with respect to fishing mortality the rate of change of the slope of the size distribution with fishing mortality could be described by:

$$r_{species} = \frac{\partial^2 \log N(L)}{\partial f \partial L} = -\frac{1}{k}\left(\frac{1}{L_\infty - L(t)}\right)$$

which means that the slope is directly proportional to fishing mortality for a given length. The rate at which the slope will change depends, however, on the growth paramters, but not on natural mortality.

The size distribution of the biomass will respond in a similar way. Assuming standard isometric growth it follows that:

$$\frac{\partial \log B(L)}{\partial L} = \frac{\partial}{\partial L}\left[\log(q * L^3) + \log N(L)\right] = \frac{3}{L(t)} + s_{species}$$

The slope of the size distribution of the biomass should therefore respond to changes in fishing mortality in exactly the same way as the size distribution of the numbers.

The size spectrum of the entire fish assemblage is estimated by summing up the abundance at size of the individual species:

$$\log \sum_{species} N_{species}(L)$$

The slope of the assemblage size spectrum is therefore:

$$s = \frac{\sum_{species} N(L) \frac{\partial \log N(L)}{\partial l}}{\sum_{species} N(L)} = \frac{\sum_{species} N(L) * s_{species}}{\sum_{species} N(L)}$$

which is equivalent to the weighted (with abundance) mean of the individual slopes. As the individual slopes decrease with increasing fishing mortality the overall slope will also decrease.

For the size spectrum we now have sufficient theoretical and emperical evidence to be confident that changes in fishing mortality should result in a long term change in the slope of the size spectrum. Provided that the growth and the relative recruitment of the constituent species do not change, the change in the slope should be directly proportional to the change in fishing mortality.

Over shorter timespans the spectrum will change due to interannual changes in recruitment. Over longer timespans changes in recruitment levels might also affect the slope. Murawski and Idoine (1992) thus suggested that the size composition was a conservative property of demersal fish assemblages, and that species replacement would counteract the effect of fishing on the size spectrum slope. Similarly it cannot be ruled out that a general environmental change could result in changes in the level of recruitment that were different for large and small species. If the level of recruitment for large species declined relatively to the level of recruitment for small species, the slope of the size spectrum would decrease in a way which might be indiscernable from the influence of an increase in overall fishing mortality. Finally, the response of the spectrum is sensitive to changes in growth, and growth changes might influence the slope of the size spectrum in way similar to fishing. With these possibilities in mind, and interpreted with care, the slope of the size spectrum seems to be a useful indicator of changes in fishing effort.

3.4.2 Diversity spectra

At previous meetings this Working Group has investigated temporal patterns in species diversity with size (ICES CM 1994/Assess/Env:1; ICES CM 1996/Assess/Env:1). The rationale behind this work has been that fishing would effect larger slower growing and late maturing species to a larger extent than smaller species with a more rapid turnover. If this is the case, changes in diversity with size are expected with changes in fishing effort. There is now some evidence that this might take place. Piet and Rijnsdorp (Working Paper) thus found that the abundance of species with a large size at maturity decreased while those with a small size at maturity increased in beam trawl surveys in the southern North Sea. However, they did not investigate how this affected diversity by size.

Even where patterns in diversity by size do show some changes over time the results are far from easy to interpret and difficult to link theoretically to fishing effort. Indeed, the modelling study of Gislason and Rice (1996) suggests that the diversity spectrum would be among the less useful measures of changes in fishing effort. The way in which the slope and intercept of the diversity spectrum changed with fishing effort differed between single species models with and without stock/recuritment relationships and multispecies (MSFOR) models. Furthermore, none of the models predicted the higher evenness at low levels of fishing mortality suggested by the analyses of survey data from the North Sea from 1906–1909 and 1990–1995 made by Rijnsdorp *et al.* (1996).

Furthermore, species diversity is assessed with a multitude of diversity indices. Each of these indices combine information on species richness and evenness into a single number. High evenness occurs when species are equal or approximately equal in abundance, low evenness when the species composition is dominated by a few abundant species. Due to the relative importance each index gives to evenness and richness, it is difficult to compare the indices. A Working Paper by Roger *et al.* compared diversity of coastal demersal fish faunas in the northeast Atlantic by diversity profiles calculated from:

$$H_\alpha = (\log \sum_i p_i^\alpha) / (1-\alpha)$$

Substituting 0.1 and 2 for the scale parameter α, H_α, will be directly related to species richness, Shannon's entropy and Simpsons dominance index, respectively. Thus for α near zero, the index will be dominated by richness, while for larger values of α, species evenness will have progressively more effect.

Without a theory to provide a causal link between fishing intensity and diversity, it will be difficult to know whether diversity is a useful measure of fishing impact. Recent work by Hall and Greenstreet (in review) suggests, however, that there are patterns in relationships between species richness, individual abundance, and size which might be linked to fisheries effects at the community level.

Hall and Greenstreet (in review) described the relationships between species diversity, the abundance of individuals, and body size in a demersal fish community. They investigated patterns in different geographic regions in the northwestern North Sea and over a 60-year period. A striking similarity with previously reported data for insect communities was observed. A dome-shaped relationship between both species richness (S) and individual abundance (I) with body size was found when data were categorised in logarithmic (to base 2) weight classes. The same power law relationship between S and I, of the form $S = aI^b$, existed for both types of fauna. The coefficient b of this relationship did not differ between regions or over time, whereas the intercept a declined over time. This decline could not be accounted for by sampling artifacts and Hall and Greenstreet suggest that it may provide an informative measure of the effect of fisheries exploitation on the community. They also demonstrated that rank abundance relationships within body size classes exhibited a similar pattern to that found in insects, of the form A r – m (where A = abundance and r = species rank). These similarities with insects and the robustness of the patterns for fish when compared over large spatial (100 km) and temporal (decadal) scales, suggest that common explanations may underlie the organisation of these communities. With respect to fisheries effects, it would appear from these data that the coefficient a of the power law relationship $S = I^b$, when data are categorised into weight classes, might be a valuable measure of the effect on fish species assemblages of fishing disturbance.

The empirical studies of changes in species diversity have been inconsistent and the theoretical understanding has not advanced to a state where the underlying process can be modelled. More work is therefore needed before predictions can be made about how fishing would affect the diversity spectrum.

3.4.3 Diversity profiles

Improvements in the measurement and interpretation of diversity have recently been made using methods of diversity ordering (Tothmeresz, 1995), where a range of diversity indices within a family show varying sensitivities to rare and abundant species. These profiles display graphically a family of diversity indices obtained by changing the scale parameter a. There are several available, but one that is recommended for large datasets is Renyi's diversity index family.

$$H_a = (\log \Sigma p^a{}_i) / (1 - a)$$

When substituting 0, 1, and 2 for the scale parameter a, H_a will be directly related to the species richness (i.e., is the log. of the species number), Shannon's entropy and Simpsons dominance index, respectively (Hill, 1973). Thus for a near zero, richness will have more effect on H_a, but for larger values of the scale parameter, species evenness has more effect. For scale parameters which increase from 1 to 4 the influence of rare species will be gradually replaced by the influence of dominant species. One community is more diverse than another if its' diversity profile is equal to or above that of another, over the whole range of the scale parameter. If the two profiles intersect at any point then they can be considered non-comparable (i.e., different diversity indices would rank the communities differently).

Diversity profiles were calculated for the demersal fish catches (number/8m beam trawl/hour) from the coastal waters of the northeast Atlantic (Rogers *et al.*, in press). Results suggest that this is a robust technique for identifying differences in diversity between assemblages, which takes account of all combinations of species richness and evenness.

3.4.4 References

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Figure 3.2.1.1.1.1. Position of the 8 boxes in the North Sea.

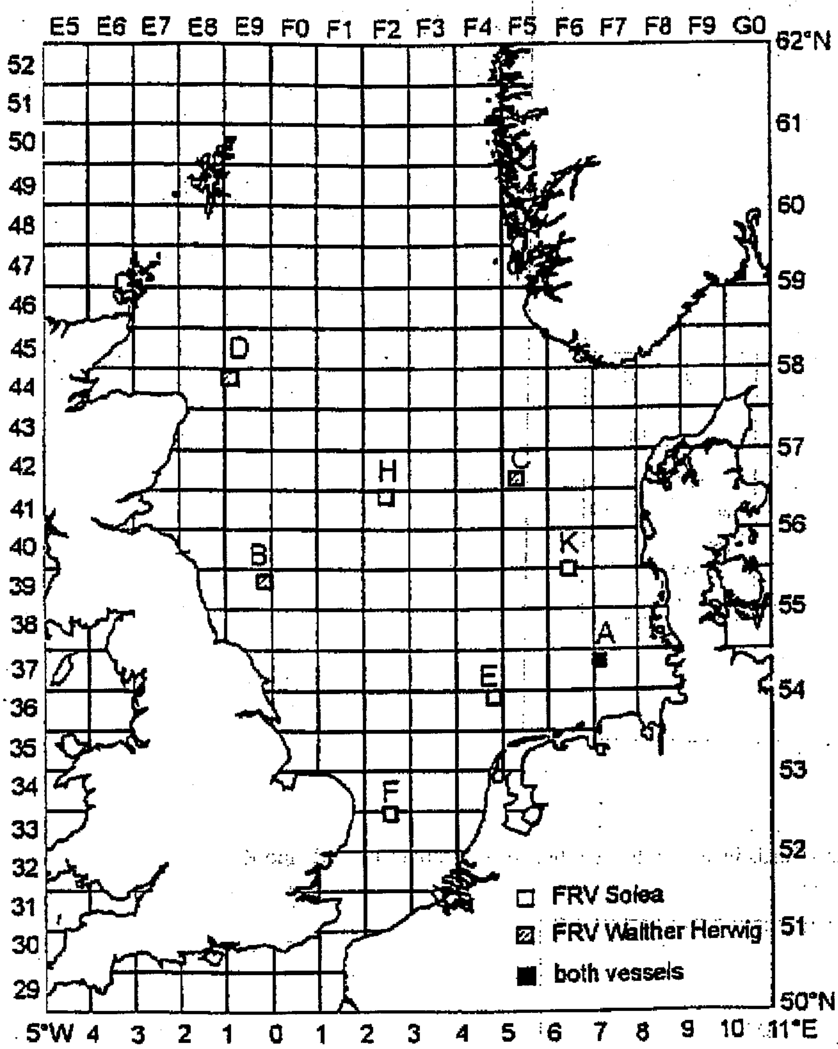


Figure 3.2.1.1.1.2. MDS plot of similarities (Bray–Curtis index) within and between the boxes A, B, C, and D.

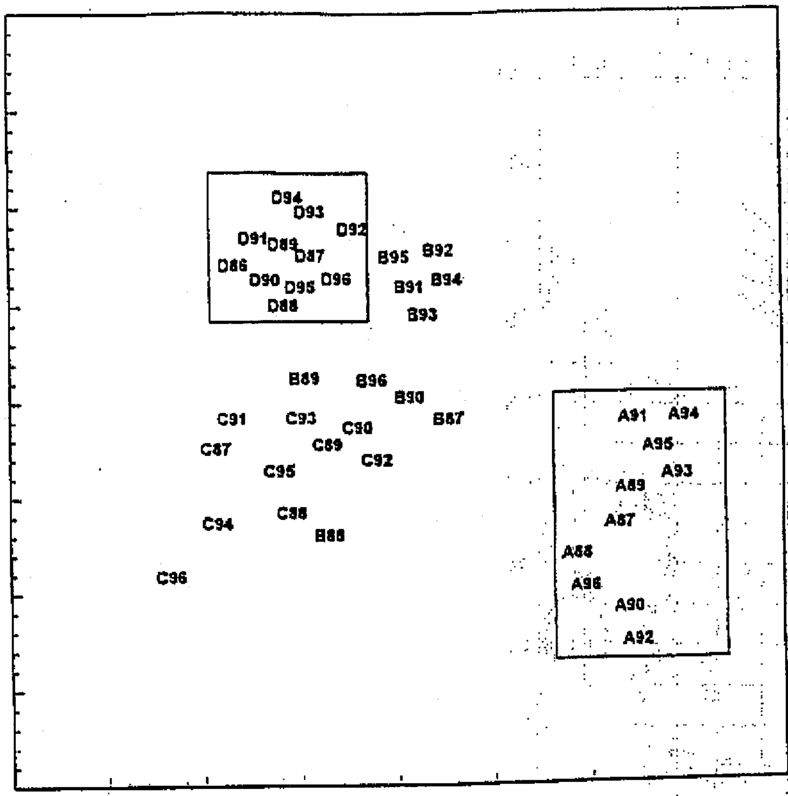
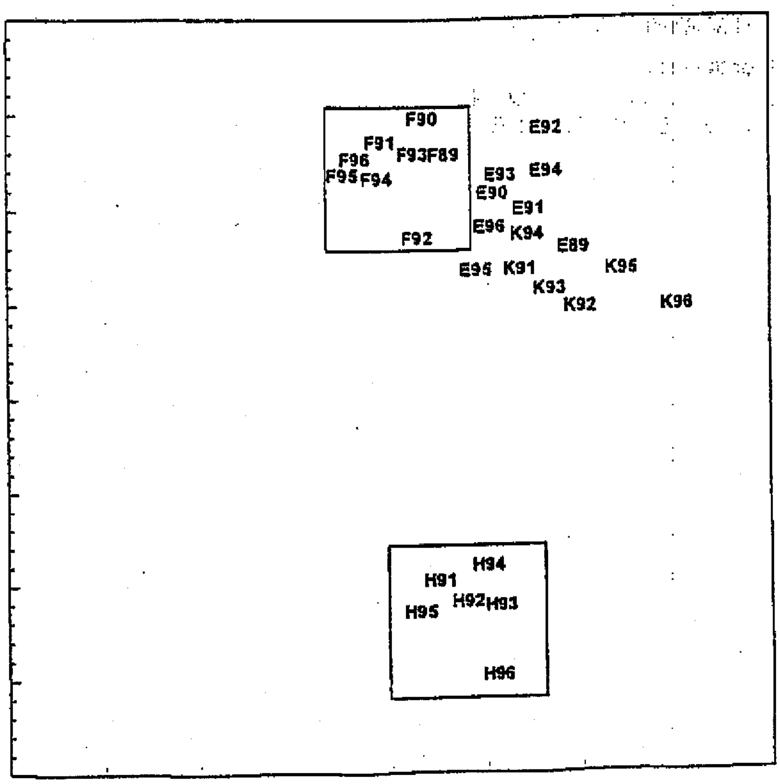


Figure 3.2.1.1.1.3. MDS plot of similarities (Bray–Curtis index) within and between the boxes E, F, H, and K.



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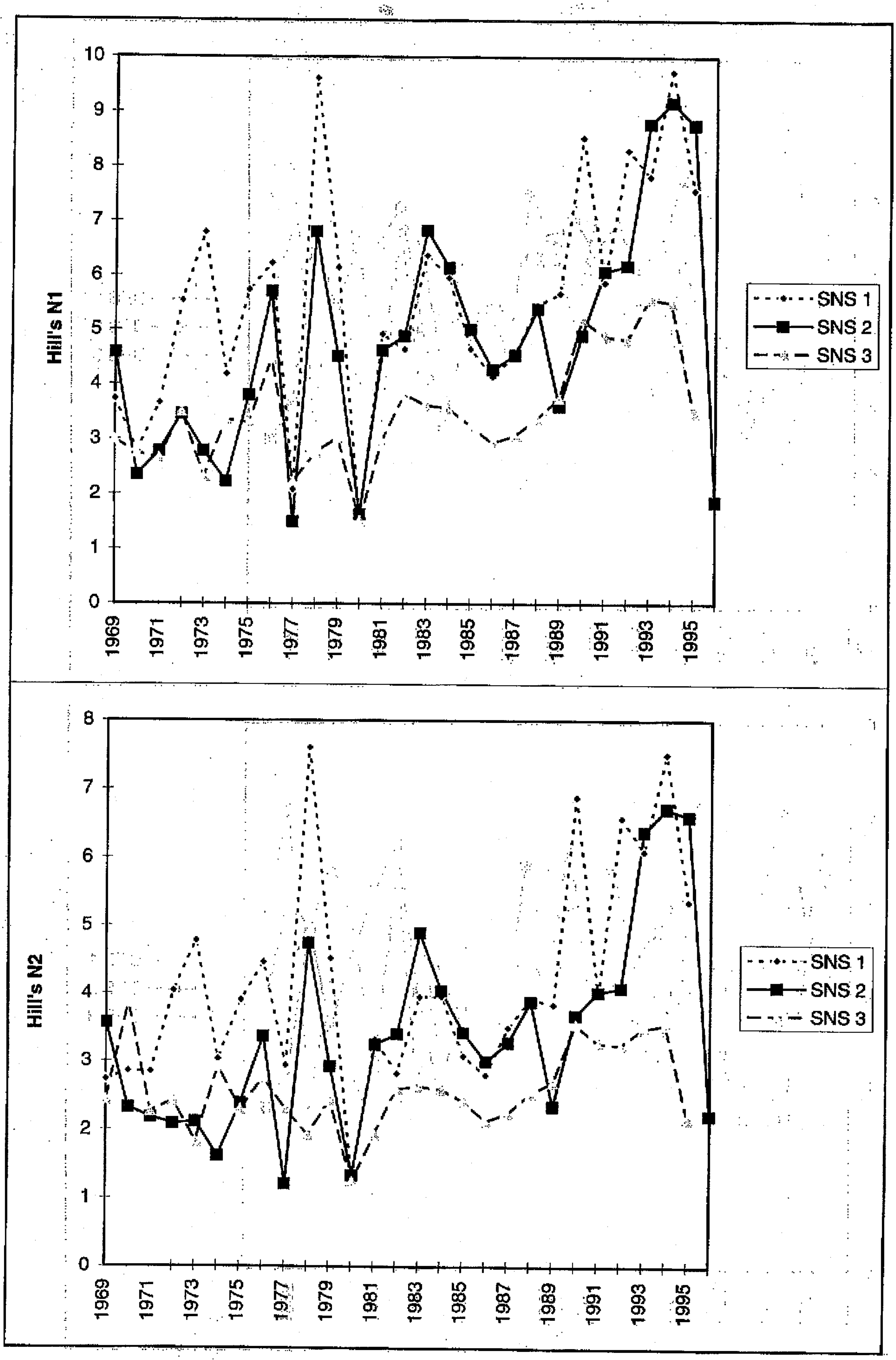

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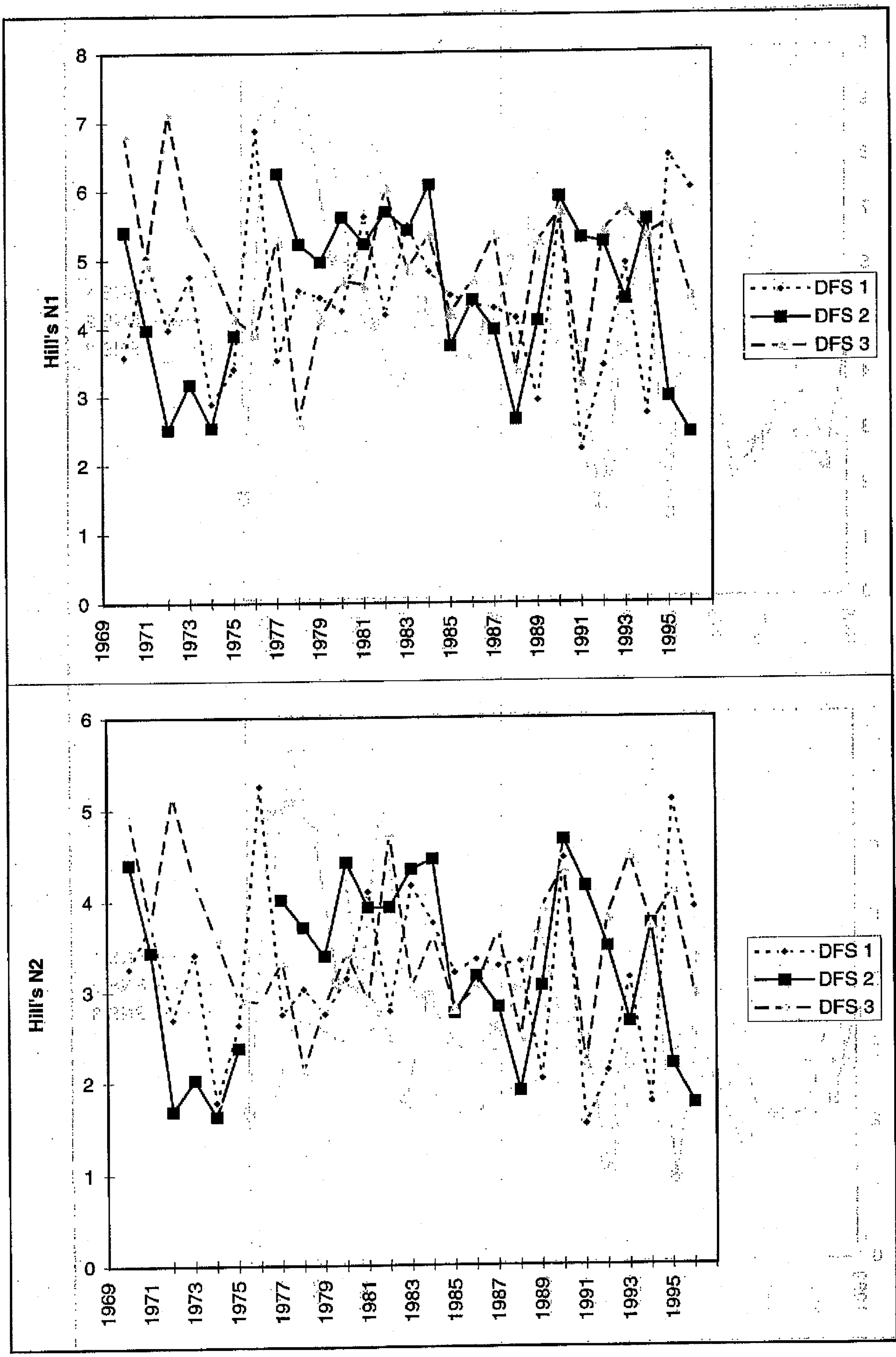

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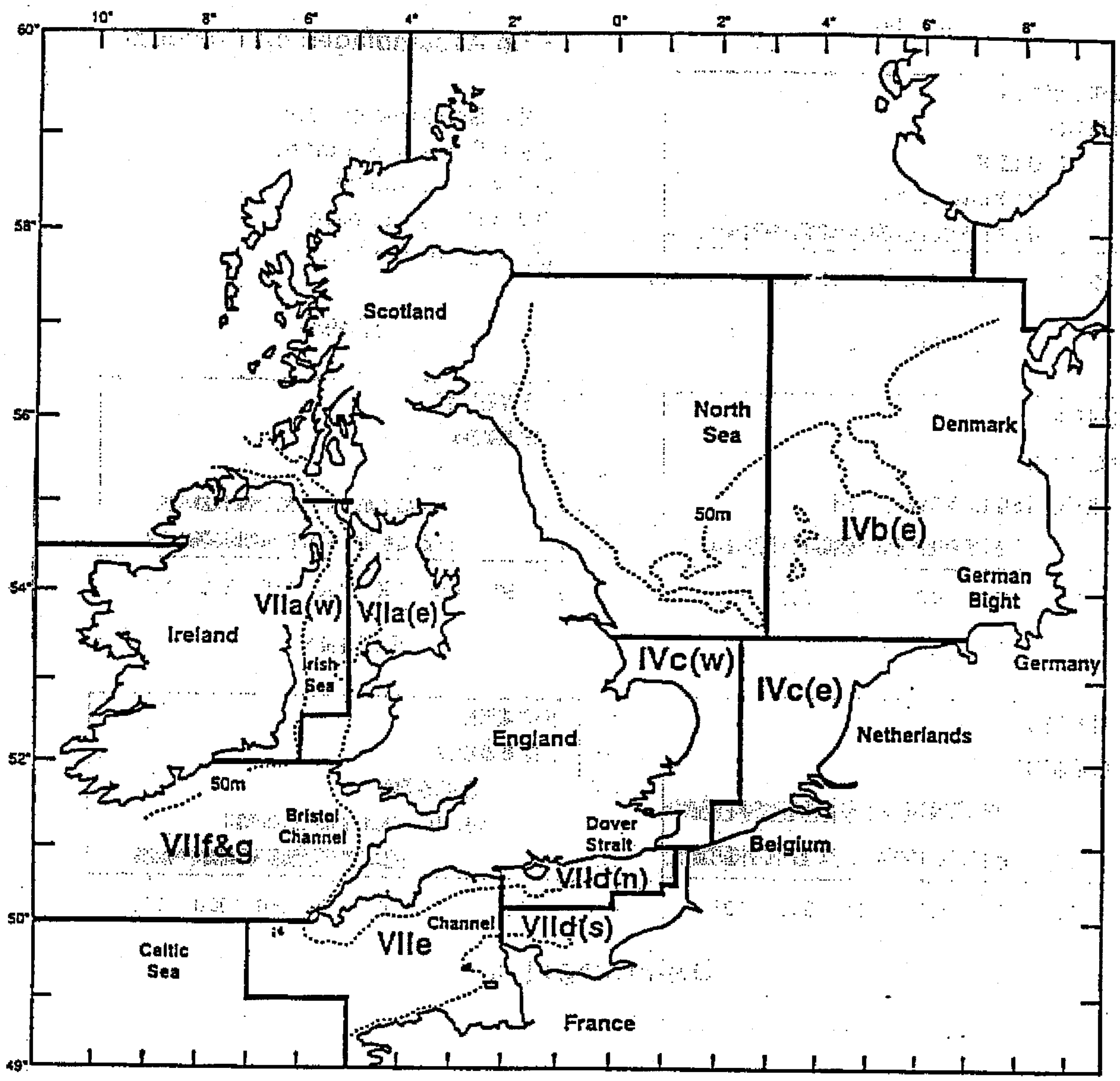

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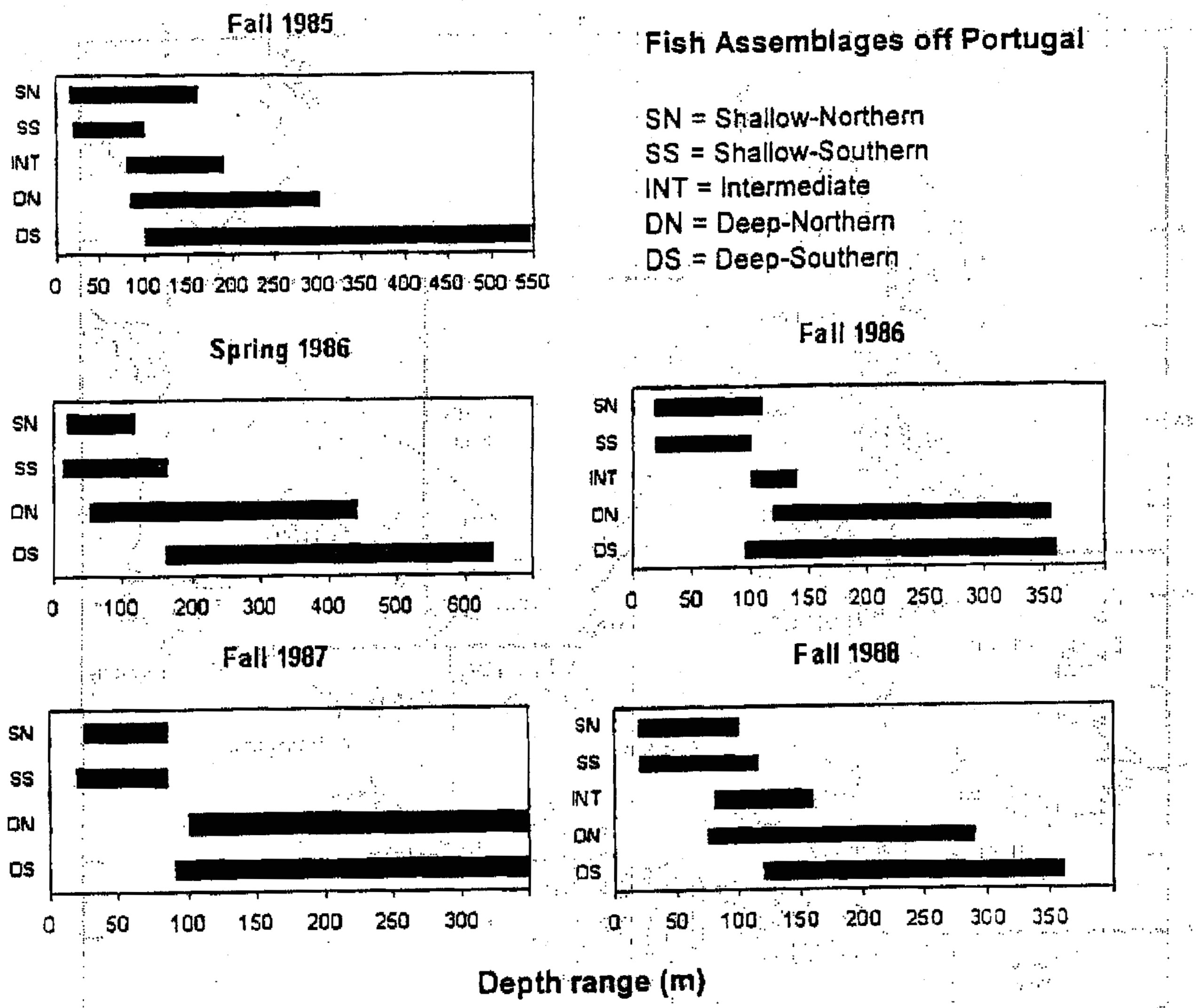

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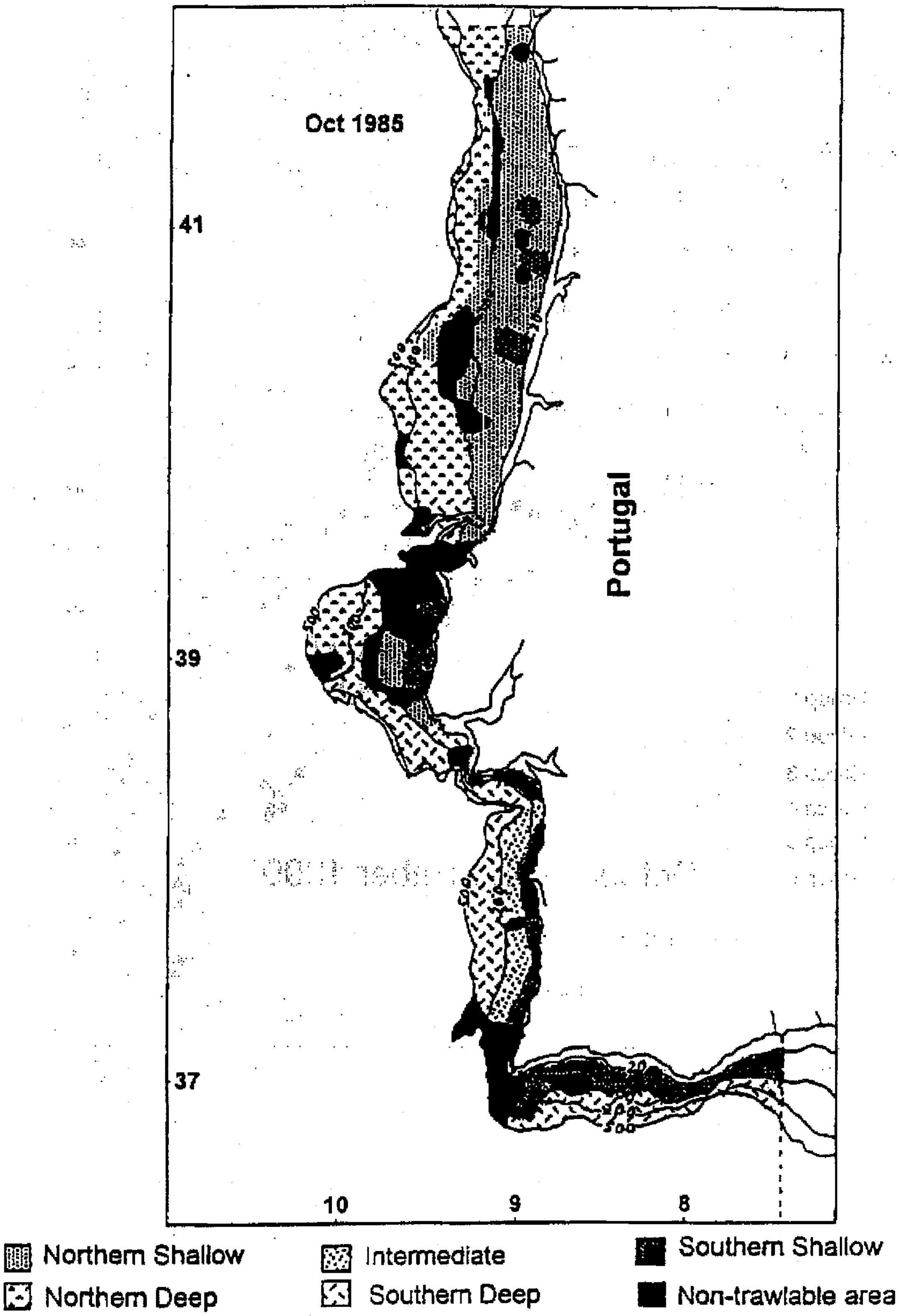

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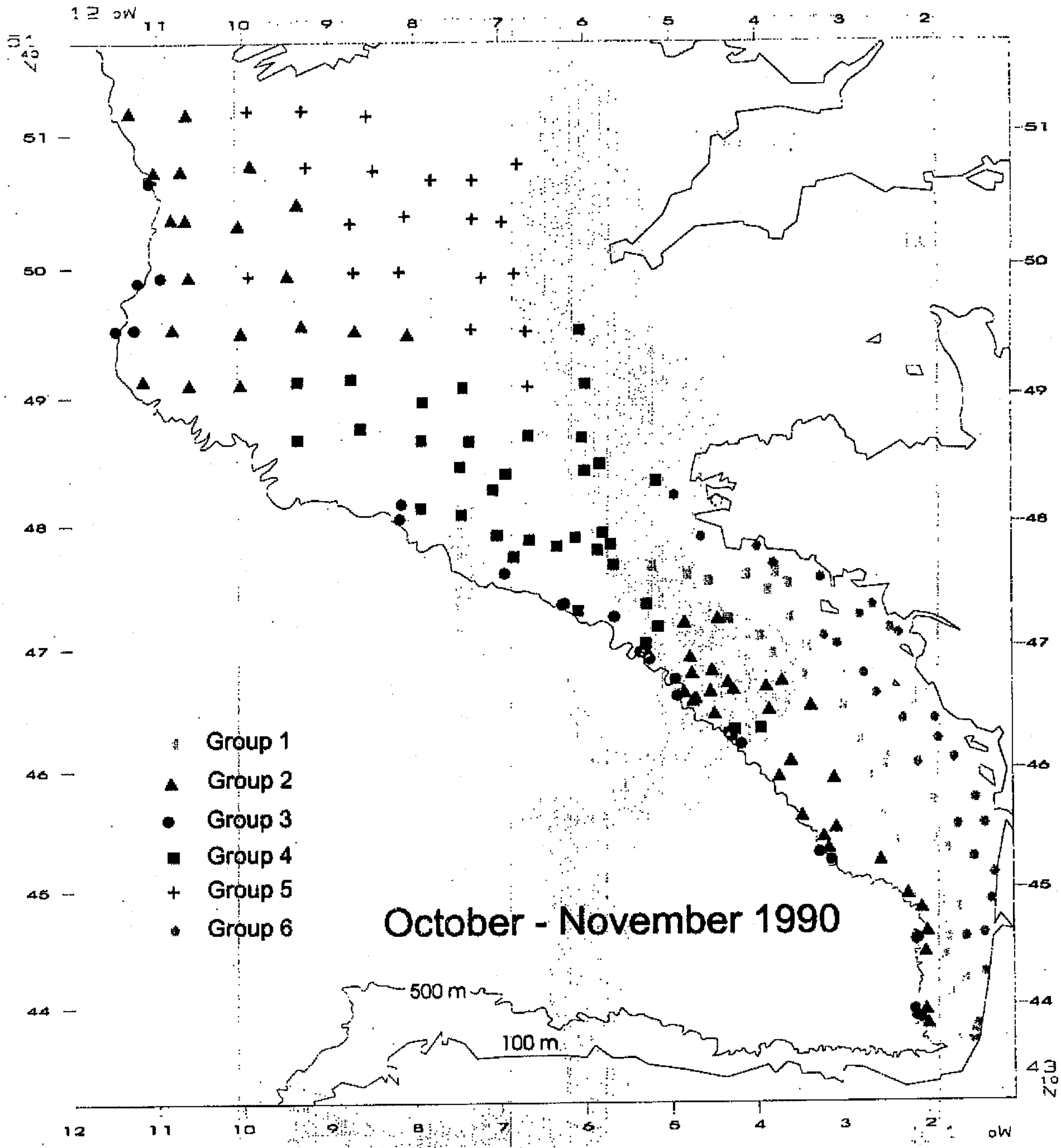


Figure 3.2.2.1.3.2.

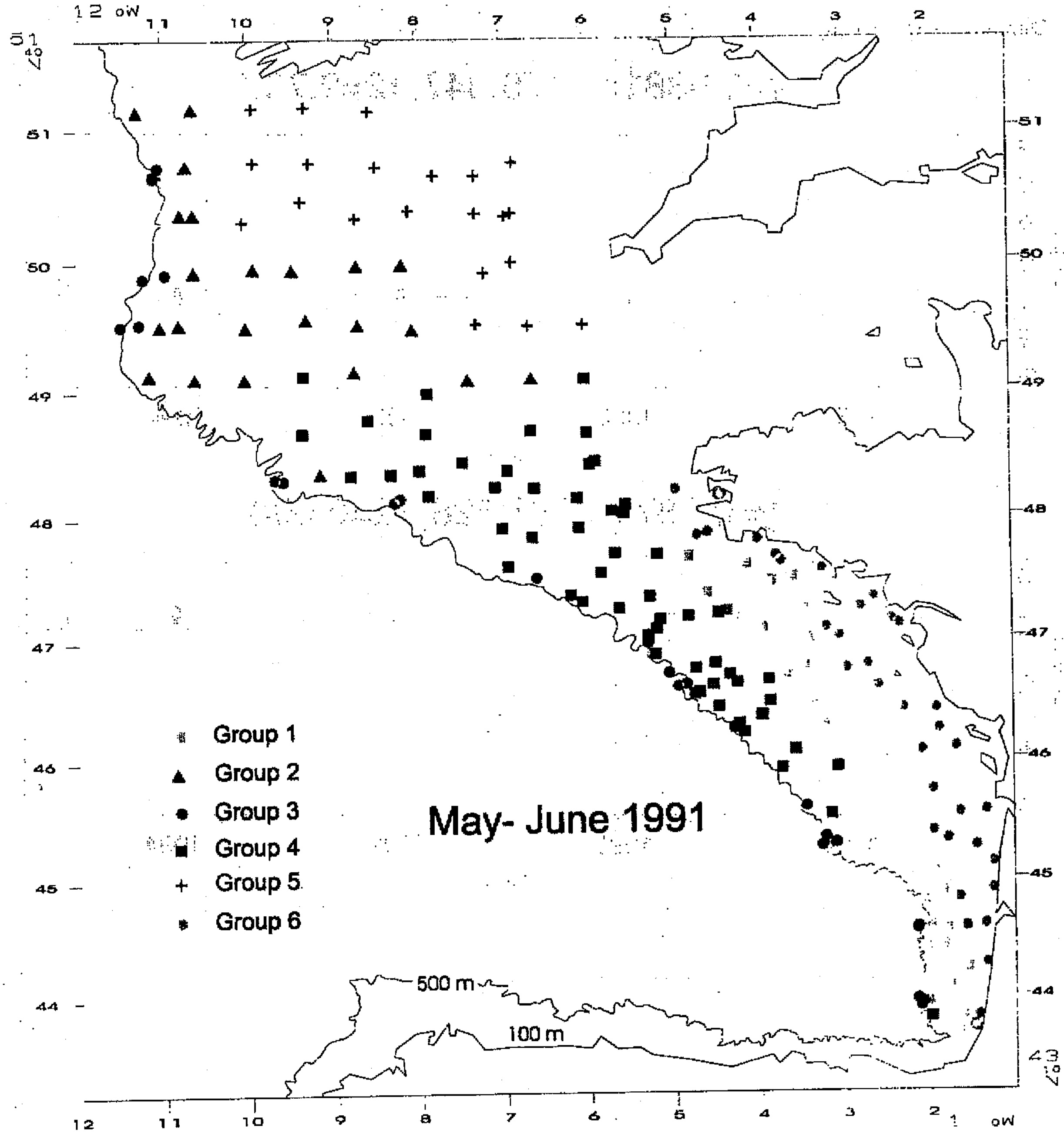


Figure 3.2.2.1.4.1. Patterns of slopes (upper) and intercepts (lower) ± standard error over years from annual regressions of ln(numbers) on ln(length) for demersal species only. Open circles: all species; dots: set of selected species.

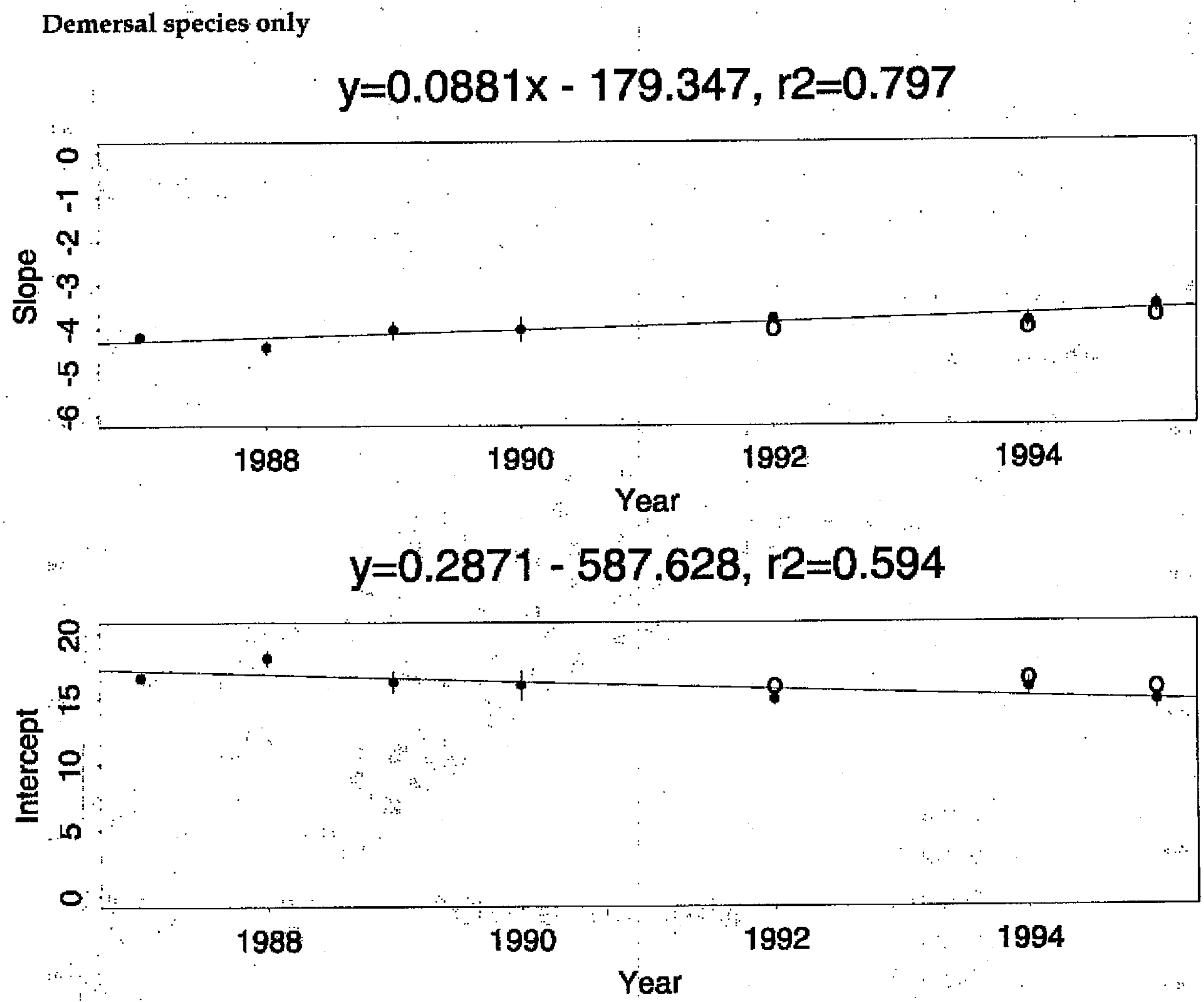


Figure 3.2.2.1.5.1. Location of trawling stations in the Gulf of Riga. The number indicate depth of a given station.

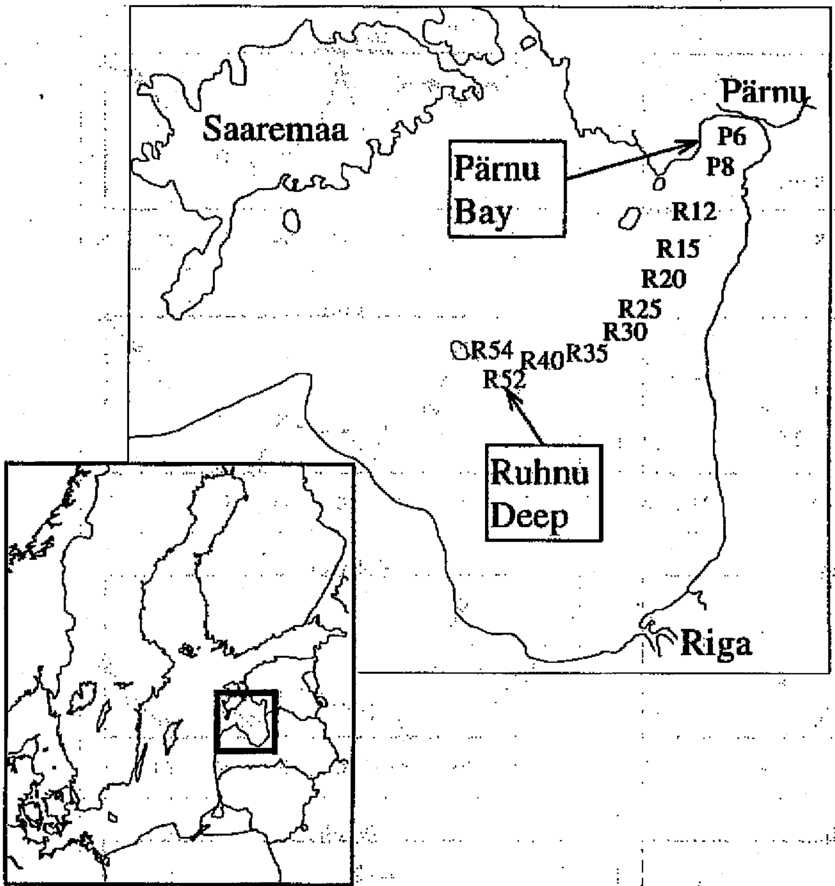


Figure 3.2.2.1.5.2. Dynamics of the abundance-based year-effect with the least significant difference (LSD) bar for the most abundant fish species in the NE Gulf of Riga over the years 1974–1986 and 1994–1996.

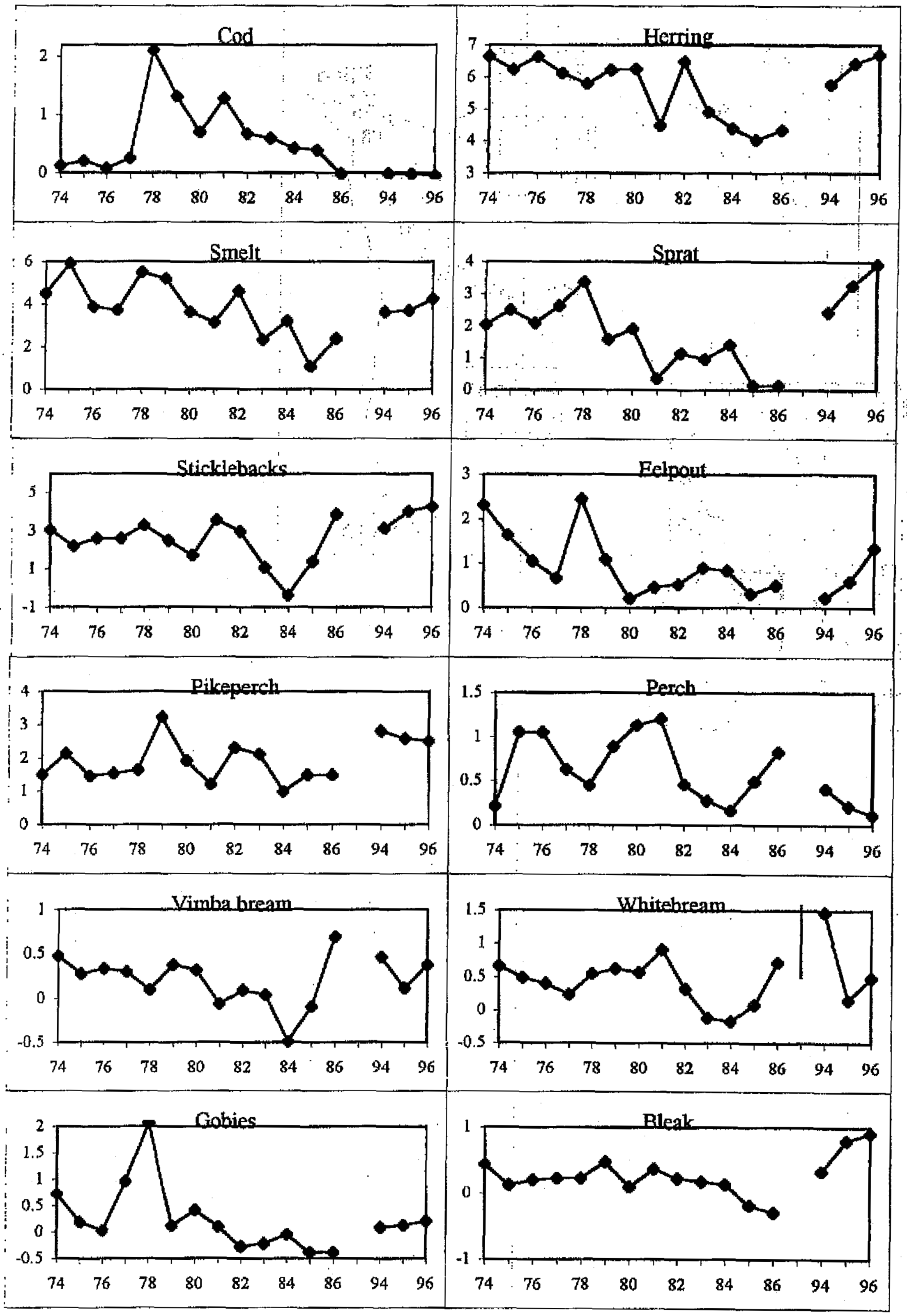


Figure 3.2.2.1.5.3. Dynamics of the mean number of fish species present in experimental bottom trawl catches in the shallow and deep areas during 1974–1986 and 1994–1996.

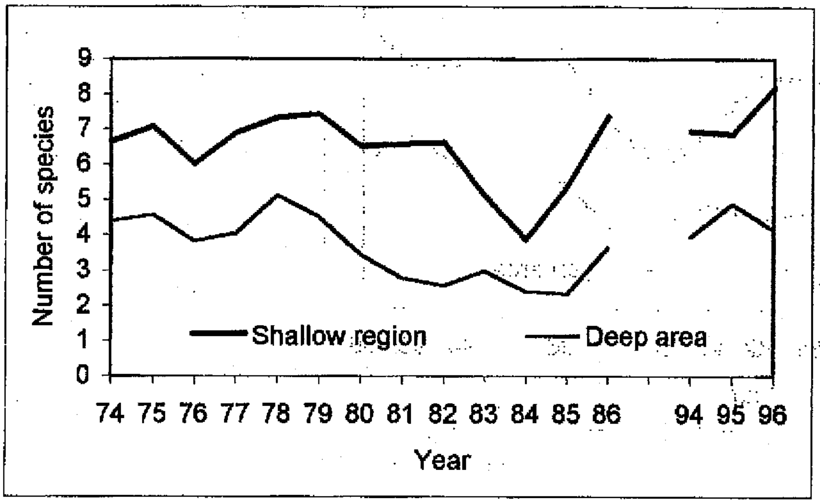
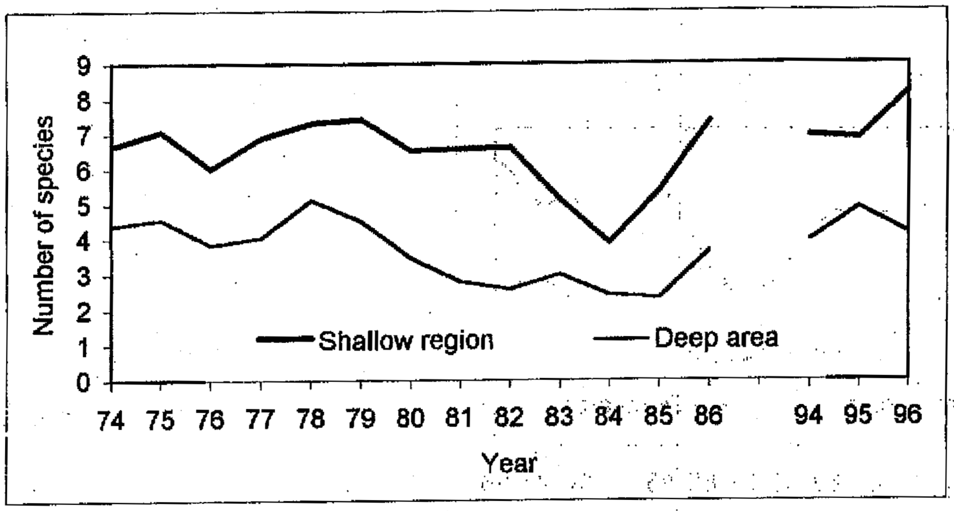
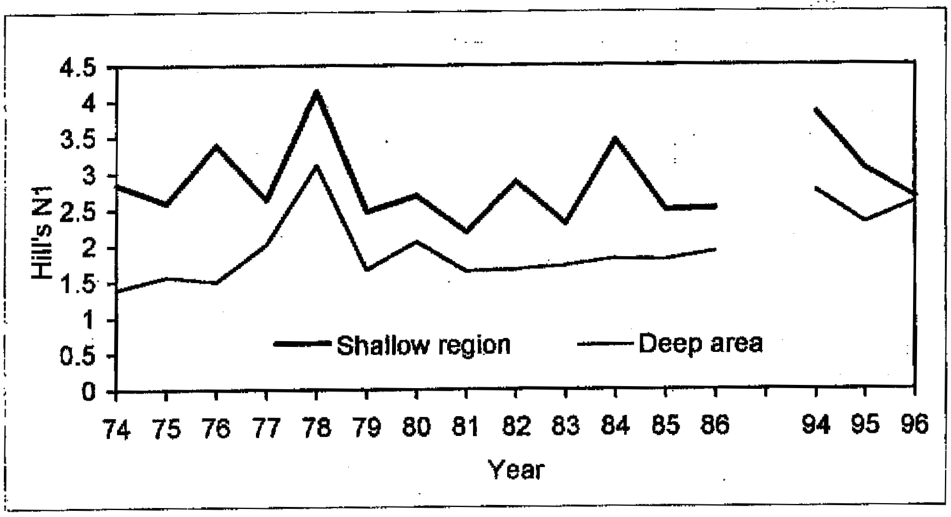


Figure 3.2.2.2.1.1.





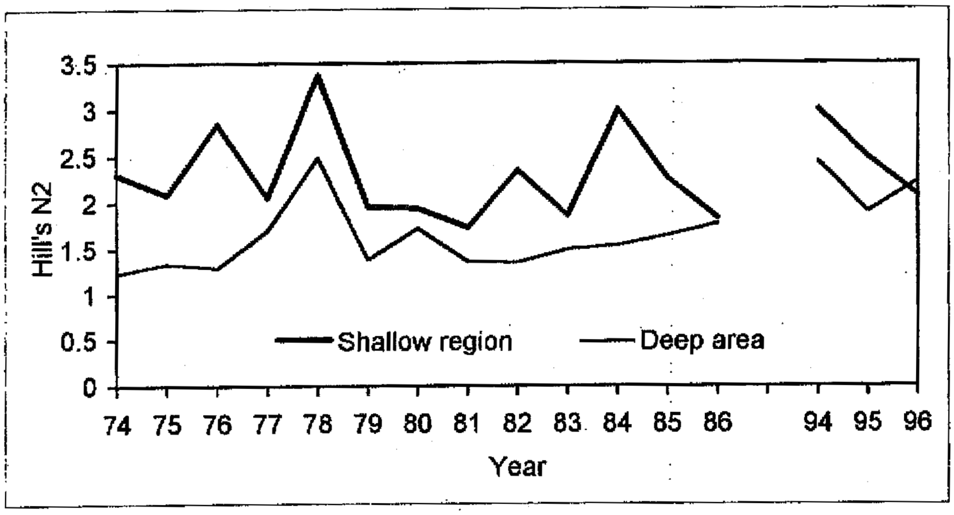


Figure 3.2.2.2.2.1. (Barents Sea case study). The survey area with subareas (A, B, C, D, D', E, and S) and strata used in the bottom trawl survey. Stations taken 1996 are also shown; fixed positions, distance = 16/24/32 nautical miles.

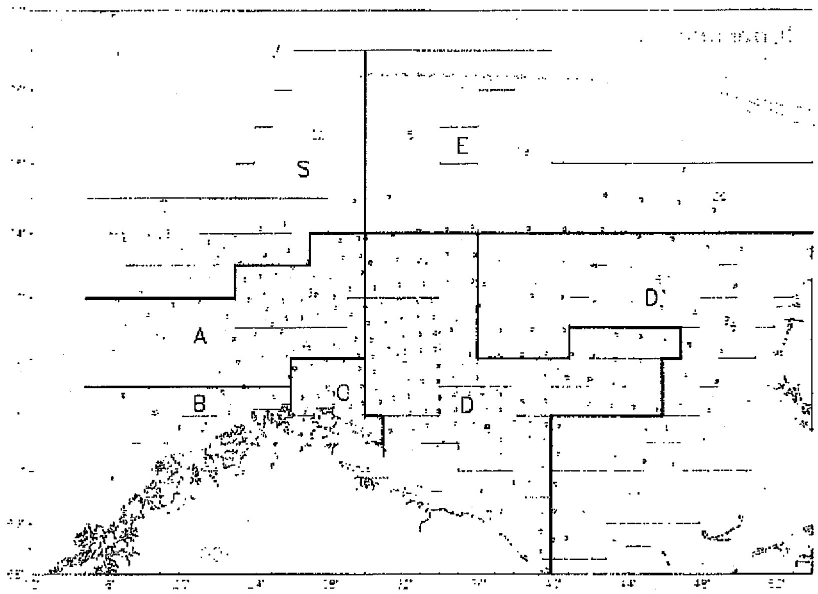


Figure 3.2.2.2.2.2. Hills N1 and N2 calculated from a series of 12 years of the Norwegian bottom trawl survey in the Barents Sea.

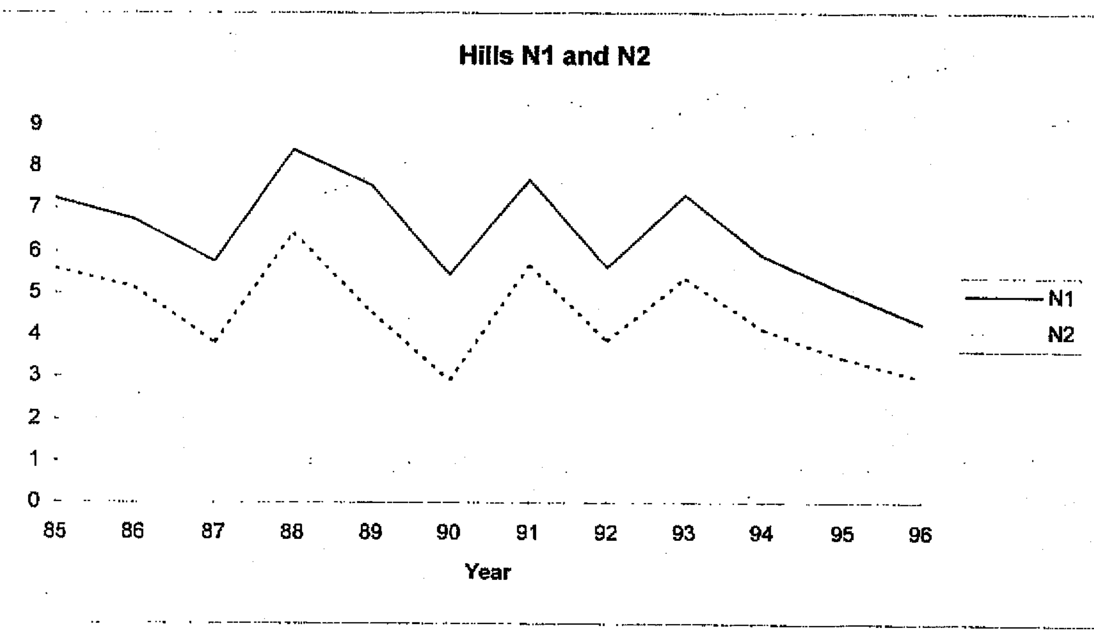


Figure 3.2.2.2.2.3. K-dominance curves. Barents Sea bottom trawl survey data.

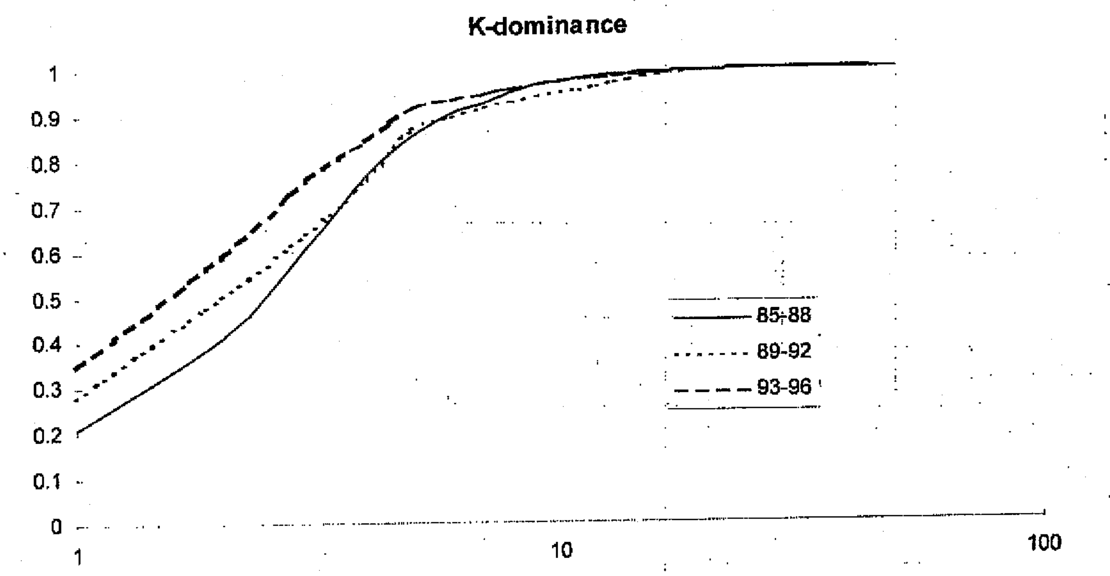


Figure 3.2.2.2.2.4. Trend in species evenness for Barents Sea bottom trawl survey.

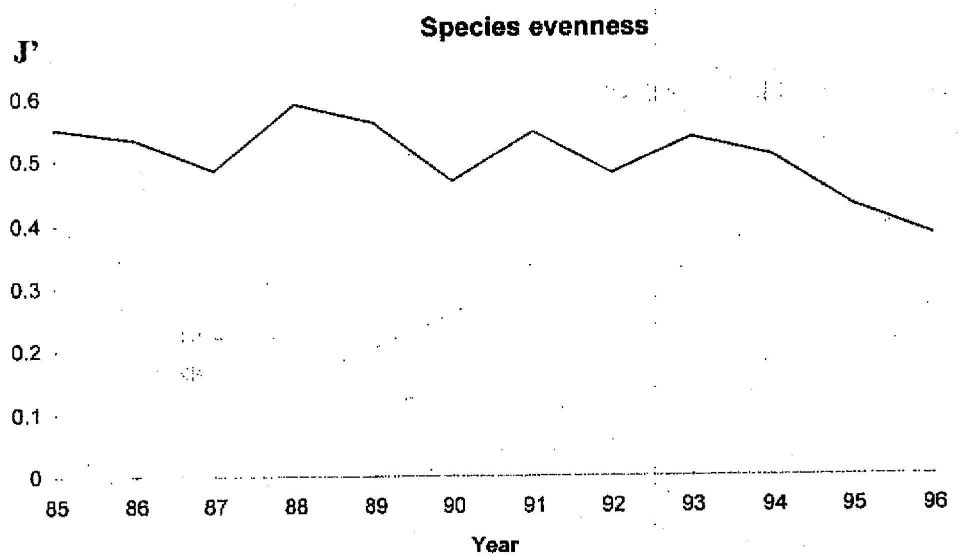


Figure 3.2.2.2.2.5. Trend in species richness for Barents Sea bottom trawl survey.

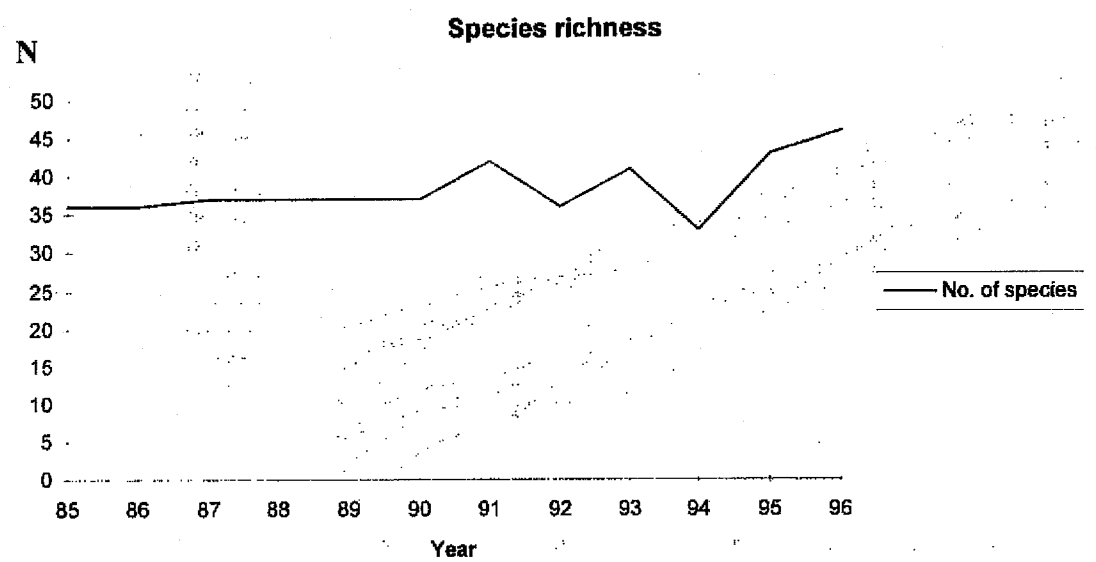


Figure 3.2.2.2.2.6. Size spectra for 12 surveys for lengths between 20 and 50 cm.

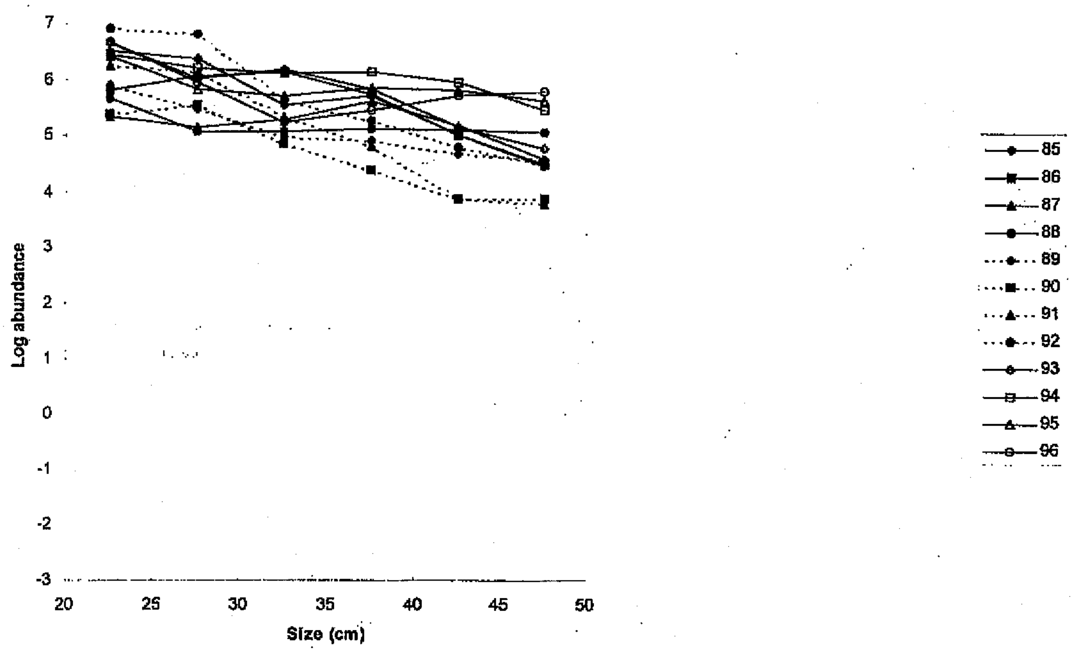


Figure 3.2.2.2.2.7. Size spectra for 12 survey for lengths between 50 and 100 cm.

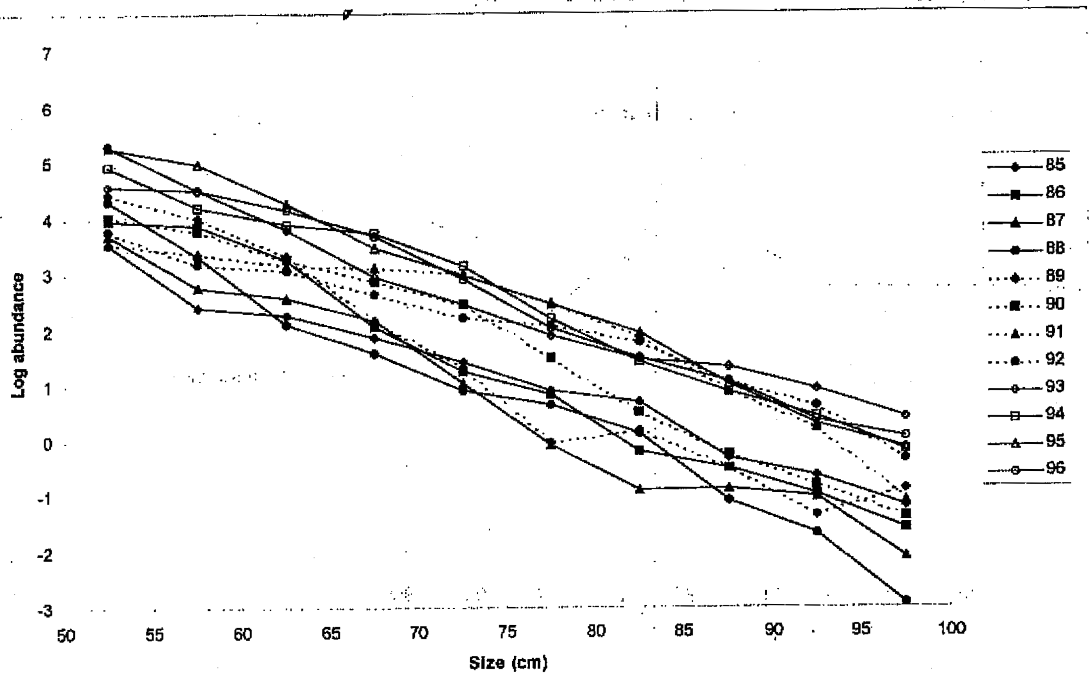


Figure 3.2.2.2.2.8. Comparison of trends in the yearly slopes of the size spectra for the length groups 20 to 50 and 50 to 100 cm.

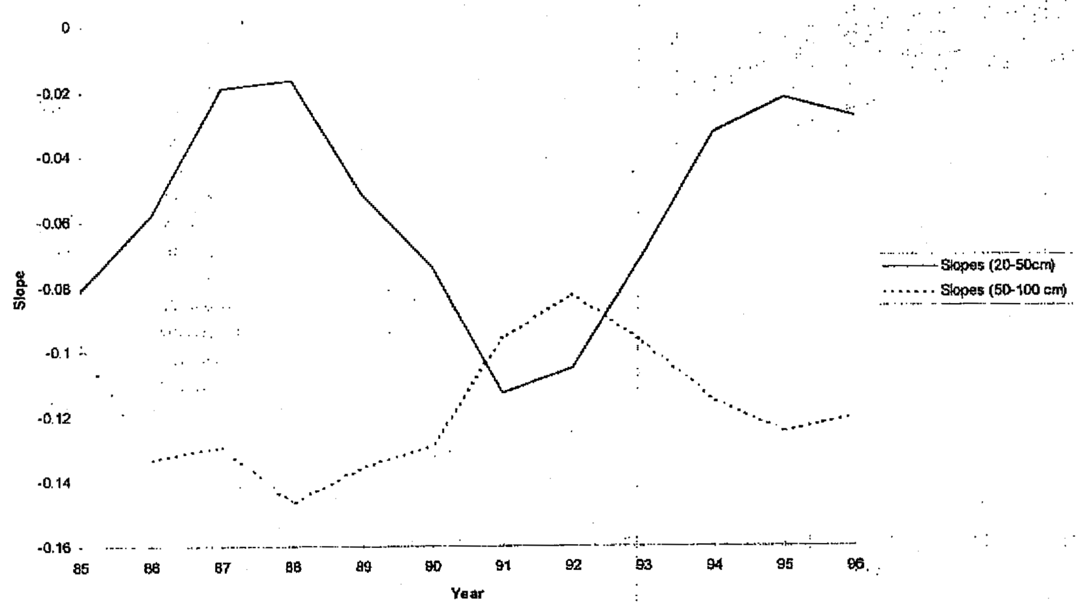


Figure 3.2.2.2.2.9. Yearly size spectra from 12 survey for lengths between 20 and 50 cm excluding data for cod and haddock.

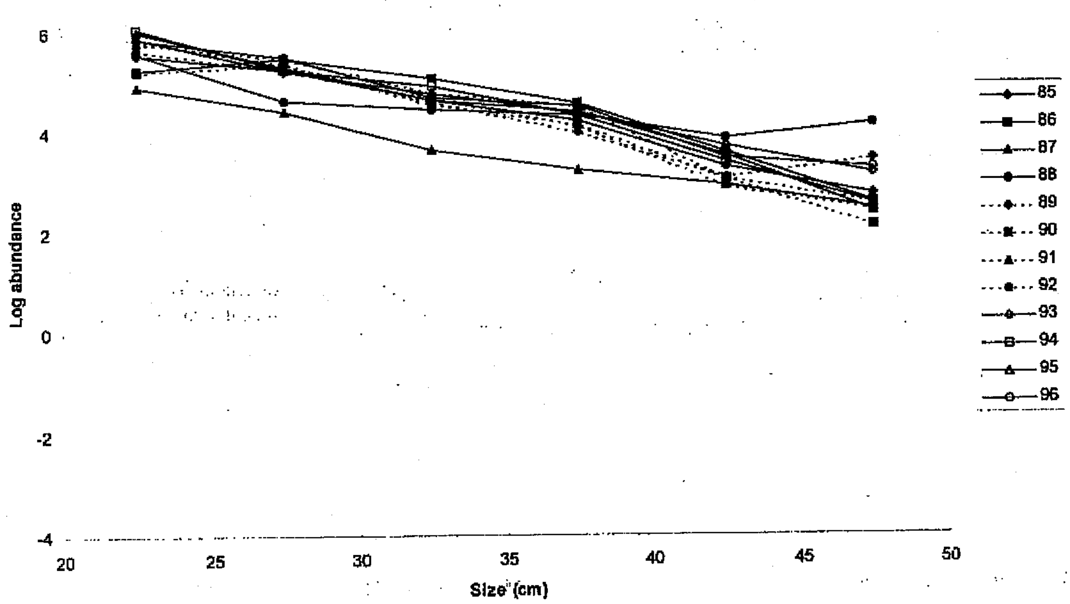


Figure 3.2.2.2.2.10. Yearly size spectra for 12 survey for lengths between 50 and 80 cm excluding data for cod and haddock.

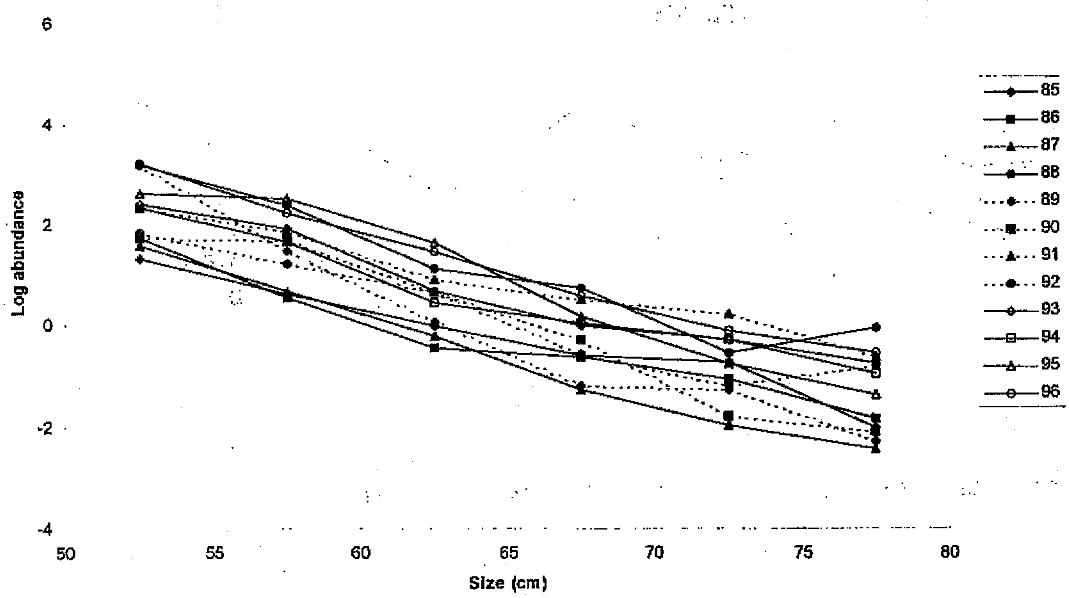


Figure 3.2.2.2.2.11. Comparison of trends in the yearly slopes of the size spectra for the length groups 20 to 50 cm and 50 to 80 cm with data for cod and haddock excluded.

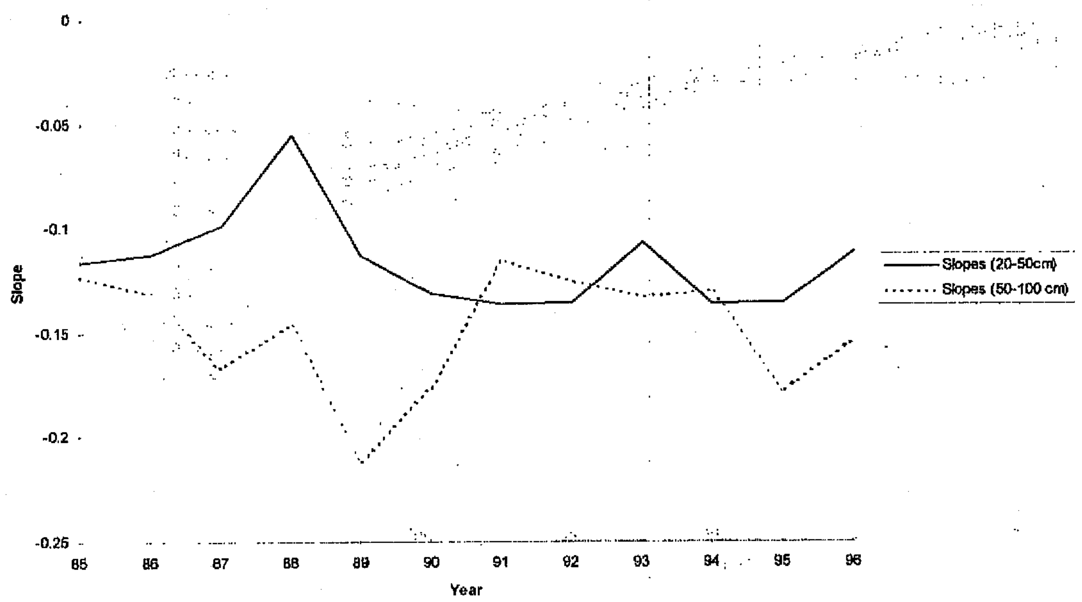


Figure 3.2.2.2.2.12. (The Barents Sea case study). Hills N1 calculated for 2 different areas in the Barents Sea. A+B+C is the western subarea while D is the eastern subarea.

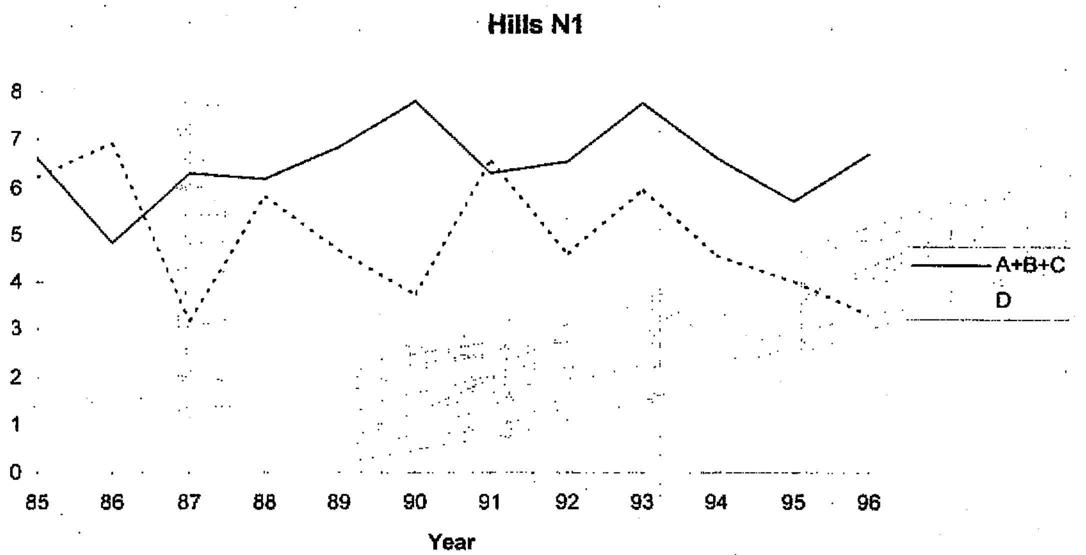


Figure 3.2.2.2.2.13. Hills N2 calculated for 2 different areas in the Barents Sea. A+B+C is the western subarea while D is the eastern subarea.

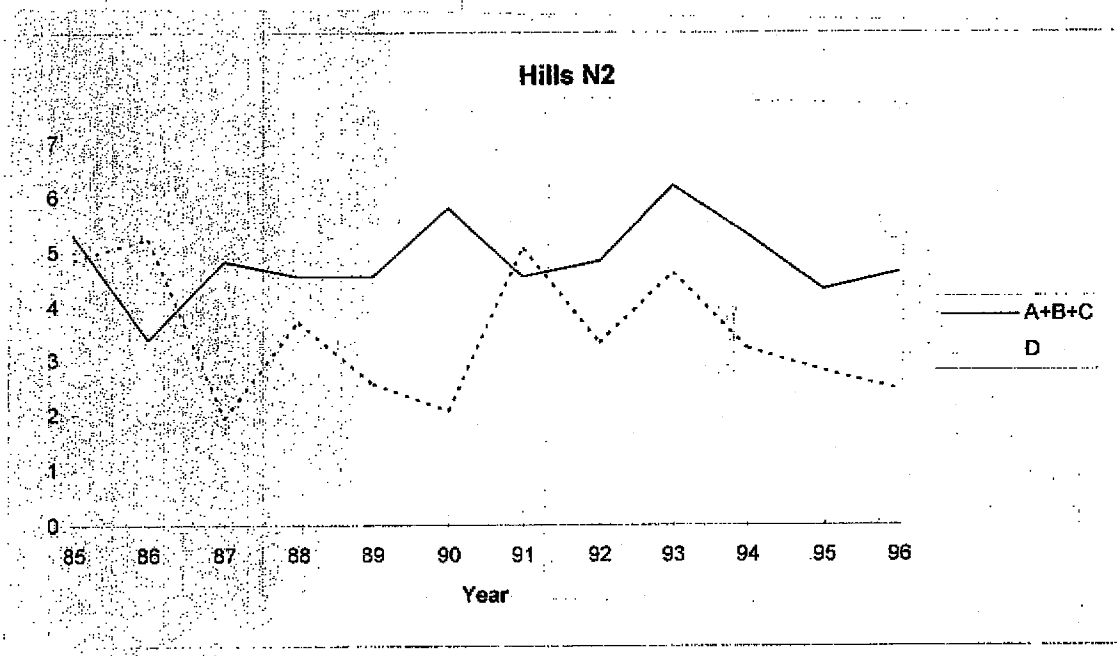


Figure 3.2.2.2.2.14.

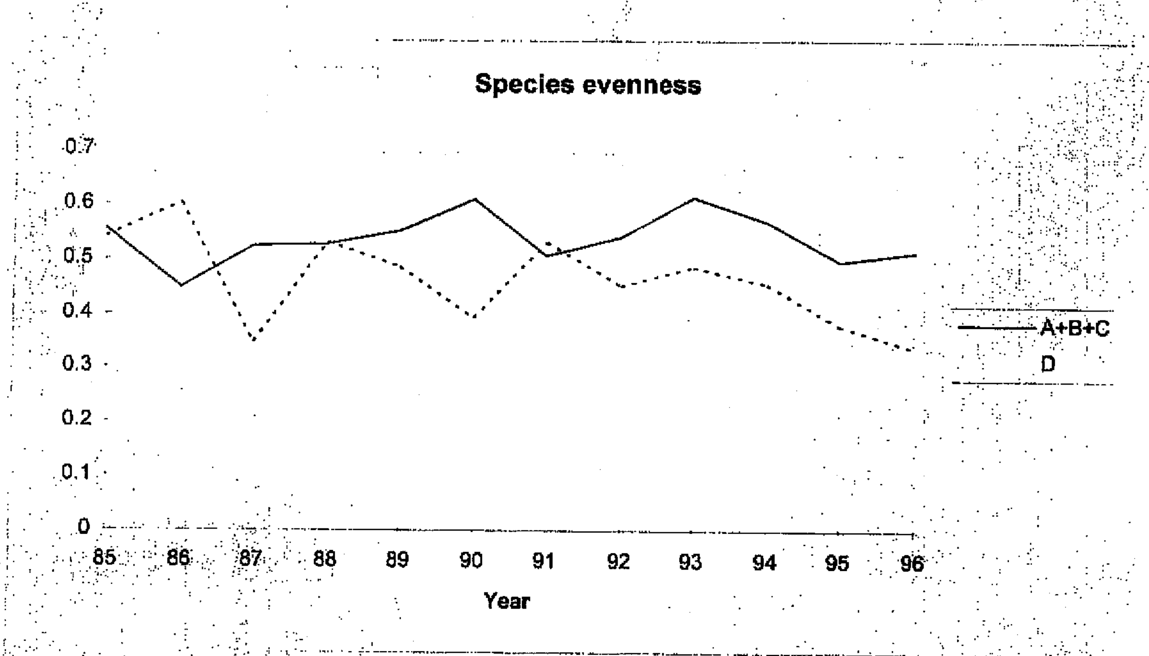
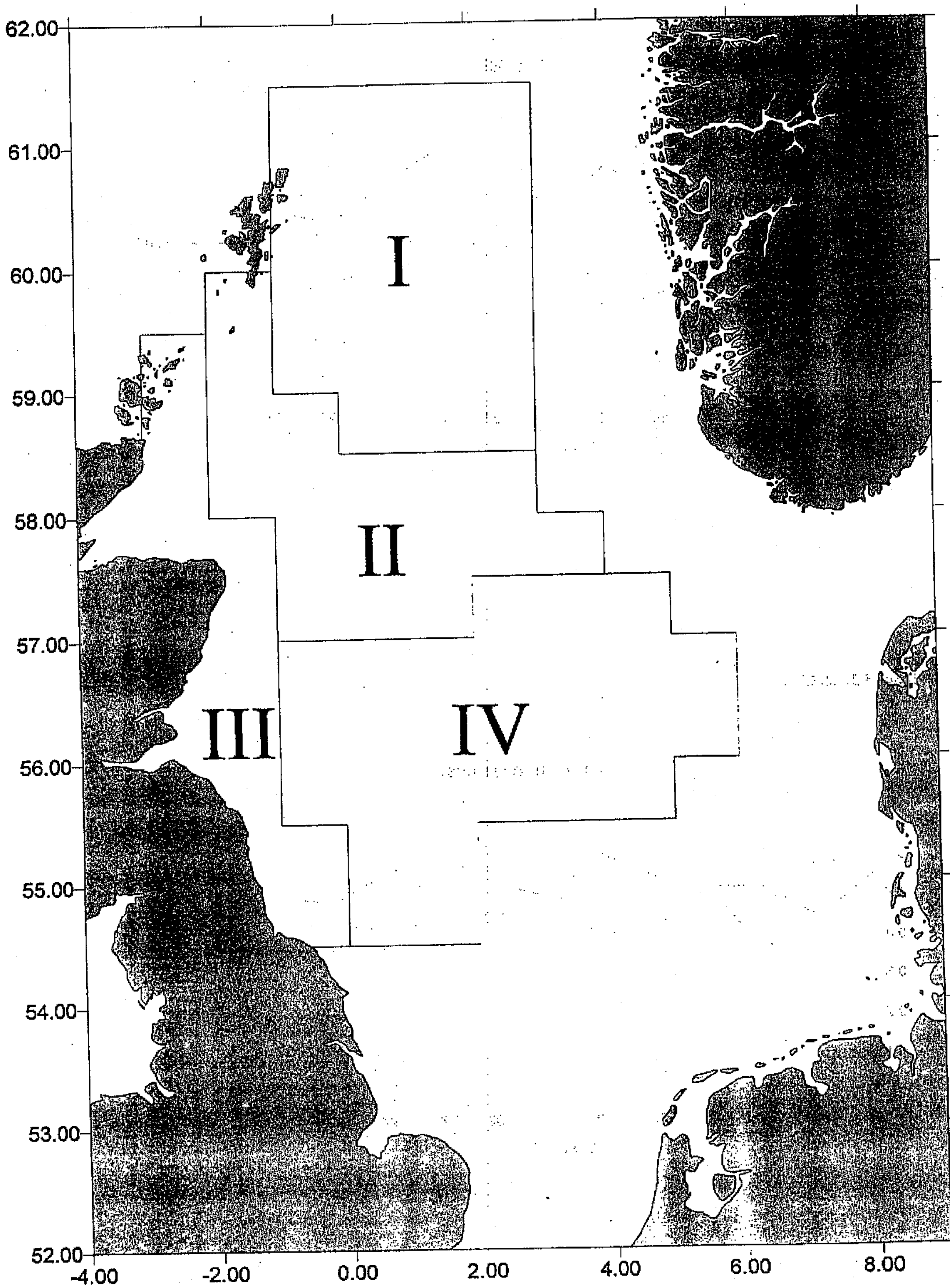


Figure 3.2.3.1.1.1. Map of the North Sea showing four identified areas.



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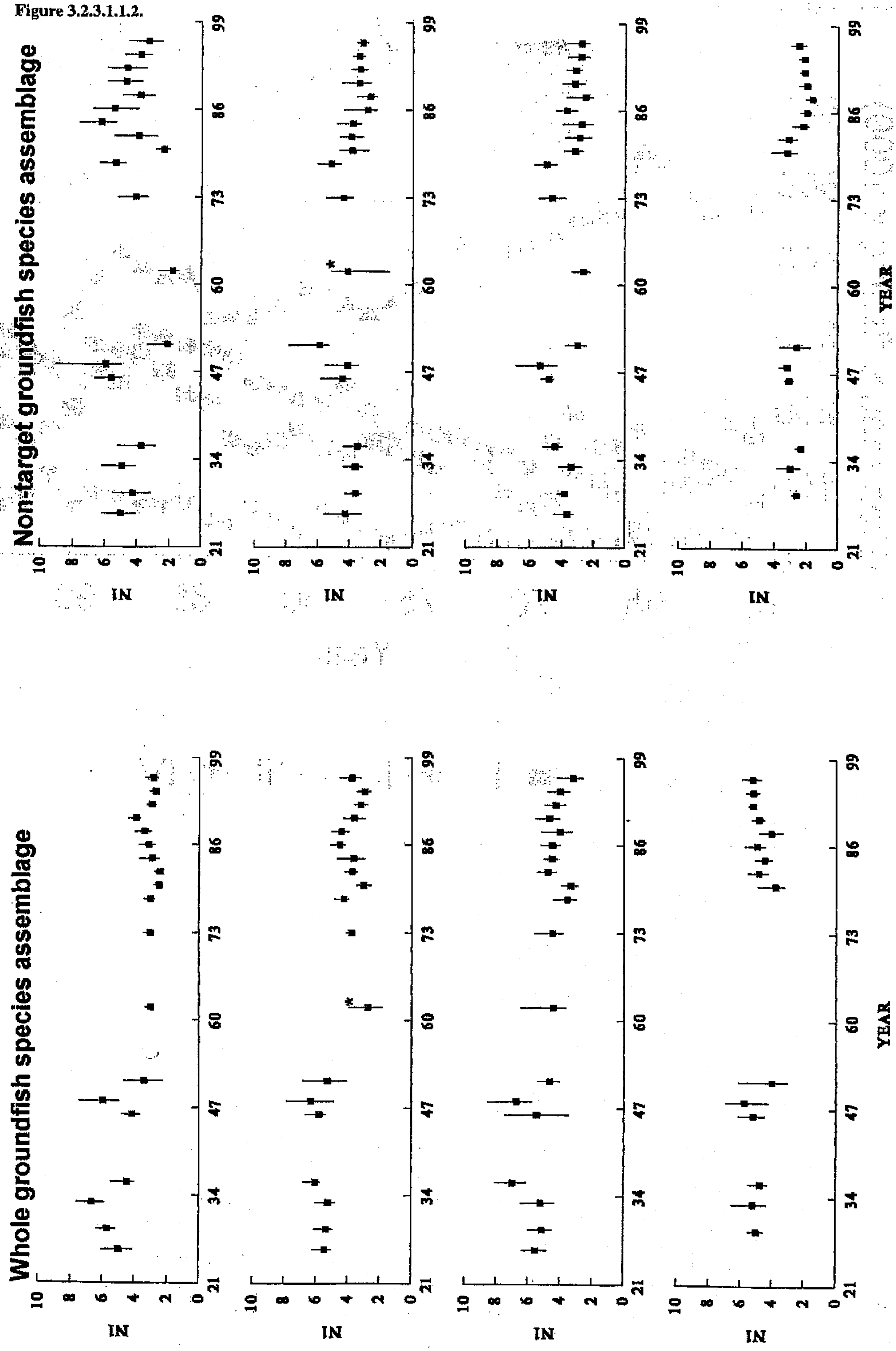


Figure 3.2.3.1.1.3.

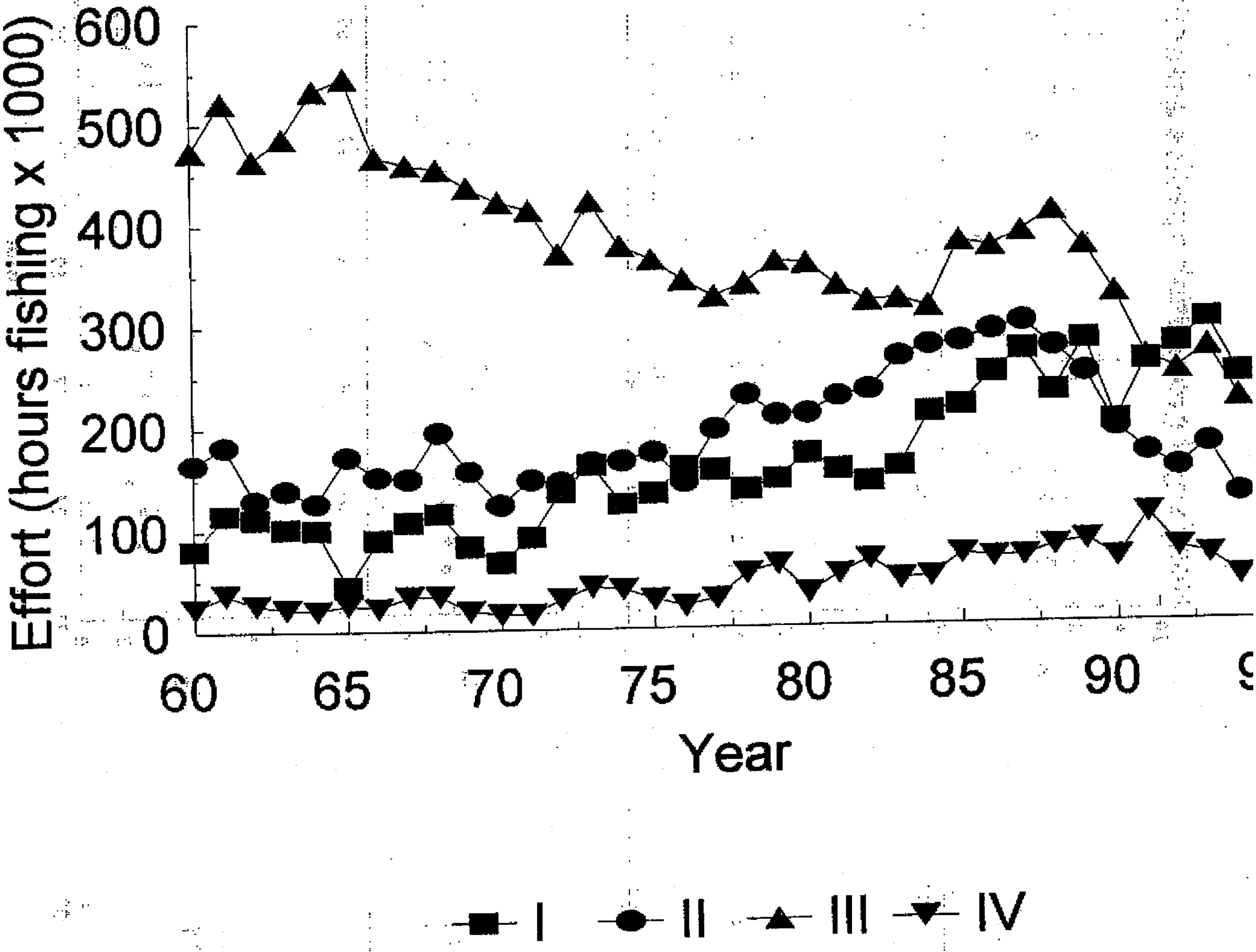
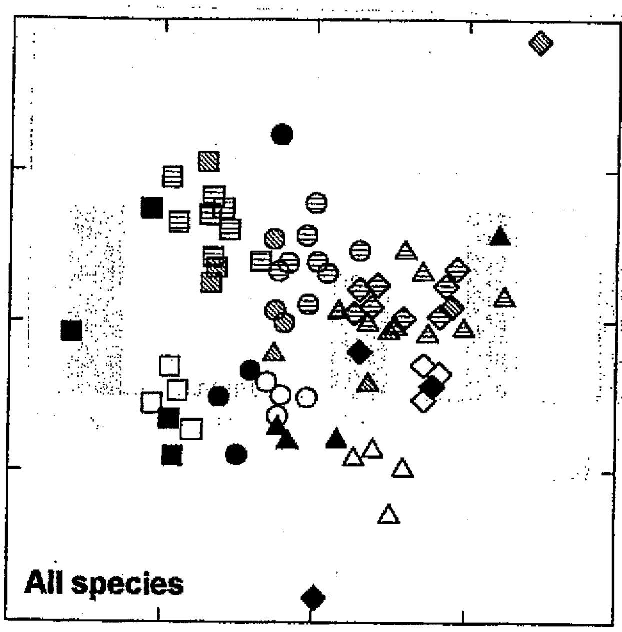


Figure 3.2.3.1.1.4.



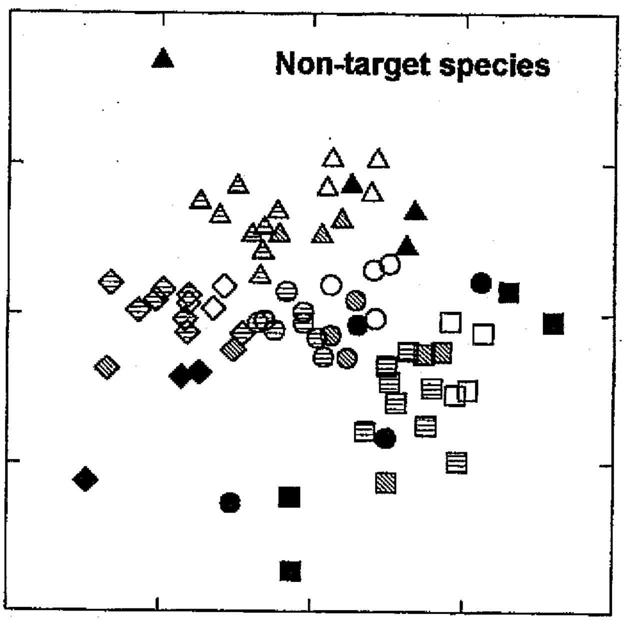


Figure 3.2.3.2.1.1. Index of total fishing effort in 1991 for the boxes.

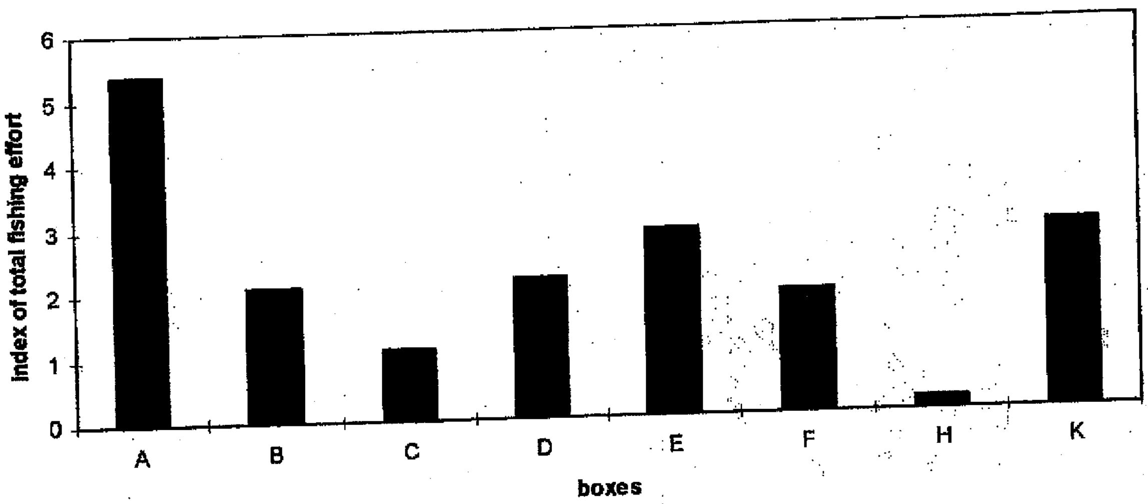


Figure 3.2.3.2.1.2. Box A–D. annual variation of Hill N1 diversity index from 1987 to 1996 of fish assemblages (including pelagic species).

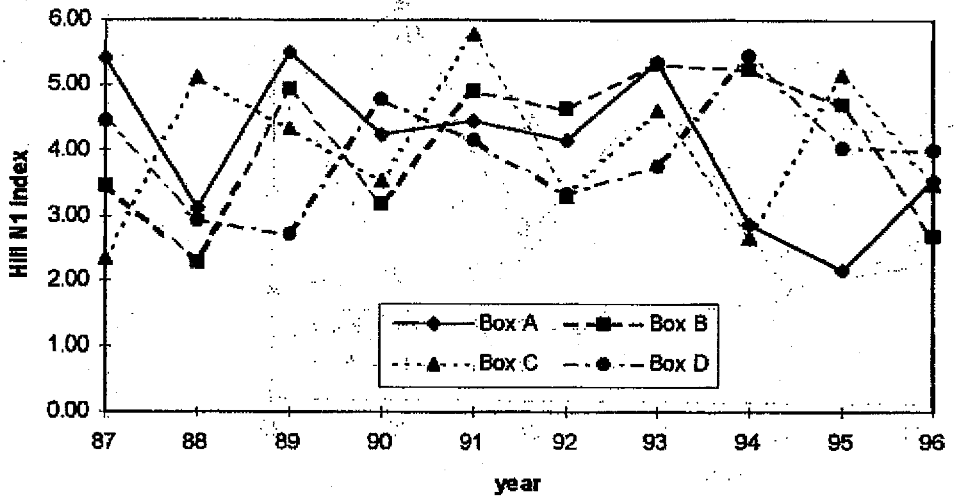


Figure 3.2.3.3.1.3. Box A–D. Annual variation of Hill N1 diversity index from 1987 to 1996 of fish assemblages (excluding pelagic species).

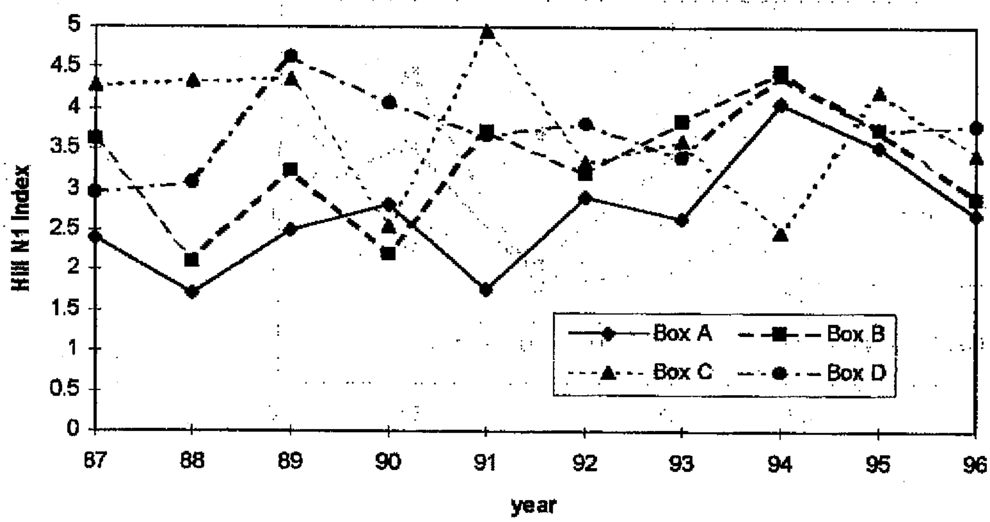


Figure 3.2.3.2.1.4. Box E–K. Annual variation of Hill N1 diversity index from 1989 to 1996 of fish assemblages (excluding pelagic species).

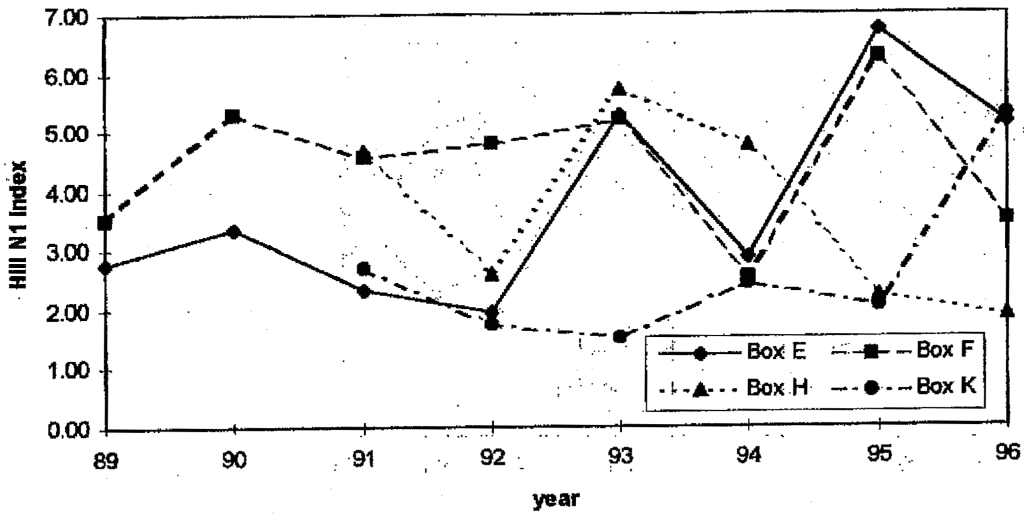


Figure 3.2.3.2.1.5. Box E–K. Annual variation of Hill N1 diversity index from 1989 to 1996 of fish assemblages (excluding pelagic species).

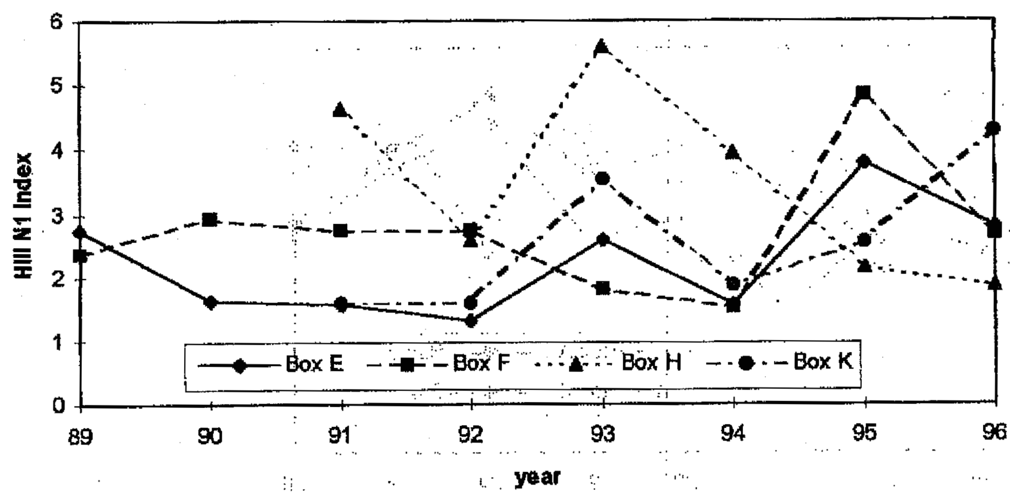


Figure 3.2.3.2.1.6. Box A–D. Annual variation of Hill N2 diversity index from 1987 to 1996 of fish assemblages (including pelagic species).

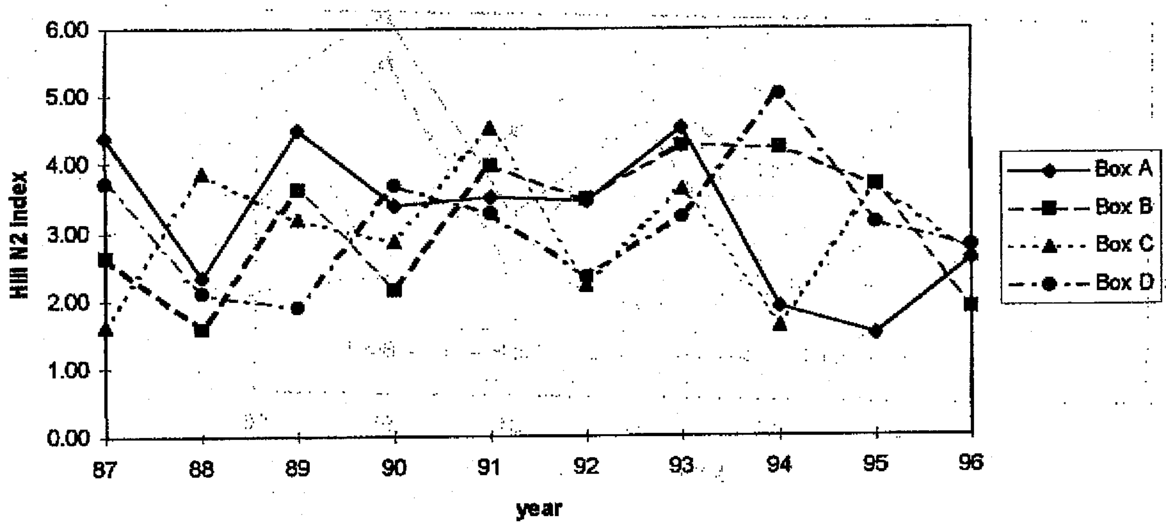


Figure 3.2.3.2.1.7. Box A–D. Annual variation of Hill N2 diversity index from 1987 to 1996 of fish assemblages (excluding pelagic species).

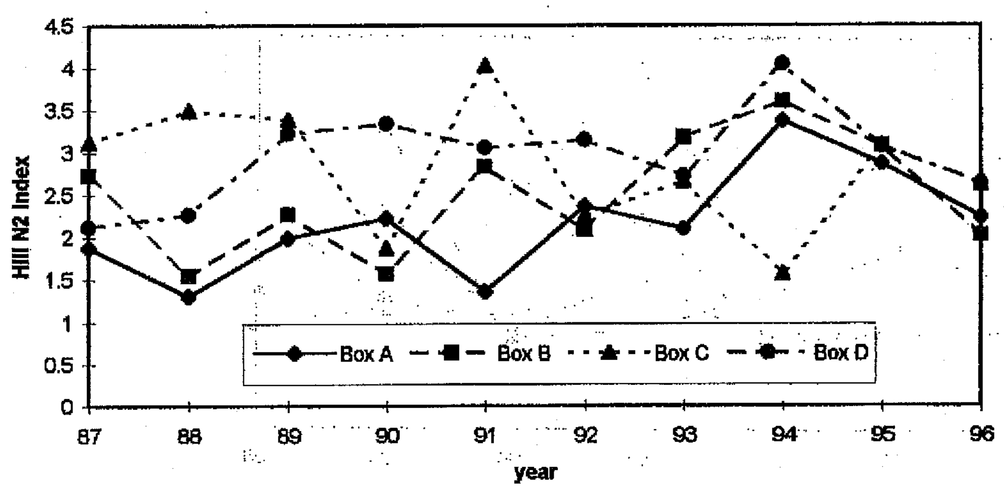


Figure 3.2.3.2.1.8. Box E–K. Annual variation of Hill N2 diversity index from 1989 to 1996 of sish assemblages (including pelagic species).

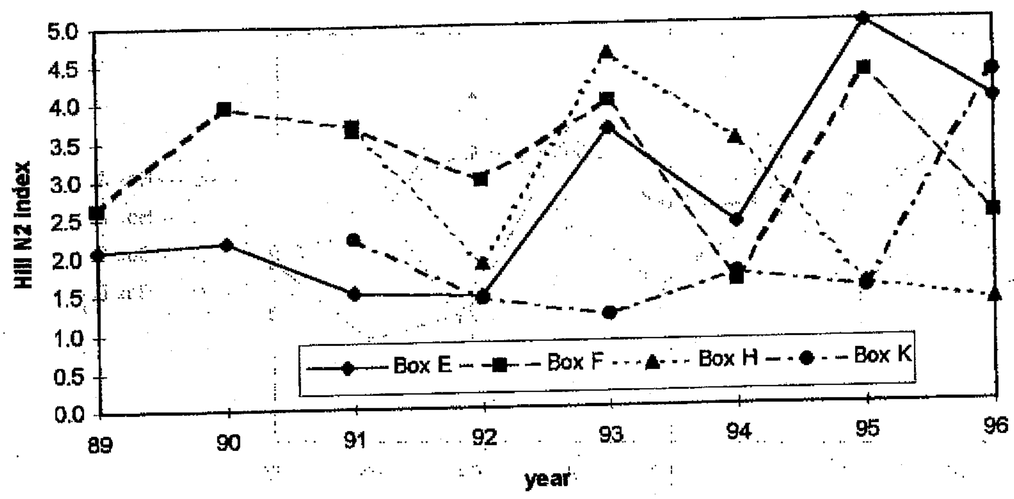


Figure 3.2.3.2.1.9. Box E–K. Annual variation of Hill N2 diversity index from 1989 to 1996 of sish assemblages (excluding pelagic species).

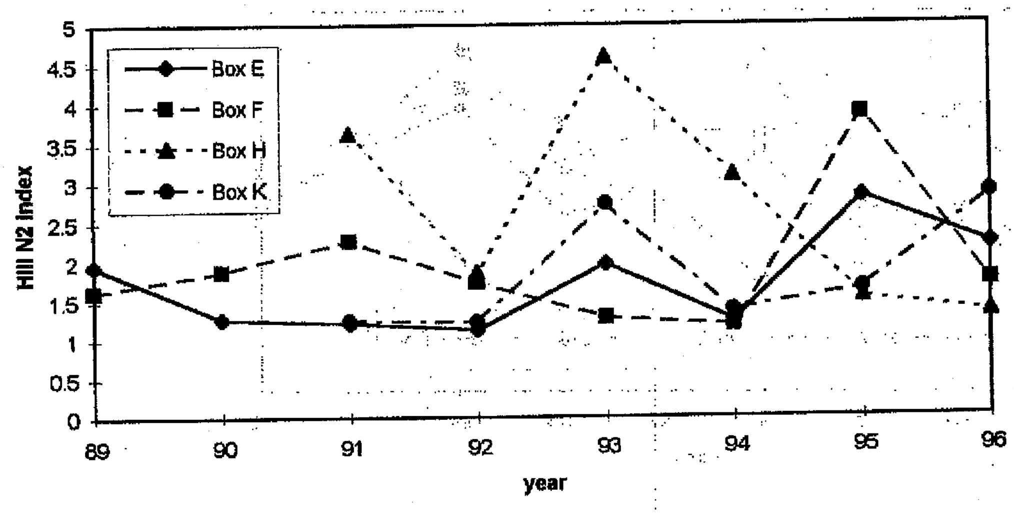


Figure 3.2.3.2.1.10. Changes in mean individual weight in the boxes A and C.

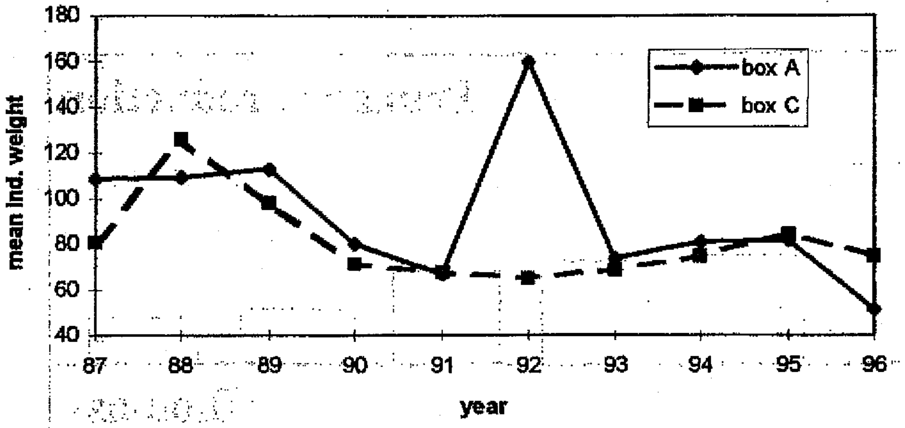


Figure 3.3.1.3.1. Summary statistics of the seven balanced models constructed, referring to system size. Systems are arranged after decreasing primary production. Note that systems are set apart in geographic rather than in regime-specific order. Also note similar trend in all four parameters primary production, total biomass (exc. detritus), total catches and total system throughput.

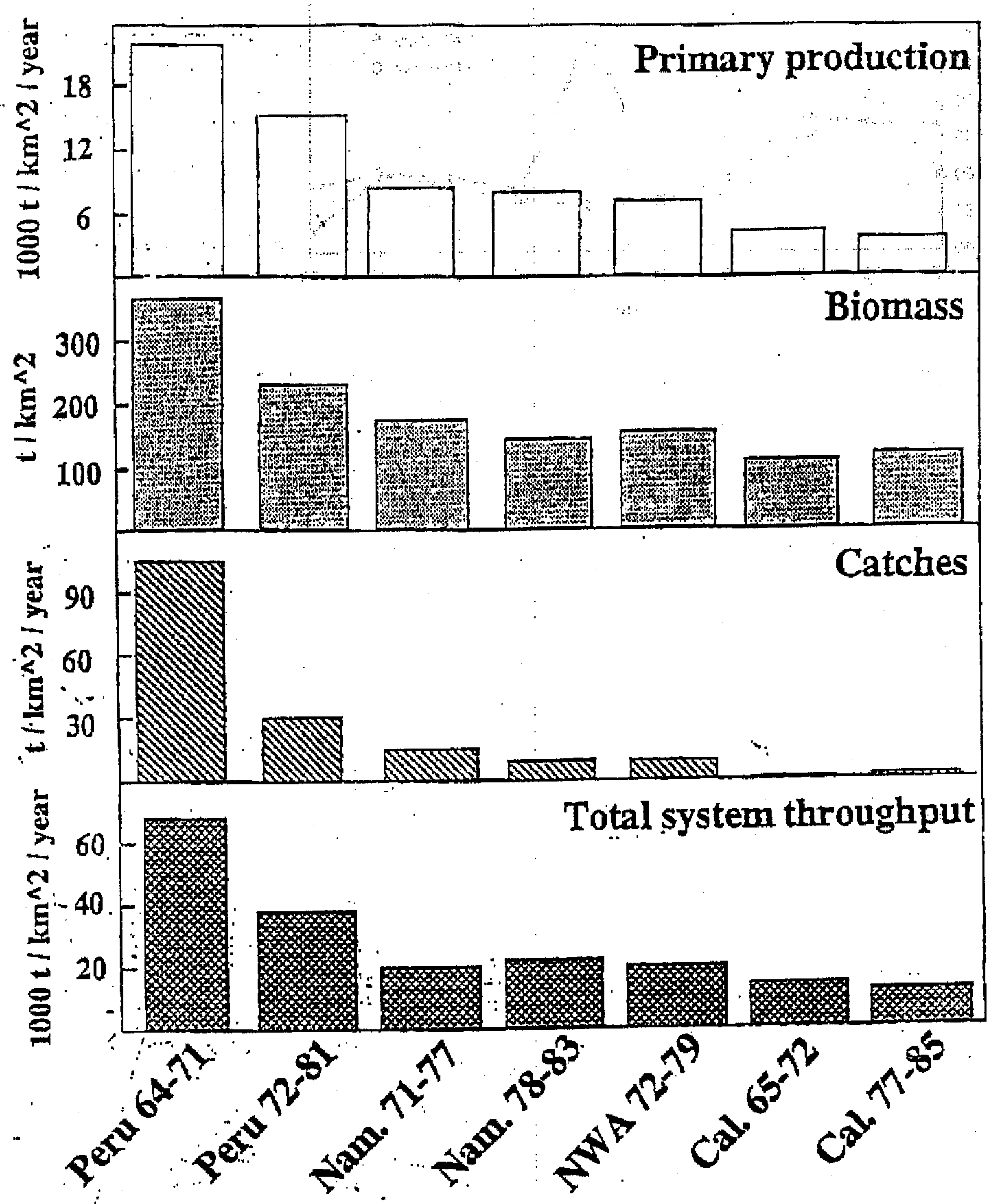


Figure 3.3.1.3.2. Mixed trophic impacts of two potential fish predators, horse mackerel and hake, and marine mammals as top predators, on the five dominant fish species, the mammals and the fishery, in the Peruvian and Californian upwelling ecosystems. Negative impacts are shown below the line, positive ones above.

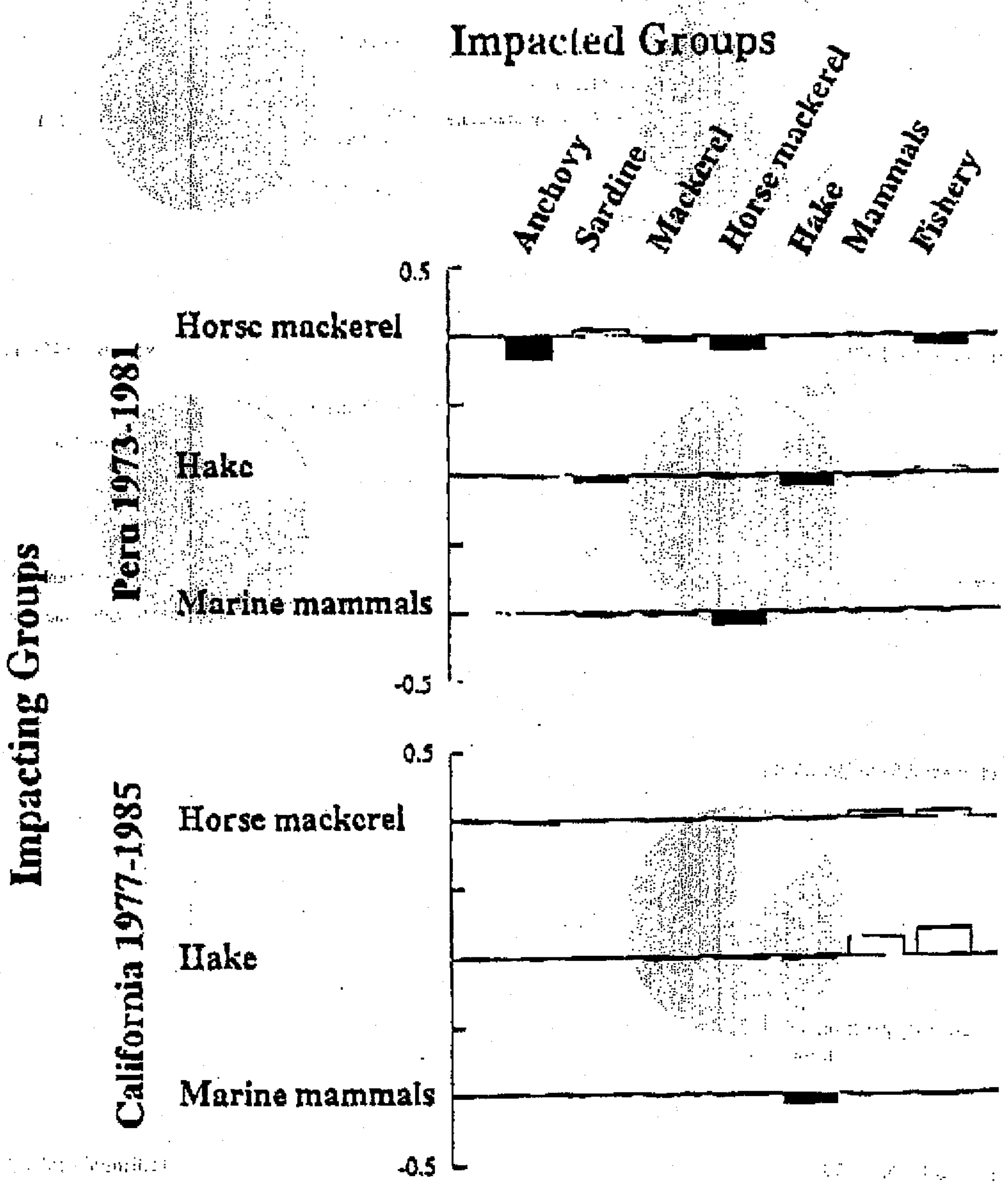


Figure 3.3.1.3.3. Partitioning of the total production of the dive dominant fish species among all fish, mammals, other top predators (marine birds and large scombrids), the fishery and other groups in the four upwelling systems modelled.

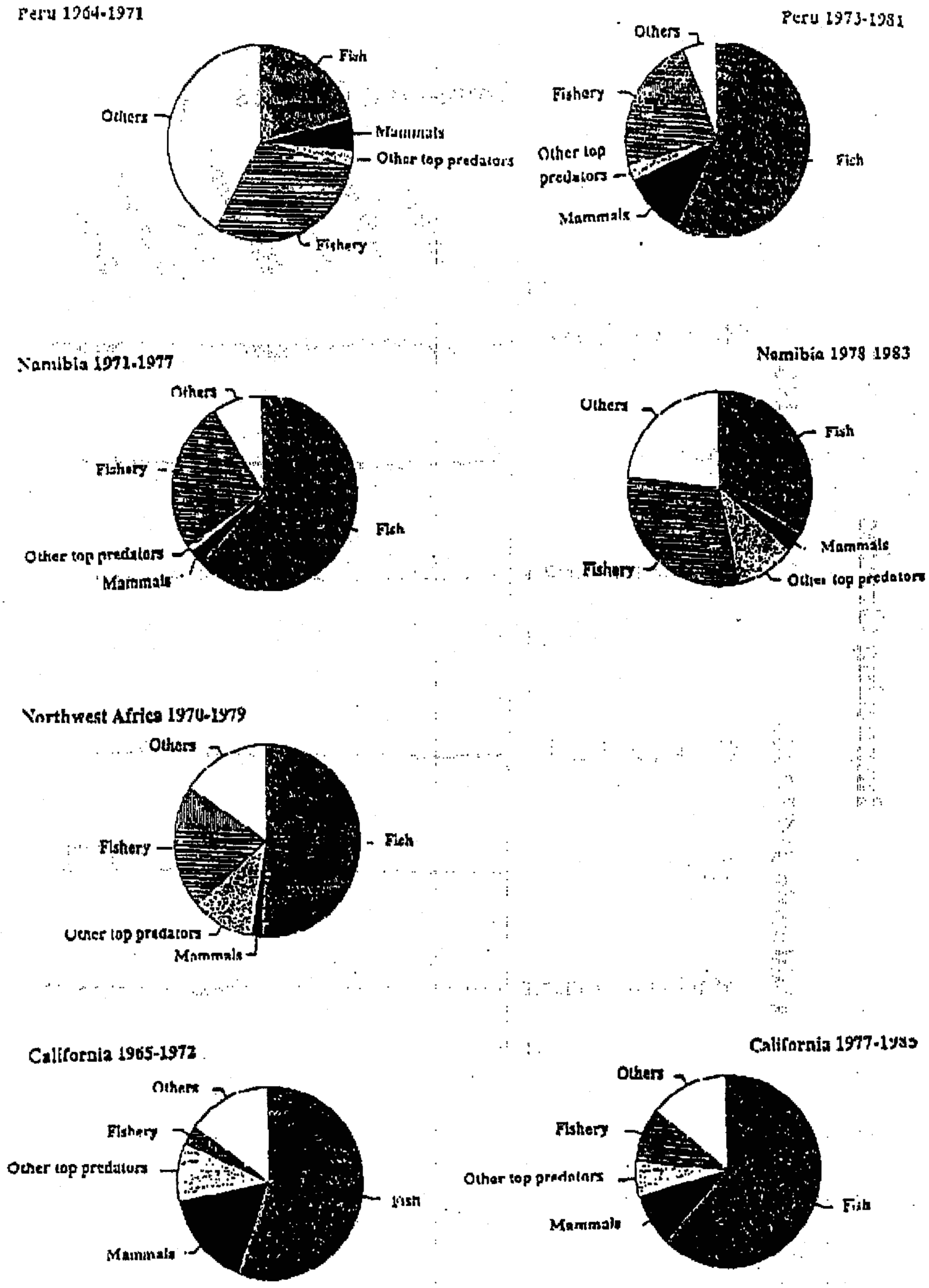


Figure 3.3.1.4. Flows from the producers' level to the fishery, both in absolute (light shading) as well as in relative (dark shading) terms. Systems are arranged after decreasing primary production.

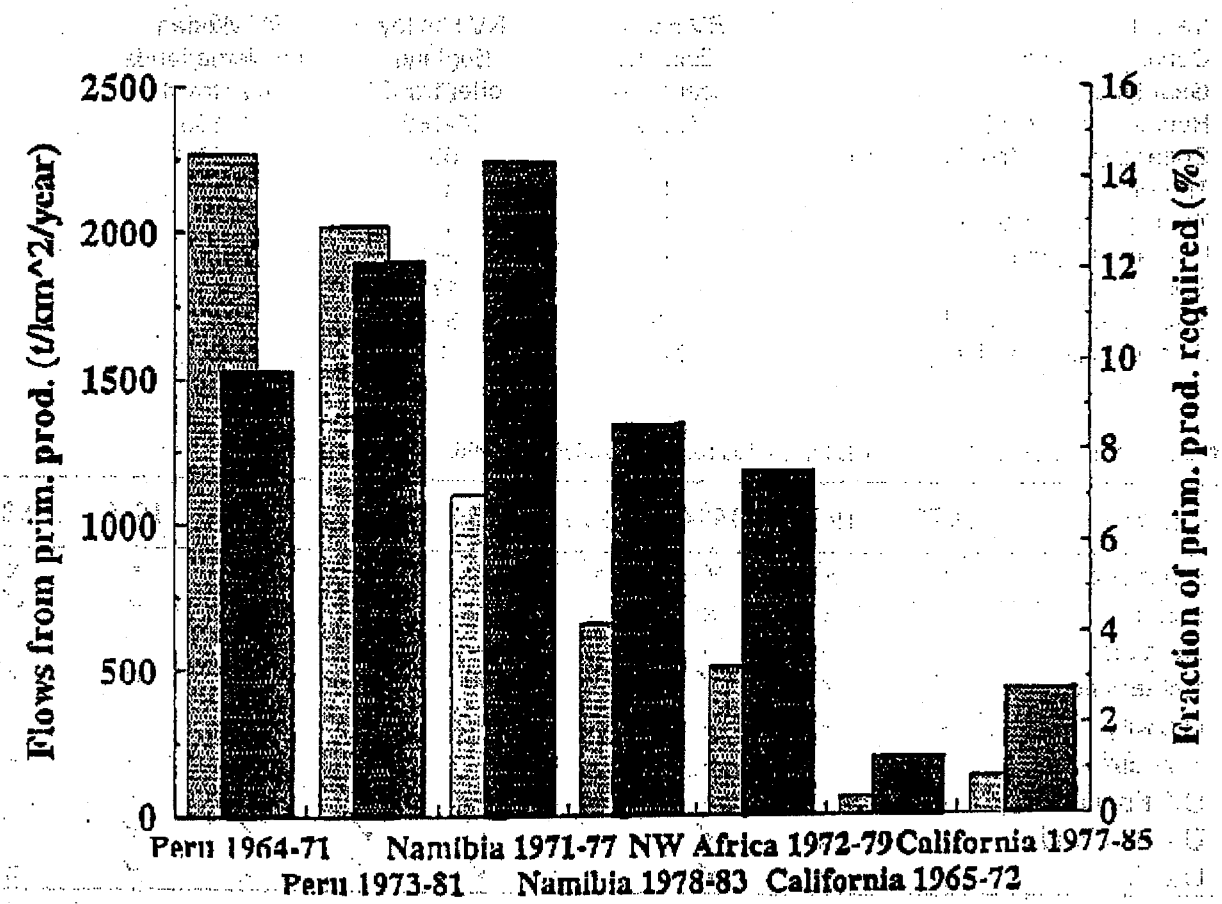


Figure 3.3.2.1. Partitioning of the total fish production in the Baltic Sea by major consumer groups.

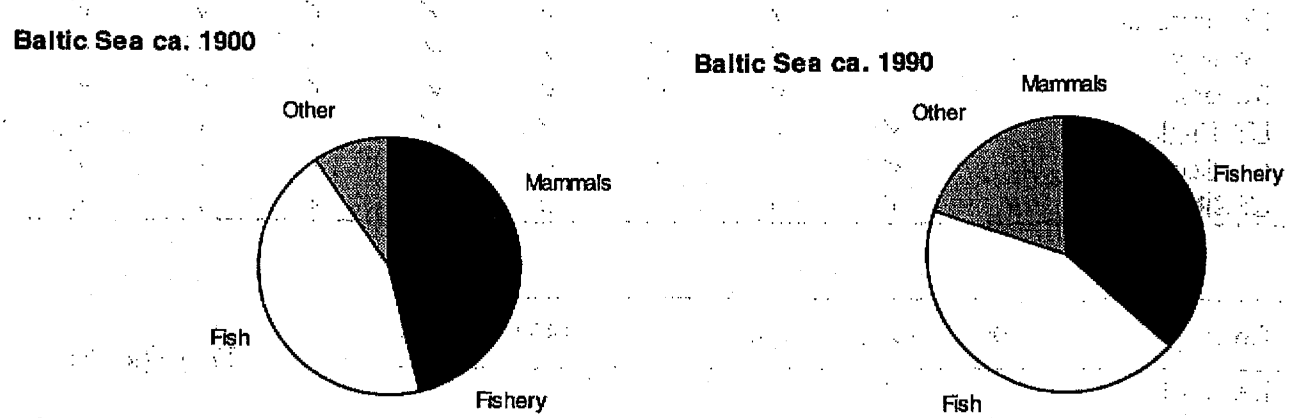


Table 3.1.1.1. Details of survey gear used in trawl surveys between 1906–1909.

Vessel	RV Huxley	RV Huxley	RV Wodan
Country of origin	England	England	The Netherlands
Gear type	beam trawl	otter trawl	otter trawl
Haul duration (min)	60-180	60-180	30-180
Codend mesh (stretched, mm)	63	68	c.40
Tickler chains	0	0	0
Towing speed (knots)	2	2	c.2
Headline length (m)	-	26	6
Sweep (m)	13	17	4
Swept area (1000 m2 h-1)	50	60	15
Relative catch efficiency	0.85	1	0.25

Table 3.1.1.2. Data available in the ICES IBTS data base as of 26 January 1996.

Country	1972	1973	1974	1975	1976	1977	1978	1979	1980
Denmark	✓	✓	✓	✓	✓	✓	✓	✓	x
France	x	x	x	x	✓	x	x	✓	✓
Germany	-	-	-	-	-	-	-	-	-
Netherlands	✓	✓	✓	✓	✓	✓	✓	✓	✓
Norway	-	-	-	-	-	-	-	-	-
Sweden	-	-	-	-	-	-	-	-	-
UK England	-	-	-	-	-	-	-	-	-
UK Scotland	x	x	✓	✓	✓	✓	✓	✓	✓
USSR	x	x	✓	✓	✓	✓	x	✓	x

Country	1981	1982	1983	1984	1985	1986	1987	1988	1989
Denmark	x	✓	✓	✓	✓	✓	✓	✓	✓
France	x	✓	✓	✓	✓	✓	✓	✓	✓
Germany	-	-	✓	✓	✓	✓	✓	✓	✓
Netherlands	✓	✓	✓	✓	✓	✓	✓	✓	✓
Norway	-	-	✓	✓	✓	✓	✓	✓	✓
Sweden	-	-	✓	✓	✓	✓	✓	✓	✓
UK England	✓	✓	✓	✓	✓	✓	✓	✓	✓
UK Scotland	✓	✓	✓	✓	✓	✓	✓	✓	✓
USSR	✓	✓	x	x	x	x	x	x	x

Country	1990	1991	1992	1993	1994	1995
Denmark	✓	✓	✓	✓	✓	✓
France	✓	✓	✓	✓	✓	✓
Germany	✓	✓	✓	✓	✓	✓
Netherlands	✓	✓	✓	✓	✓	✓
Norway	✓	✓	✓	✓	✓	✓
Sweden	✓	✓	✓	✓	✓	✓
UK England	✓	x	x	x	x	x
UK Scotland	✓	✓	✓	✓	✓	✓
USSR	x	x	x	x	x	x

First Quarter.

✓ = Data available

\- = No data available

x = No survey made

Table 3.1.1.3. Summary of all beam trawl surveys targeted on sole and plaice in the North Sea and English Channel, describing their specification (ICES, 1990).

Country		Area surveyed	Start year	Survey date	Gear used	Target species
Belgium		Belgian coast	1970	August - October	6 m BT	0, 1 sole, plaice
"	*	S North Sea	1985	August - October	8/4 m BT	1, 2, 3+ sole, plaice
France		French coast, VIId	1977	August - October	2.7/4.5 m BT	0, 1 sole, plaice
Germany		German coast, N. Sea	1972	August - October	3 m BT	0,1 sole, plaice
"	*	German Bight	1976	June	7 m BT	2,3+ sole, plaice
UK		English coast, N. Sea	1974	September	2 m BT	0,1 sole, plaice
"	*	VIId	1988	August - September	4 m BT	1,2,3+ sole, plaice
"	*	VIIe (English coast)	1984	August - September	6/4 m BT	1,2,3+ sole, plaice
The Netherlands		Dutch and Danish coast, N. Sea	1969	August - October	6 m BT	1,2+ sole, plaice
"		Wadden Sea, Scheldt, Dutch coast	1970	August - October	3/6 m BT	0,1+ sole, plaice
"	*	Southern N. Sea	1985	August - October	8 m BT	1,2,3+ sole, plaice

Table 3.2.2.1.4.1. Individual sum-of-squares tables for linear models of ln(size-class), year, depth stratum, and ln(size-class) within depth stratum and year, with and without interaction, fit to ln(numbers). Zero samples treated as missing values (16 occurrences for demersal + pelagic species, 19 for demersal only).

Source	Df	Sum of Sq	Mean Sq	F Value	Pr(F)
a) Demersal + pelagic species					
Model with interaction					
Lt	1	3424.647	3424.647	5640.209	0.00000000
Stratum	6	23.525	3.921	6.457	1.47281e-6
Year	6	45.108	7.518	12.382	0.00000000
Stratum:Year	36	52.718	1.464	2.412	1.63198e-5
Lt %in% (Stratum*Year)	48	59.716	1.244	2.049	9.05059e-5
Residuals	474	287.805	0.607		
Model without interaction					
Lt	1	3424.647	3424.647	5468.265	0.0000000
Stratum	6	23.525	3.921	6.260	0.0000023
Year	6	45.108	7.518	12.004	0.0000000
Stratum:Year	36	52.718	1.464	2.338	0.0000303
Lt %in% Stratum	6	19.244	3.207	5.121	0.0000401
Lt %in% Year	6	8.876	1.479	2.362	0.0292662
Residuals	510	319.401	0.626		
b) Demersal species only					
Model with interaction					
Lt	1	1152.996	1152.996	3554.789	0.00000000
Stratum	6	36.899	6.150	18.960	0.00000000
Year	6	55.717	9.286	28.630	0.00000000
Stratum:Year	36	40.490	1.125	3.468	3.9547e-10
Lt %in% (Stratum*Year)	48	75.534	1.574	4.852	0.00000000
Residuals	471	152.769	0.324		
Model without interaction					
Lt	1	1152.996	1152.996	3350.404	0.0000000
Stratum	6	36.899	6.150	17.870	0.0000000
Year	6	55.717	9.286	26.984	0.0000000
Stratum:Year	36	40.490	1.125	3.268	0.0000000
Lt %in% Stratum	6	48.657	8.110	23.565	0.0000000
Lt %in% Year	6	5.168	0.861	2.503	0.0213910
Residuals	507	174.477	0.344		

4 LEVELS OF PREDATION ON BENTHOS (CF) SUBSTRUCTURE DETERMINED BY GROUP (TOR B)

4.1 Introduction

The effect of fisheries on target species is well established (Pope and Macer, 1996; Rijnsdorp and Millner, 1996). The direct mortality of benthos arising from the use of heavy gears has also been demonstrated (e.g., Lindeboom and de Groot, 1997). The indirect ecosystem effects of fishing are less clear. These include, changes in nutrient cycling caused by physical disturbance of the sediment water interface and the addition of offal to the system, the consequences of the continued removal of fixed carbon from the marine to the terrestrial system and the changes in the food chain arising from manipulation of the density and size structure of the target populations.

Exploitation of fish stocks has altered the abundance of fish in the seas and, frequently, the size composition of the fish populations (Pope and Knights, 1982; Pope *et al.*, 1988). Marine communities frequently exhibit size structured food webs and these changes are therefore likely to lead to changes in the quantities and types of prey consumed. One of the most dramatic examples of ecosystem changes arising from fisheries exploitation of the controlling predators are the changes in the abundance and utilisation of krill in the Antarctic following exploitation of the baleen whales (Dayton *et al.*, 1995). Other examples, which demonstrate a cascade of effects through the ecosystem following heavy fishing mortality or destructive fishing practices include the changes induced in the Californian kelp forests following hunting of the sea otters (Simenstad *et al.*, 1978; Estes, 1996), the changes in the intertidal areas of Chile induced by the removal of predators by fishers (Moreno *et al.*, 1986) and the changes induced by the rapid development of the demersal fishery in the Gulf of Thailand (Pauly, 1988). The widespread nature of fishing, and the migratory behaviour of many species, means that there are no opportunity for comparisons between exploited regions and unexploited controls. Therefore these studies all provide only indirect evidence of cause and effect and rely on inferential arguments and deductive reasoning (Dayton *et al.*, 1995).

In the North Sea, populations of the benthic feeding gadoids, cod, haddock and whiting, have declined over the last 30 years or so (Pope and Macer, 1996, Serchuk *et al.*, 1996). During the same period populations of long rough dab, common dab and lemon sole have increased (Heessen and Daan, 1996). Therefore there is a need to provide an assessment of the consequences of these changes for the level of predation pressure exerted on the benthos.

4.2 Methodology and Approach

4.2.1 Trends in stock biomass

For eight benthic feeding species time series of abundances (biomass) and feeding rate data were available. These were: sole (*Solea solea*), plaice (*Pleuronectes platessa*), cod (*Gadus morhua*) haddock (*Melanogrammus aeglefinus*), whiting (*Merlangius merlangus*), long rough dab (*Hippoglossoides platessoides*), common dab (*Limanda limanda*) and lemon sole (*Microstomatus kitt*).

VPA estimates of total stock biomass and stock biomass at age for sole, plaice, cod, haddock, and whiting were available from the ICES Working Group on Assessment of Demersal Stocks in the North Sea and Skagerrak (ICES, 1998). Time trends in stock biomass prior to the time period covered by the assessment group were taken from Pope and Macer (1996) for cod, haddock and whiting; and Rijnsdorp and Millner (1996) for plaice. Details of the calculation of stock biomass for these species are given below.

Plaice. As discarding of plaice is high, about 50 % by numbers (van Beek, 1990), the estimates of stock biomass of plaice from the assessment working group were corrected. Discarding mainly affects age groups 1, 2, and 3 and the VPA stock numbers were, therefore, corrected by multiplying these by a factor of 1.687, 1.495, and 1.018, respectively, taken from Table 9.4.3 run B in ICES (1998). Stock weights at age in the VPA were based on 1st quarter catch weights (cw). In order to get more realistic stock weight at age (swi), the weights were calculated as $swi = (cwi + cwi + 1)/2$; giving stock weights for age groups 1, 2, and 3 of $sw1 = 0.0328$ kg, $sw2 = 0.107$ kg, and $sw3 = 0.197$ kg, respectively. As the correction for discards and the corrected stock weights worked in opposite directions, the corrected time series of stock biomass was only marginally different from the original one given by the ICES working group (ICES, 1994) (Figure 4.2.1). For the periods 1929–1938 and 1948–1994, a time series of stock biomass was available from Rijnsdorp and Millner (1996). The estimates between 1957 and 1994 were close to the values obtained above. Thus, the stock biomass estimates prior to 1957 were accepted as the best available estimates even though these did not take account of the discarding. As such they may underestimate the actual stock biomass.

Sole:

Stock biomass at age 1 to 7+ were calculated from the VPA results given in Table 7.4.5 of ICES (1998). As there is no substantial discarding in this species, the reconstructed stock numbers should give an accurate estimate of the numbers at age in the stock. Stock weights in the analysis were the catch weights during the second quarter spawning period.

Cod, haddock, and whiting

VPA estimates for stock biomass (ICES, 1998) and spawning stock biomass values (Pope and Macer, 1996; see their Figures 4–6) were used for a regression analysis on the relationship between spawning stock biomass and stock biomass during the period 1963–1996. This relationship was then used for hindcasting the stock biomass for the period prior to 1963 (1920–1937 and 1945–1962). Current high levels of fishing mortality lead to a mean population around 30 % less than the 1 January population. However, over the timeseries considered here fishing mortality has varied greatly. As the purpose of this exercise was to assess changes in the levels of predation on the benthos, rather than accurately estimating predation per se, the resulting over estimate of the quantity of benthos consumed was seen as being conservative for this purpose.

The North Sea abundance of long rough dab, common dab, and lemon sole were calculated from the IBTS database by Heessen and Daan (1996) for the period 1970–1993. These estimates are utilised here. Daan *et al.* (1990) provide total stock biomass estimates for these species over the period 1977–1986 (1985 for long rough dab). The relationship between these and the abundance data were used to estimate total stock biomass for the reminder of the series.

4.2.2 Diet and daily rations

The most comprehensive data on fish diets for species utilised in the VPAs comes from the 'year of the stomach' programmes. These sampled in 1981 and 1991. Greenstreet *et al.* (Working paper No: 11) has shown that the estimated consumption of benthos varies little whether 1981 or 1991 stomach data are used. As the earliest diet data are based on the 1981 'year of the stomach surveys', if benthic prey abundances have changed over time, whether the result of fishing or not, the data may not accurately reflect predator prey selection earlier in the twentieth century. It is therefore acknowledged that these rates are unlikely to be representative of earlier times when the composition of the benthos was probably different and so the results presented here must be used with caution.

4.2.3 Gadoids

Diet compositions for cod, haddock, and whiting were derived from Tables 2–4 in Greenstreet (1996) which are based on the 1981 stomach sampling programme. In these tables, diet compositions are given as percentage values for each quarter of the year and for seven age groups (0, 1, ..., 6+). To derive annual consumption rates (gram prey per fish), these diet proportions were combined with quarterly consumption data from Table 6 in Greenstreet (1996).

Daily rations for all three gadoid species were around 1.5 % body weight per day.

4.2.4 Flatfish

Diet data for plaice and sole were taken from Rijnsdorp and Vingerhoed (in prep.), while for long rough dab we used data from Ntiba and Harding (1993). For lemon sole and common dab, data were derived from Tables 19 and 20 in Greenstreet (1996). These data were pooled to an annual average taking account of quarterly differences in the percentage feeding. Annual food consumption rates for lemon sole and dabs were derived from data in Greenstreet (1996; Table 21), suggesting an average daily consumption of 1 % of their body weight (irrespective of age). For plaice and sole, we used Greenstreet's Table 16 to estimate the annual consumption for five age groups (0, 1, 2 & 3, 4–6, 7+). The average daily ration was 2.8 % of the body weight, consistently higher than that of the other species.

4.3 Results

4.3.1 Flatfish

4.3.1.1 Trends in stock biomass

Common dab had the highest stock biomass of all flatfish species (Figure 4.3.1.1.a). Over the period 1970 to 1993 the stock biomass of dab varied greatly between years. Some of this variability is likely to be an artifact of the use of survey data to generate the stock biomasses. The high biomass is consistent with other estimates. Daan *et al.* (1990) for example report the mean biomass for 88 fish species in the North Sea, and the common dab had the greatest biomass.

The biomass of plaice (Figure 4.3.1.1.a) was greatest in the years 1967 and 1988 but has declined in recent years. The biomass over the period 1929 to 1938 was somewhat lower than from 1948 and until recent years. Long rough dab has shown a steady increase in biomass during the period 1970 to 1993.

The stock biomass of sole showed very little changes over the period 1957 to 1993 (Figure 4.3.1.1.b). The stock biomass of lemon sole, estimated from trawl surveys, has increased sharply over the period and estimates varied greatly from year to year.

4.3.1.2 Diet composition

Diet of plaice and sole were rather similar (Figure 4.3.1.2). Fish contributed less than 10 % in the diet. Of the benthic invertebrates, annelids were dominant in the diet of both plaice and sole. The proportion of annelids decreased with age, whereas that of echinoderms and molluscs increased. The proportion of crustaceans was relatively small (< 10 %).

The diet of dab was dominated by echinoderms (mainly ophiuroids) and to a lesser extent crustaceans. Long rough dab mainly took crustaceans and annelids. The diet of lemon sole was dominated by annelids (Table 4.3.1.2).

Table 4.3.1.2. Diet composition (%) for common dab, long rough dab, and lemon sole (see text for sources).

	Common dab	Long rough dab	Lemon sole
Echinodermata	50	4	2
Crustacea	37	48	5
Annelida	10	32	58
Mollusca	1	5	11
Fish	0	11	1
Other benthos	2	0	24

4.3.1.3 Food consumption

The quantity of benthos consumed by flatfish in the North Sea has increased steadily over the period 1970–1993 (Figure 4.3.1.3.a). While the quantity removed by plaice has remained fairly constant, the expansion of the common dab population has lead to them being the dominant consumer in this group (Figure 4.3.1.3.b). There are no clear trends in the composition of the fauna consumed, year on year variations in the relative abundance of the various species of flatfish driving the patterns of prey taken (Figure 4.3.1.3.a).

4.3.2 Gadoids

4.3.2.1 Trends in stock biomass

Over the period 1929 to 1939 there was a sharp decrease in the stock biomass of cod (Figure 4.3.2.1.a). Between the years 1944 and 1980, the biomass showed considerable inter-annual variation but no trends were apparent. In the last ten years or so, a decline in the stock biomass is evident.

The biomass of haddock decreased gradually over the period 1920 to 1963 (Figure 4.3.2.1.b). From 1963 to present time the biomass was more variable but at a higher mean level. Stong year classes lead to peaks in biomass in 1963 and 1968.

Over the period 1920 to 1939 the stock biomass of whiting was low compared to the period after the Second World War (Figure 4.3.2.1.c). As with cod, whiting biomass has decreased during the last decade.

4.3.2.2 Diet composition

The proportion of benthic invertebrates in the diet of the gadoids showed a clear decline from about 50 % in age 1 to less than 10–30 % in age group 6+ (Figure 4.3.2.2). Haddock showed the highest relative proportion of benthic invertebrates in their diet (Figure 4.3.2.2.b), followed by cod (Figure 4.3.2.2.a), and whiting (Figure 4.3.2.2.c). Within the benthos there was some difference in the prey types selected. Cod and whiting mainly took crustaceans, whereas haddock took crustaceans, echinoderms, and annelids in near equal proportions. Molluscs were insignificant in all three gadoid diets (Figure 4.3.2.2).

4.3.2.3 Food consumption

Gadoid consumption of food peaked in the late 1960s (Figure 4.3.2.3.a) at around 39 million t annually, of which approximately 17 million t was benthos (Figure 4.3.2.3.b). Quantities of food consumed declined from 1968 to 1989 in line with the decreasing stock sizes. Compared to the values used by the ICES Multispecies Assessment Working Group (ICES CM 1997/Assess:16) the values for gadoid consumption calculated here are higher. According to their Table 3.1.2.1 cod, haddock, and whiting would each consume between 2.3 and 3.5 million t of 'other food' in the period from 1990 to 1995.

The benthic prey were dominated by crustaceans (~55 % of the benthic food, by weight), but annelids and echinoderms also made important contributions to the diet (~20 % each). Molluscs represented a minor dietary component (~5 %) (Figure 4.3.2.3.a). Comparisons of the taxonomic composition of the consumed material must be regarded as suggestive only given the underlying assumptions. Since species and size specific diets were assumed to be constant, variation in the total consumption of different prey types was caused entirely by variation in abundance and size composition of the gadoids over the time series. Given this, the consumption of echinoderms during the period from the second world war to the early 1960s was only about half that for the remainder of the time series (Figure 4.3.2.3.a). The echinoderms were mainly replaced by crustaceans.

4.3.3 Combined effects and implications for system productivity

The data sets considered here allow an evaluation of the fish predation pressure over the period 1970–1993 for eight of the most abundant demersal species. In spite of the declines in target fish populations (gadoids and plaice), the overall level of predation on the benthos has increased from around 23 million t year^{-1} in 1970 to 29 million t year^{-1} in 1993 (Figure 4.3.3). In addition, there are indications of a decrease in the proportion of crustaceans and molluscs in the diet and an increase in the importance of echinoderms (primarily ophiouroids).

To evaluate the potential effect on the benthic fauna from the predation by the eight fish species, we compared estimates of production and consumption. Due to the limited extent of published data, it was possible to make this comparison only at a coarse taxonomic level. Production estimates for North Sea benthos were derived from two sources; Christensen (1995, using his groups 'echinoderms', 'polychaetes', and 'other macrobenthos') and Greenstreet *et al.* (1997). Based on these data, the predation rate of fish on benthos appears high (20–45 % of the benthic production being used by the eight fish included in this analysis, Table 4.3.3.1).

Table 4.3.3.1. Annual production and consumption rates (1970–1993) by benthic invertebrates and fish in the North Sea. The consumption to production ratio is given in the right column.

	in '000 tonnes per year	*consumption / production*
Benthic production estimates		
based on Christensen (1995)	119700	0.19
based on Greenstreet *et al.* (1997)	51152	0.44
Consumption by fish		
based on basic calculations for the 8 species included in this study	22698	

4.4 Discussion

Fish predation is commonly seen to be the principal way that fish influence benthic communities (e.g., Whitman and Sebens, 1992; Sala and Zabala, 1996; Sala, 1997). Benthic feeding fish do not take prey in proportion to their availability, they exhibit some degree of selection (e.g., Packer *et al.*, 1994), thereby altering relative abundances of benthic species. The removal of preferred prey may release resources for utilization by other, less preferred, species while the act of predation may cause small scale physical disturbance to the system and contribute to the spatial heterogeneity of the benthos (Hall, 1994). The most incontrovertible evidence of these effects comes from experimental studies which have shown that fish predation can act to control both the number of individuals in the system and the relative abundance of the species (see Wilson, 1990). However, this is not always the case and in certain areas the abundance and species composition of the benthos has been shown to be controlled by factors, such as physical disturbance, emigration/immigration, or benthic predators, rather than fish predation (Ambrose, 1984, 1991). The target species of fisheries can also structure the benthic community through indirect interactions, such as by predation on in-coming larvae (Langton and Robinson, 1990). Hence alterations in the abundance and size structure of target populations can potentially influence the benthos through direct competitive or predative interactions and by indirect routes.

This study has demonstrated that the consumption of North Sea benthos may have changed as stock sizes have changed (Figure 4.3.3). The principal factor influencing fish stock size of exploited species is fishing, and that the expansion in the, non-target, dab population may be due to competitive or predatory release. There is therefore a case for believing that the observed changes in benthos consumption have resulted from the increase in fishing mortality on the target species. Given that demersal fish biomass has decreased the increase in predation on the benthos may seem surprising. However, fishing has removed the larger gadoids, whose diet was principally piscivorous and allowed expansion of flatfish and young gadoids which prey upon benthos to a greater extent (c.f. Figures 4.3.1.2 and 4.3.2.2). However, the differences in diet of the various species would also appear to have influenced the composition of the benthos consumed. Overall, crustaceans have declined in importance while echinoderms (predominately ophiouroids) have increased.

This finding must be interpreted with caution as the composition of the benthos in the diet used in the model formulation are based on a studies which have sampled in the recent past (Ntiba and Harding, 1993; Greenstreet, 1996; Rijnsdorp and Vingerhoed, in prep.). Therefore they take no account of long term changes in the composition of the benthos arising from natural temporal trends, climate driven variation, the changing levels of fish predation or direct impacts of fishing activities. The diet composition of plaice used here closely matched data obtained in the 1980s (de Clerch and Buseyne, 1985). However, a comparison with studies carried out at the beginning of this century (Todd, 1915), suggested a higher proportion of annelids and a lower proportion of molluscs than at present, thereby implying that changes in the composition of the benthos have driven changes in fish diets. Results from studies which have compared the compositon of fauna in the early part of the century in various parts of the North Sea with the contemporary one support this proposition (Riesen and Reise, 1982; Reise, 1982; Kroncke, 1990). These demonstrated shifts in the composition of the fauna towards increased dominance by species which share opportunistic life-history traits. These authors suggest that among sources of anthropogenic disturbances, fishing was probably the dominant factor in causing these changes.

As in all modelling studies the reliability of the findings are directly related to the validity of the underlying assumptions. Three principal assumptions underpin our models. They are that: (i) the composition of the diet has not changed over time (discussed above); (ii) the biomass at age estimate derived from VPA and survey data are valid; and (iii) the total biomass can be used to predict predation levels outside the period covered by VPA.

In the basic calculations we have only included data for the age groups reported in stock assessment reports. For cod and the flatfish, no data were available on the number of age 0 fish and for these species the population consumption rates are therefore underestimates. That this underestimation can be substantial is indicated by the fact that in this study for whiting and haddock, 49 % and 35 % of the population consumption of benthos was attributed to age 0 fish. To some extent this may be mitigated by our use of the stock biomass at 1 January. This gives an overestimate of mean stock size, particularly for recent years when fishing mortality has been high. Given the fact that the former is likely to outweigh the latter our conclusion that fish predation on benthos is intensive and has increased during the last 20 years as fishing has reduced gadoid stocks and dab populations have boomed, is likely to be conservative.

As the only data available prior to VPA were stock biomass estimates, we needed to make assumptions about size composition of these populations. However, Greenstreet *et al.* (Working paper No.11) have shown that, at least for their model construct, the consumption by gadoids is primarily driven by the biomass *per se* rather than age composition of that biomass. If this result is robust then the estimates derived here can validly be used over the entire time series not just the portion derived from VPA.

Estimates of benthic productivity in the North Sea are generally of the order of 51 to 120 million t WW $year^{-1}$ (Greenstreet et al., 1997; Christensen, 1995). Our estimates of the amount of this material consumed by the eight dominant benthivorous fish species (23 million t $year^{-1}$) amounts to less than 45 % of this production.

That predation by fish can strongly influence the prey community structure, is well known for pelagic food webs in freshwater and has also been suggested for marine areas (c.f. Section 8.3.3). For example, in the Baltic Sea zooplanktivorous fish have been estimated to consume more than 50 % of the annual production of dominant crustacean zooplankton (e.g., Rudstam *et al.*, 1992; Arrhenius and Hansson, 1993). Effects of fish predation on benthic communities are, however, less well documented. The intensity of fish predation on North Sea benthos found in this study (20–45 % of the production is consumed by fish), is similar to the 39 % of macrobenthic consumption consumed by fish estimated for this area by Greenstreet *et al.* (1997). However, in the an ECHOPATH model of the North Sea, Christensen (1995) concluded that benthic communities were primarily structured by internal dynamics and not by fish predation. That fish predation on marine benthos can be intensive has also been shown for other areas (e.g., Creutzberg and Duineveld, 1986; Berghahn, 1996).

It must be remembered that in addition to the indirect effects outlined here there are direct effects of fishing on the benthos (ICES, 1988). Recent studies of the EU-DGXIV-funded IMPACT projects (de Groot and Lindeboom, 1994, 1997) showed that the direct mortality caused by beam trawling, estimated as the total mortality associated with one fishing event, was species dependent and varied from 10–40 % in gastropods, starfish, crustaceans, annelid worms and sea mouse, from 10–50 % for the sea urchin *Echinocardium cordatum* and the masked crab *Corystes cassavelaunus*, and from 30ñ80 % for a number of bivalves. At the population level, the mortality imposed by the trawl fishery will depend on the level of direct mortality, the trawling frequency and the overlap in spatial distribution between the fishery and the benthic organisms. Taking account of the patchy distribution of the beam trawl fisheries, annual fishing mortality rates on benthic invertebrates in the heavily trawled southern North Sea were estimated between 7ñ45 % of the individuals (de Groot and Lindeboom, 1994; Piet *et al*, Working Paper 16). Compared to the estimated percentage of the benthic production that is consumed by fish predators (~45 %), the estimated fishing mortality rates are lower. In combination, therefore, direct fishing mortality rates and indirect changes in predation pressure further support the hypothesis that intensive trawling may have caused shifts in benthic assemblages from large slowly reproducing species to small species with a high reproductive rate. As such trawling may have played a role in the increase in growth rate observed in bottom dwelling flatfish (de Veen, 1976; Millner and Whiting, 1996; Rijnsdorp and van Leeuwen, 1996).

The data produced here shown that consumption of benthos by fish predators has changed in both quantity and compostion during the period when fish biomass has been altered by fishing. Alterations, at the ecosystem scale, in the distribution of biomass between compartments and species with ecosystem compartments is likely to have further indirect effects on ecosystem function. These including alteration of the movement of nutrients and carbon around the system (Rowe *et al.*, 1975) and potentially changes in the balance of top-down and bottom up control of the system (Posey *et al.*, 1995).

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Figure 4.2.1. Biomass of plaice estimated from VPA (ICES, 1998) and these estimates after correcting for 'weight at age' for group 1+, 2+, and 3+ fish and for discards.

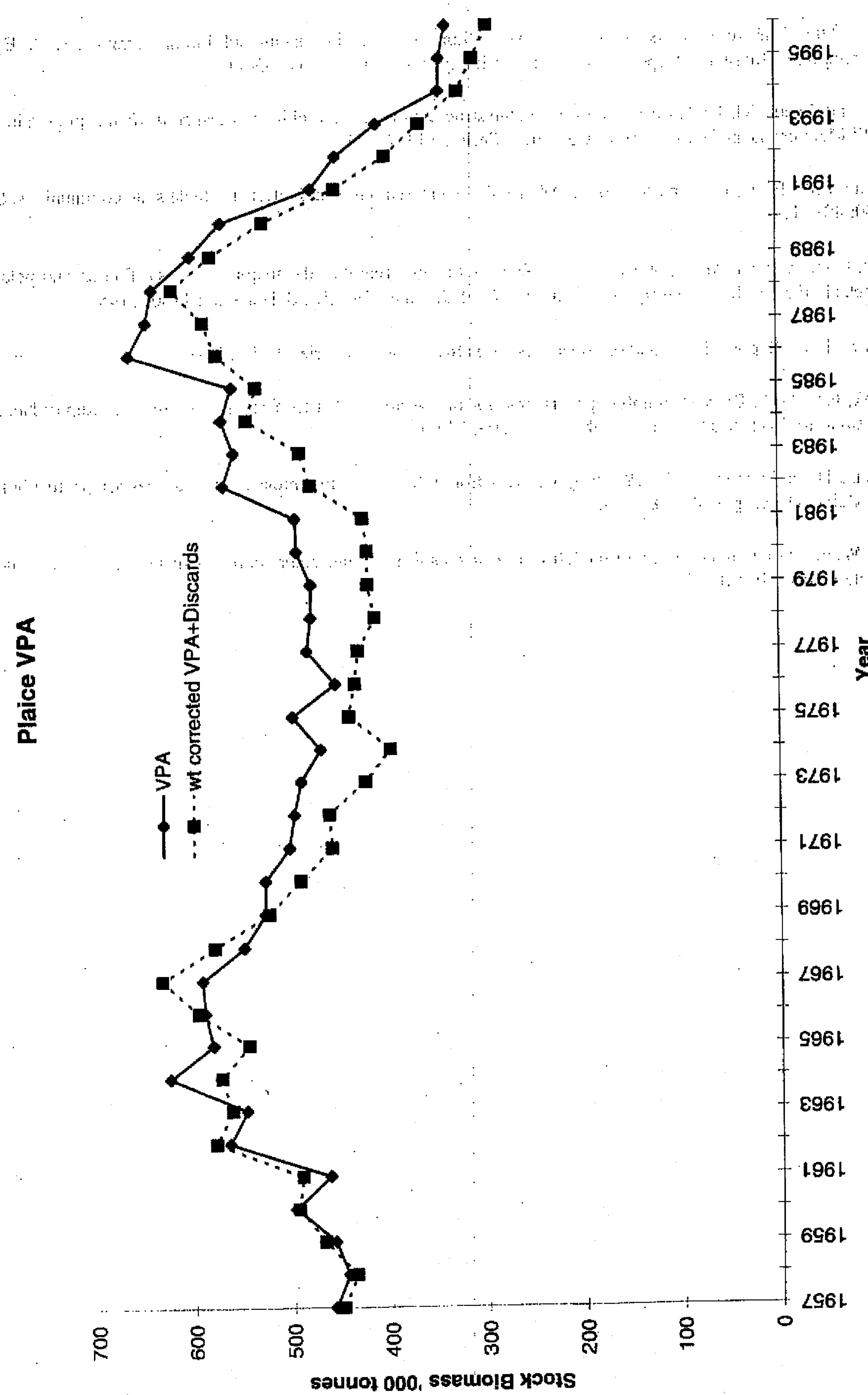


Figure 4.3.1.1.a. Trends in stock biomass for plaice, common dab, and long rough dab. Note change of scale on the biomass axis.

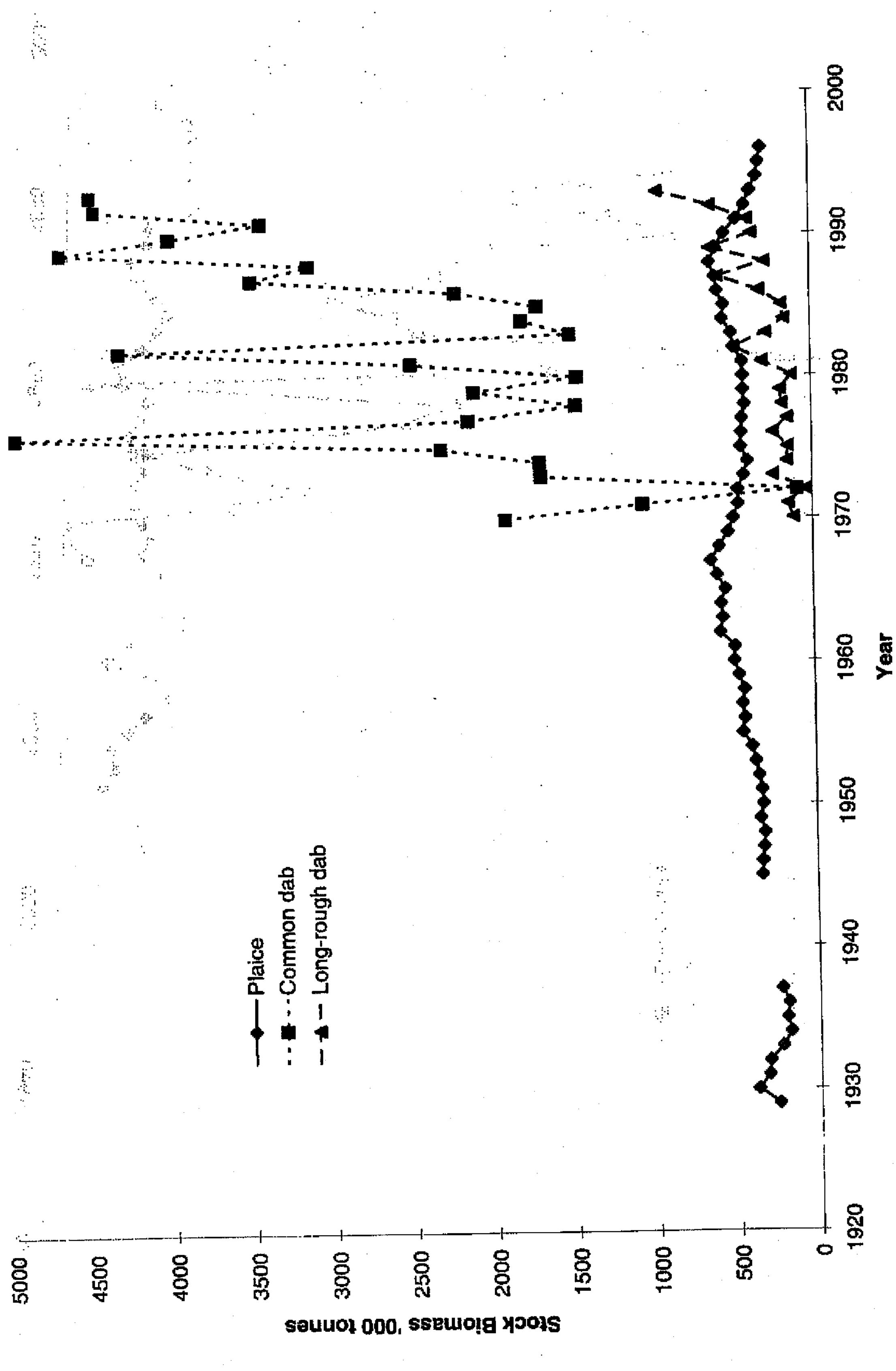


Figure 4.3.1.1.b. Trends in stock biomass for sole and lemon sole. Note change of scale on the biomass axis.

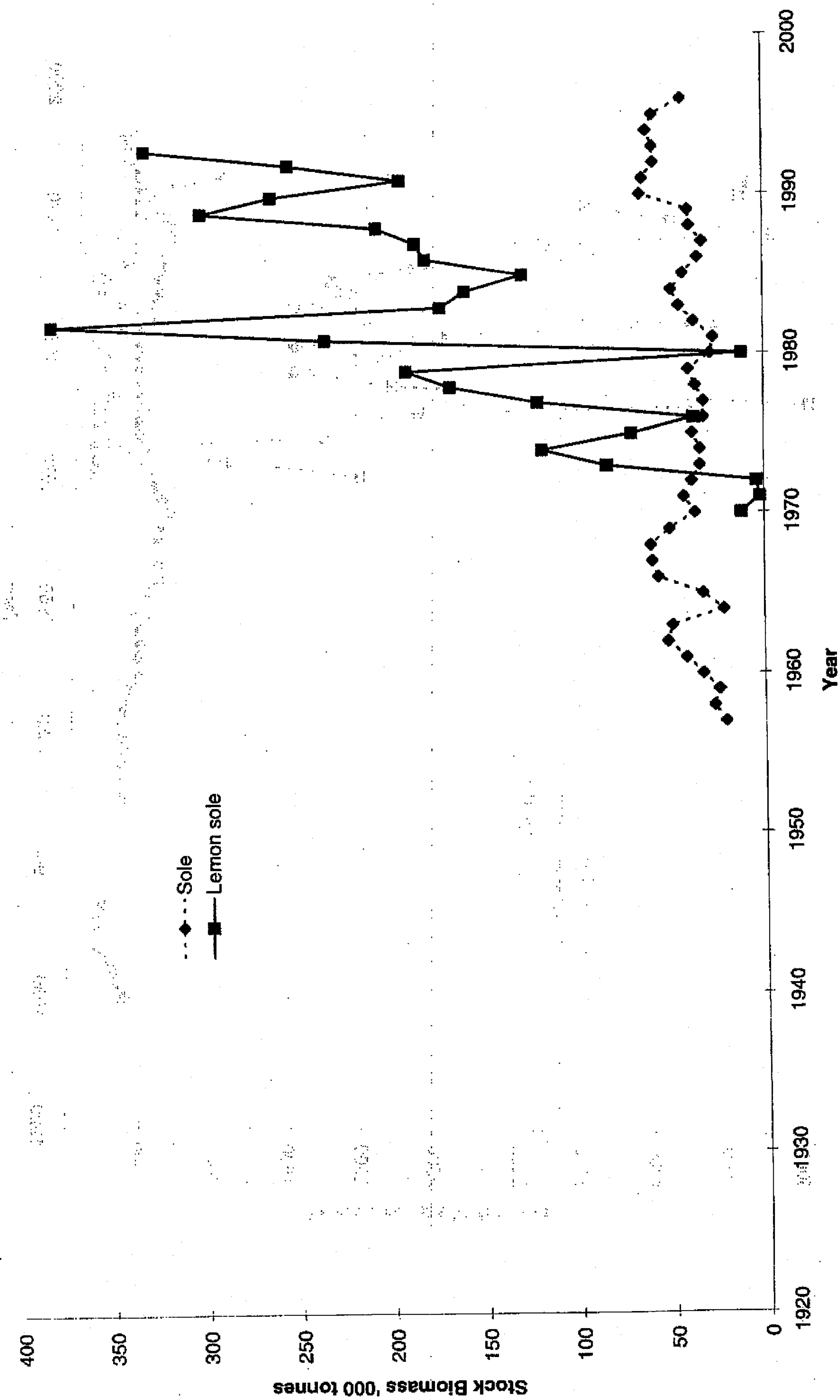


Figure 4.3.1.2.a. Age-specific diet composition of plaice in the North Sea.

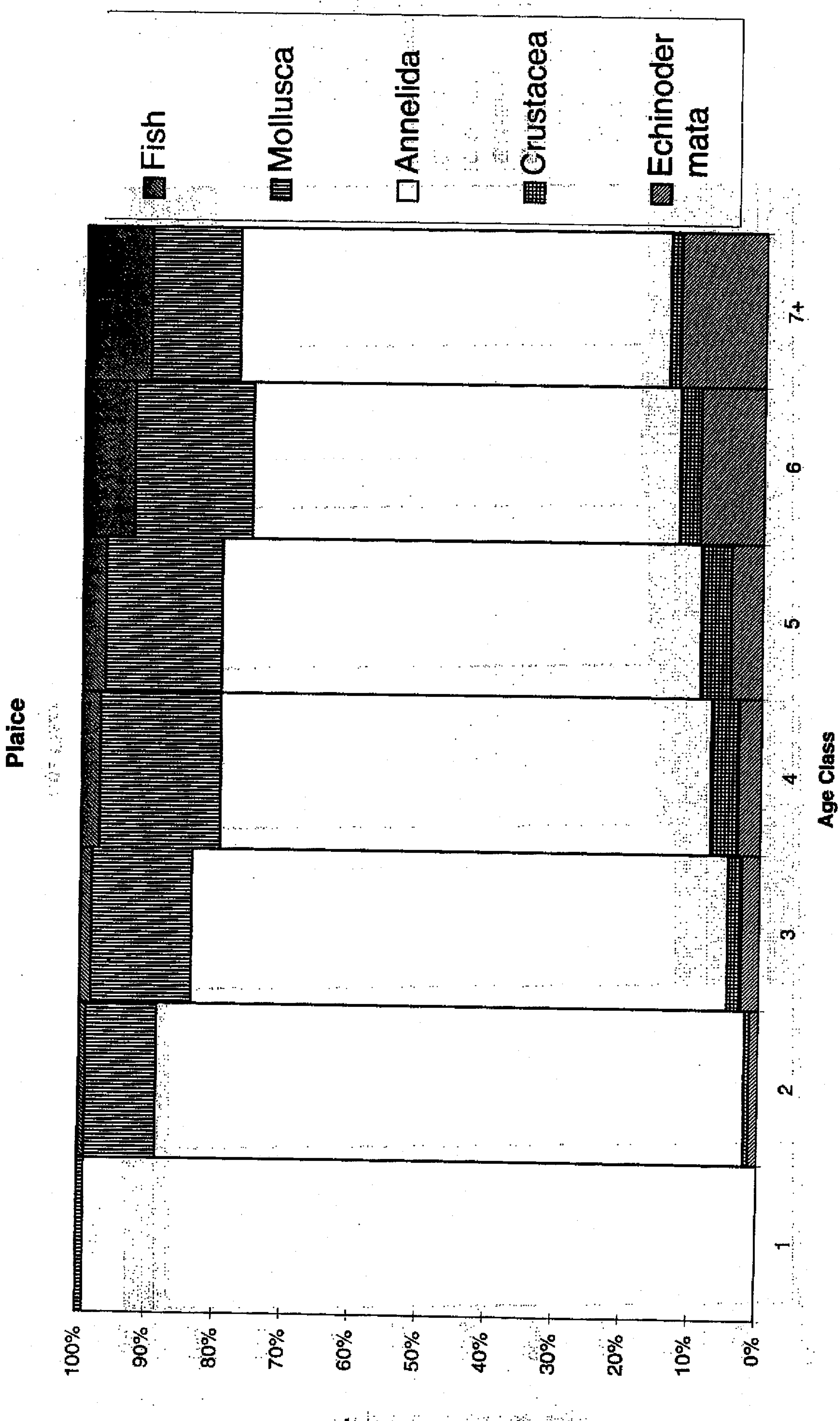


Figure 4.3.1.2.b. Age-specific diet composition of sole in the North Sea.

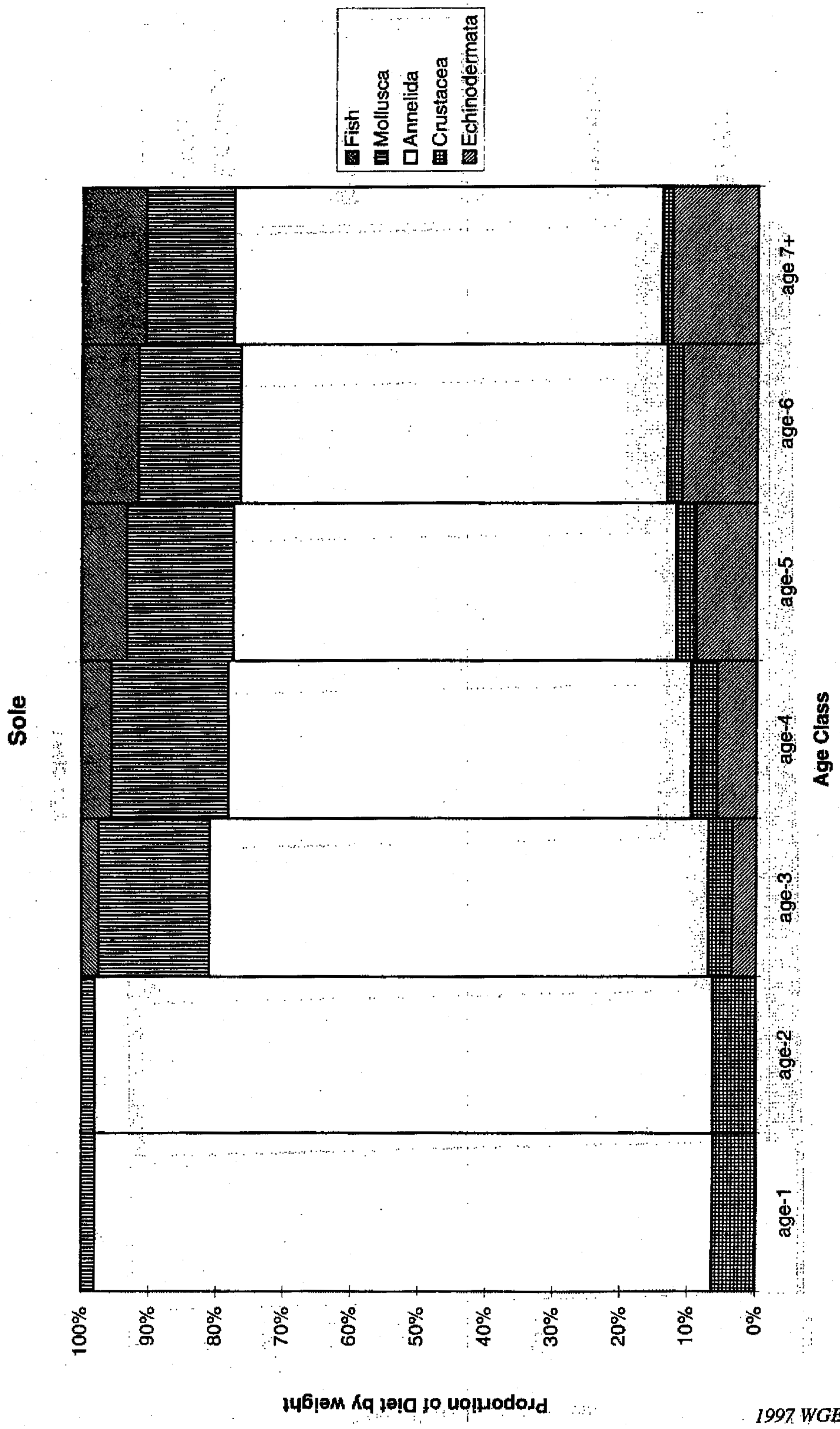


Figure 4.3.1.3.a. Estimated food consumption by flatfish from 1970–1993 by taxonomic group of prey.

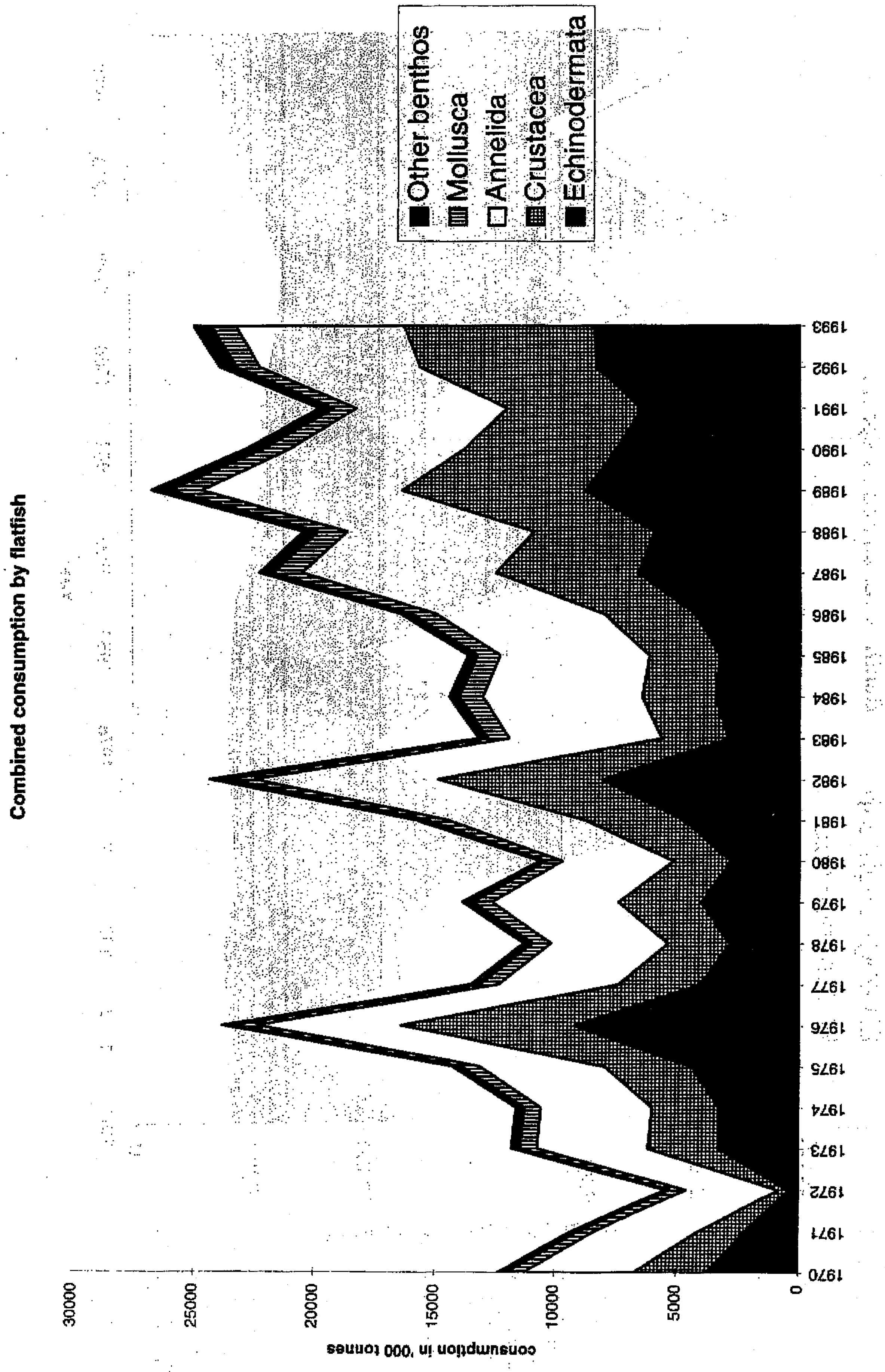


Figure 4.3.1.3.b. Estimated food consumption by flatfish from 1970–1993 by fish species.

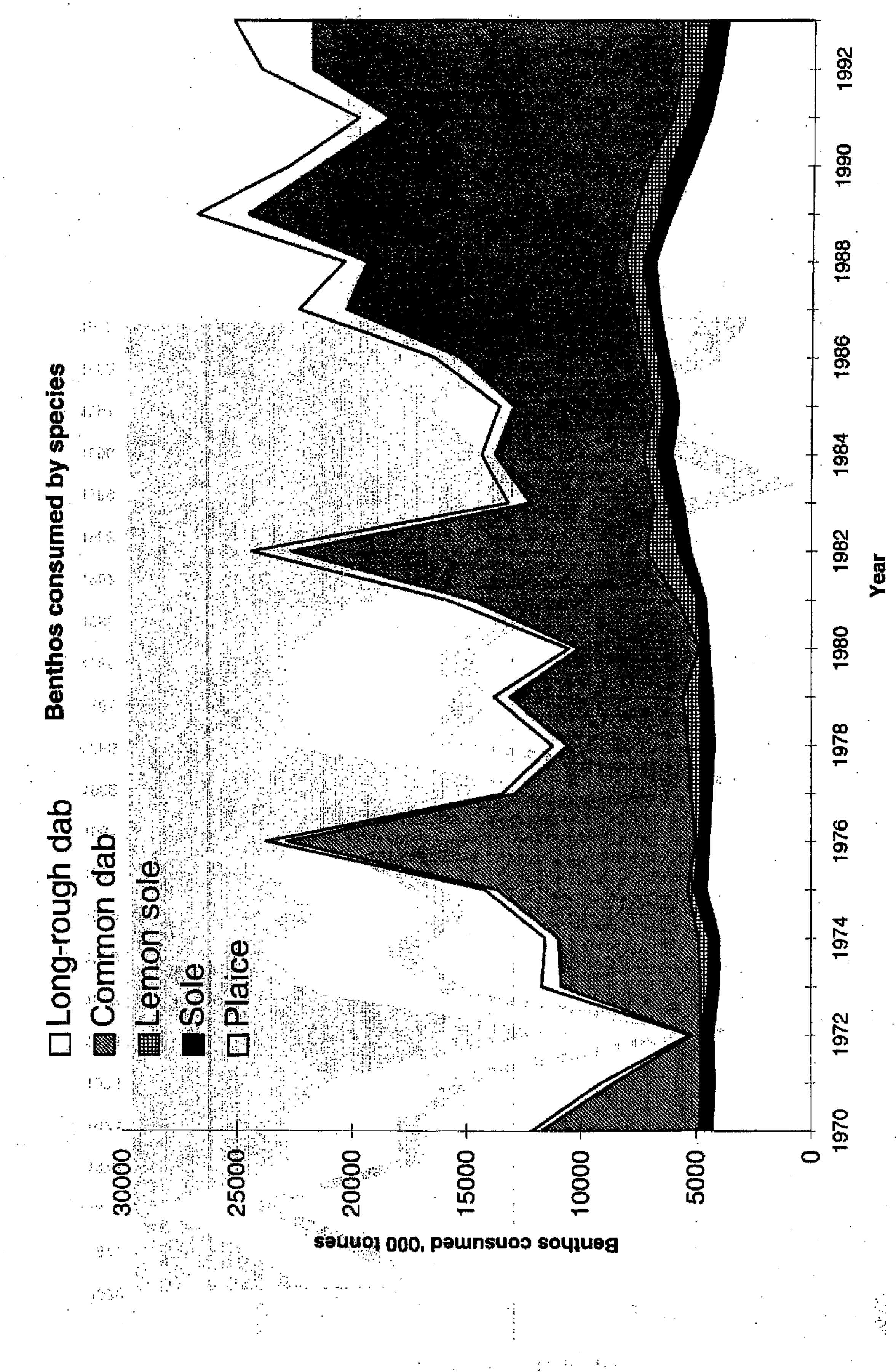


Figure 4.3.2.1.a. Trends in stock biomass derived from VPA for cod.

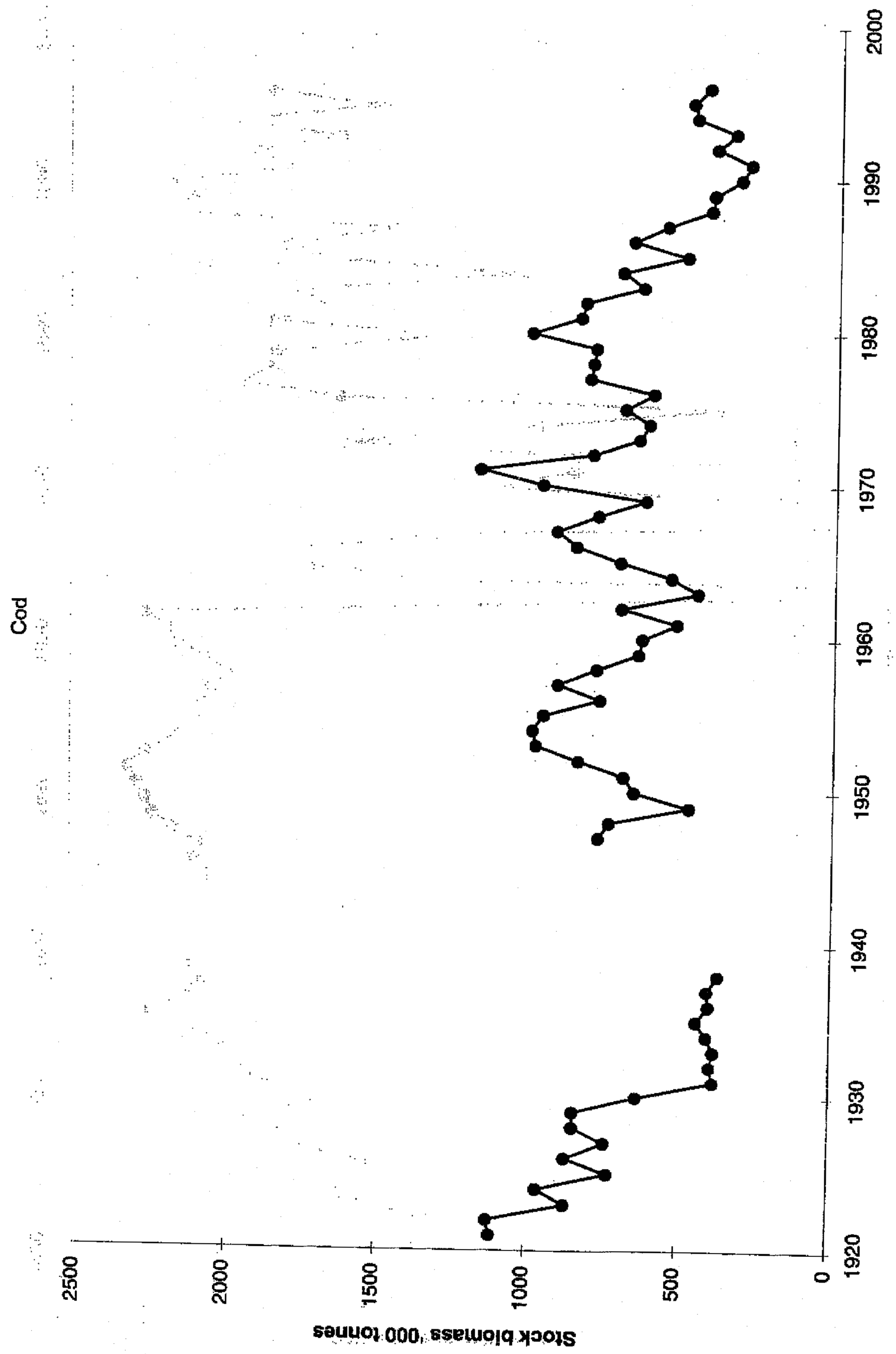


Figure 4.3.2.1.b. Trends in stock biomass derived from VPA for haddock.

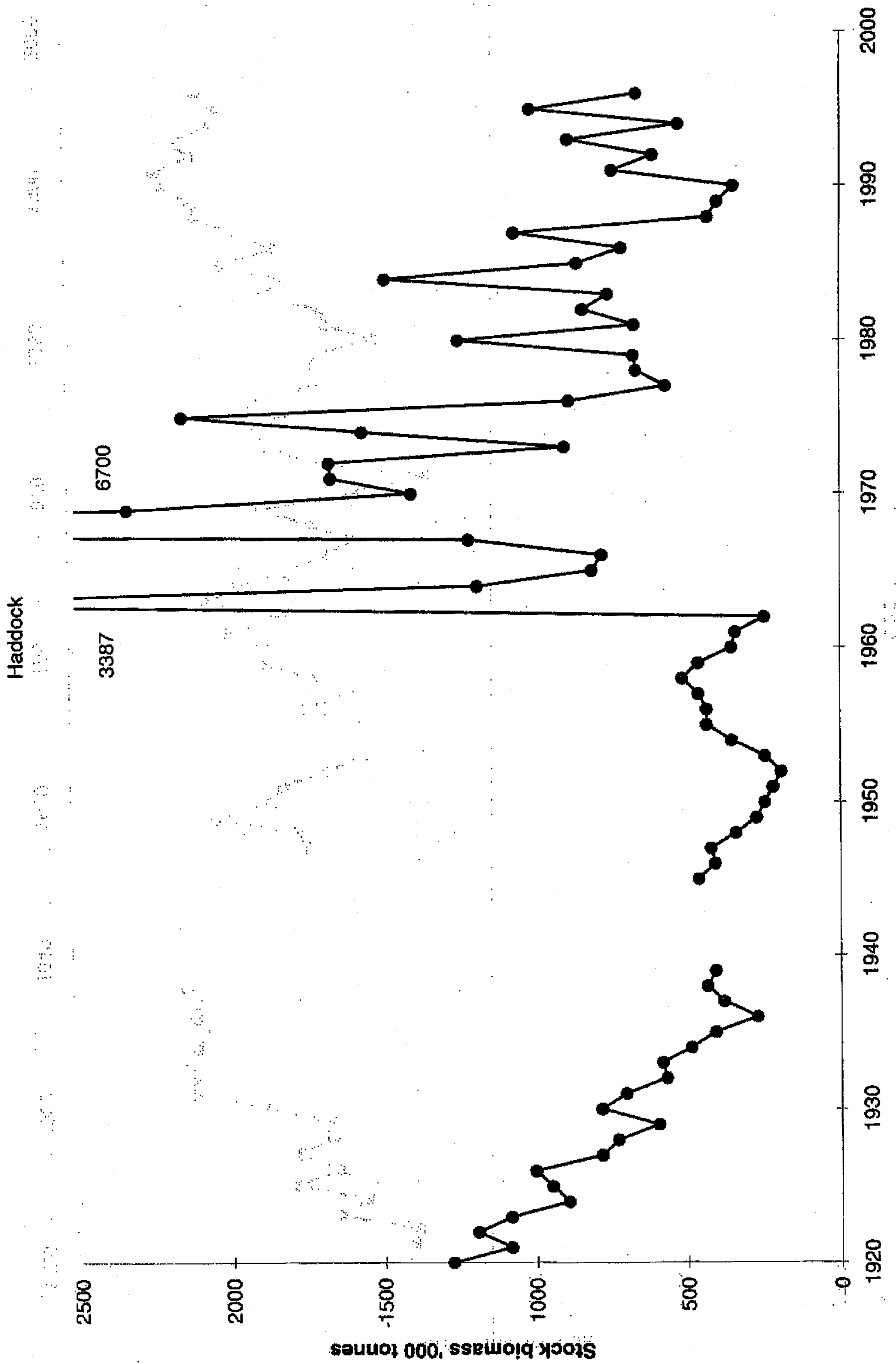


Figure 4.3.2.1.c. Trends in stock biomass derived from VPA for whiting.

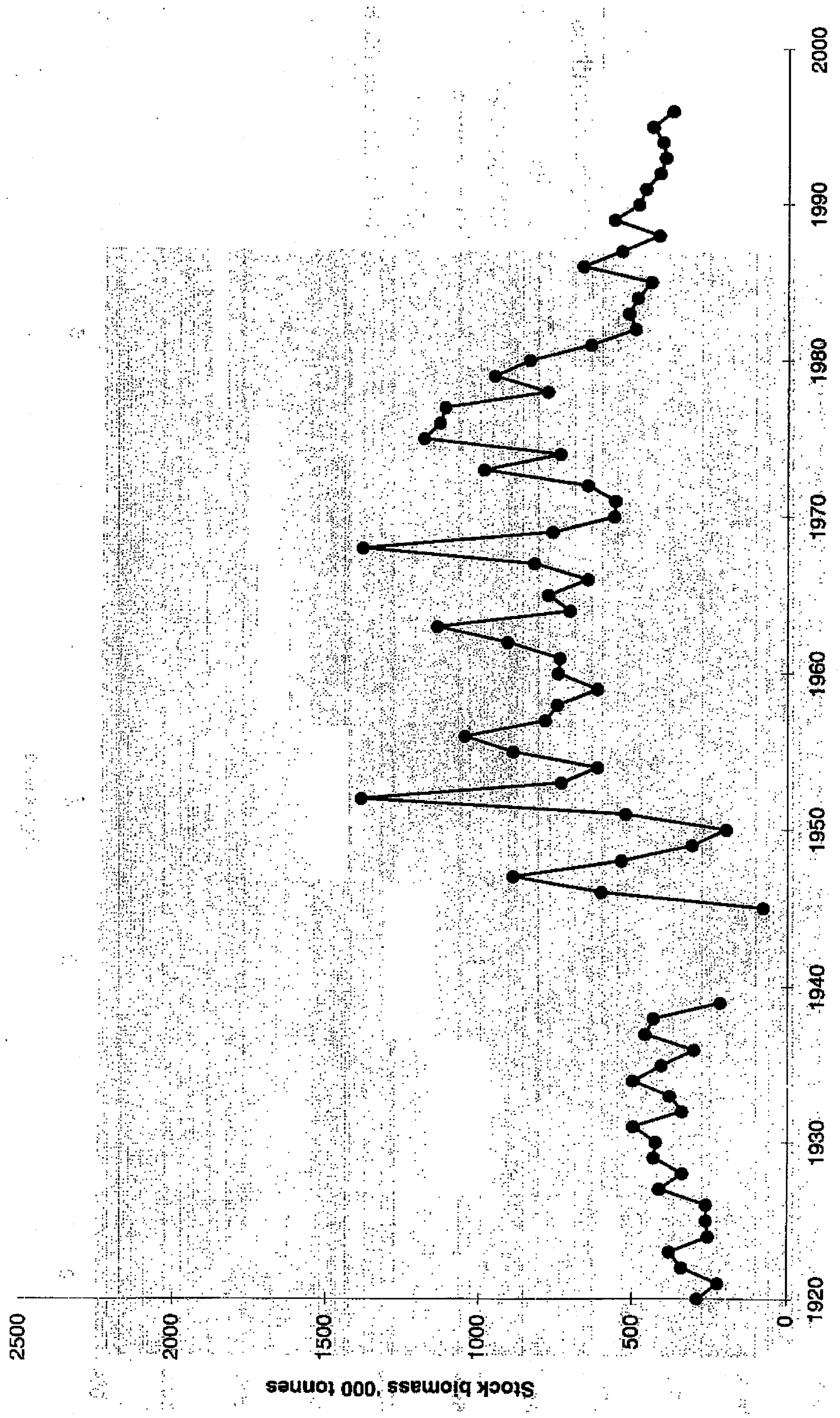


Figure 4.3.2.2.a. Age-specific diet composition of cod in the North Sea.

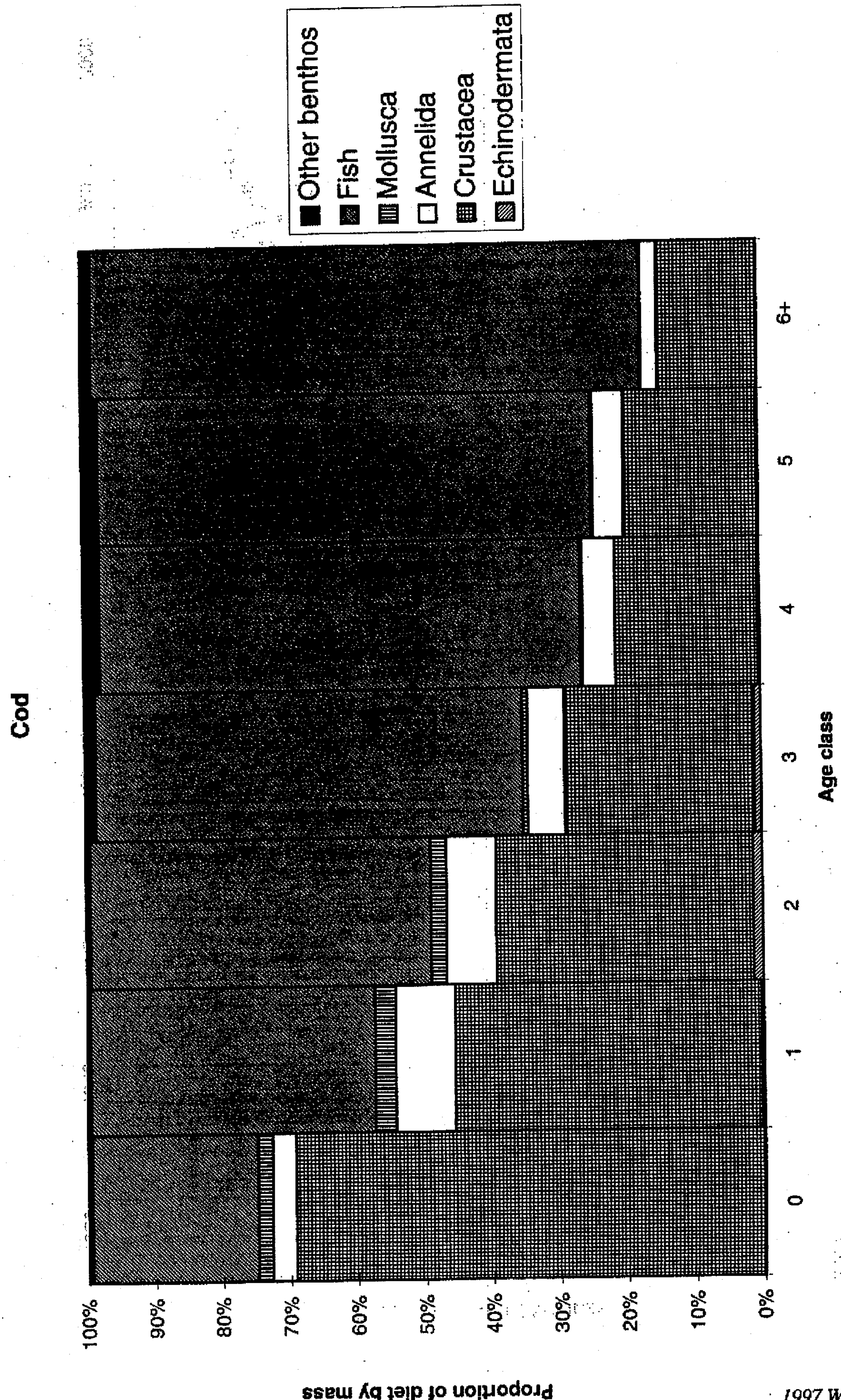


Figure 4.3.2.2.b. Age-specific diet composition of haddock in the North Sea.

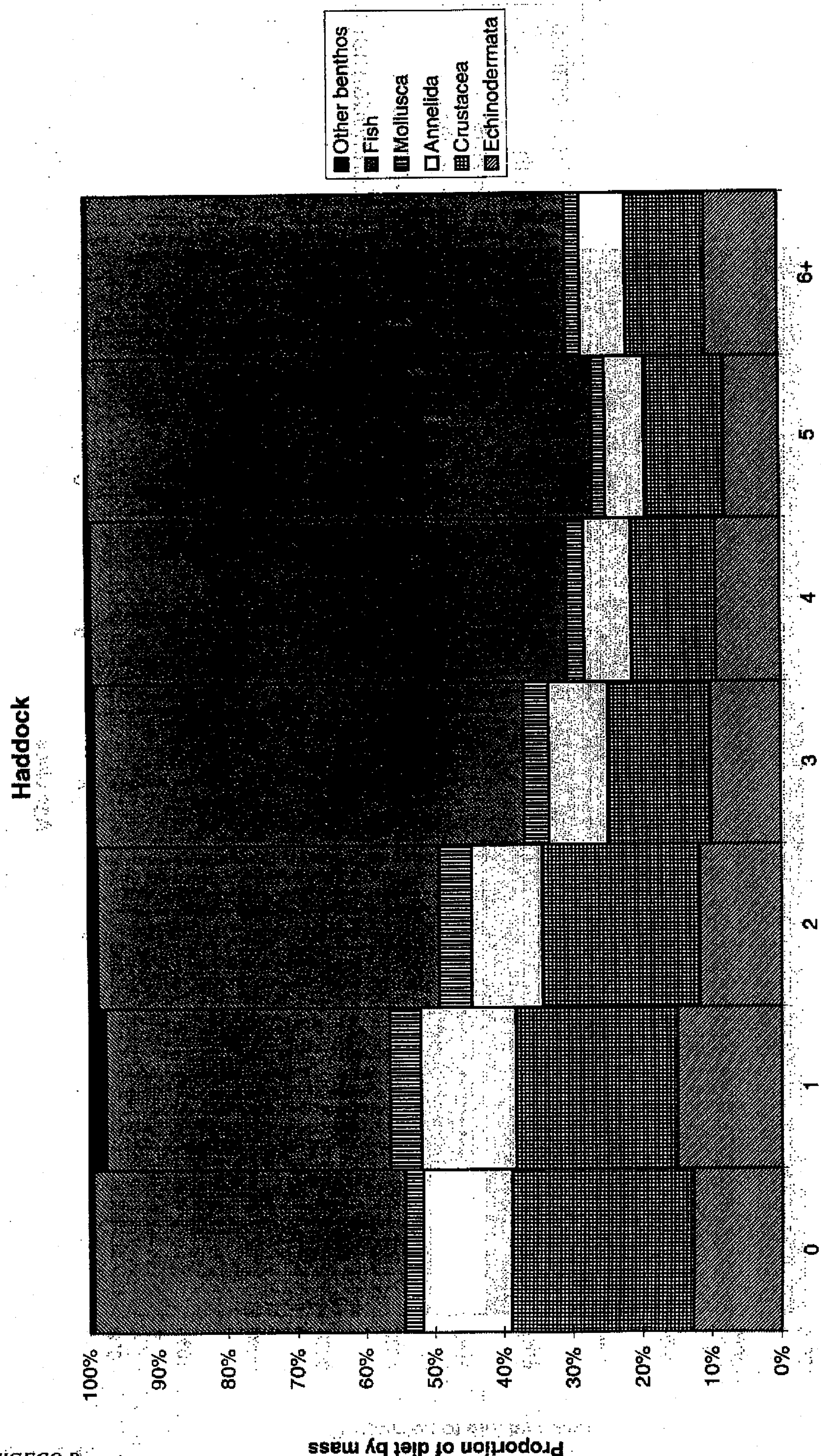


Figure 4.3.2.2.c. Age-specific diet composition of whiting in the North Sea.

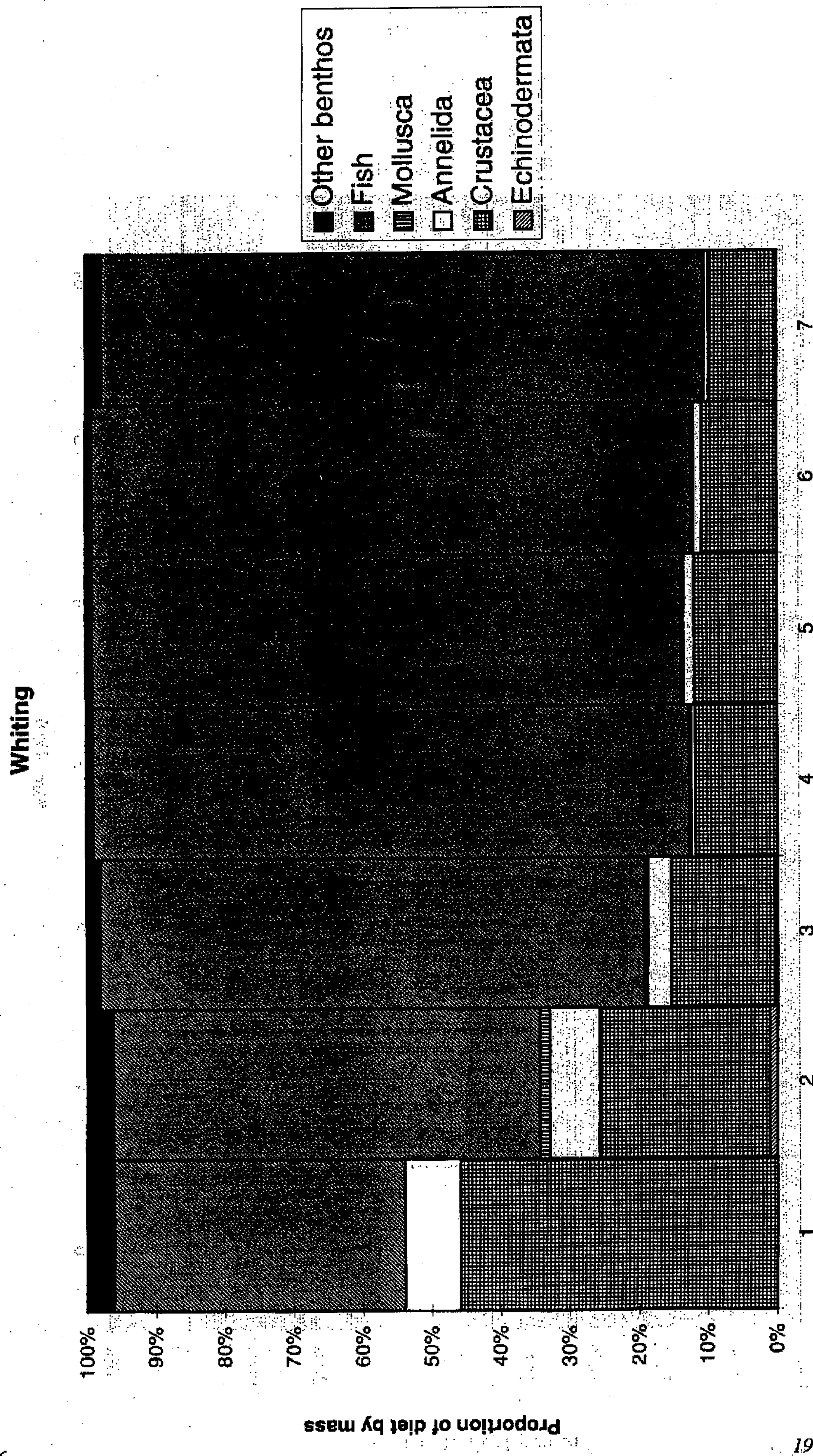


Figure 4.3.2.3.a. Estimated consumption of food by gadoids from 1920–1995 by prey type.

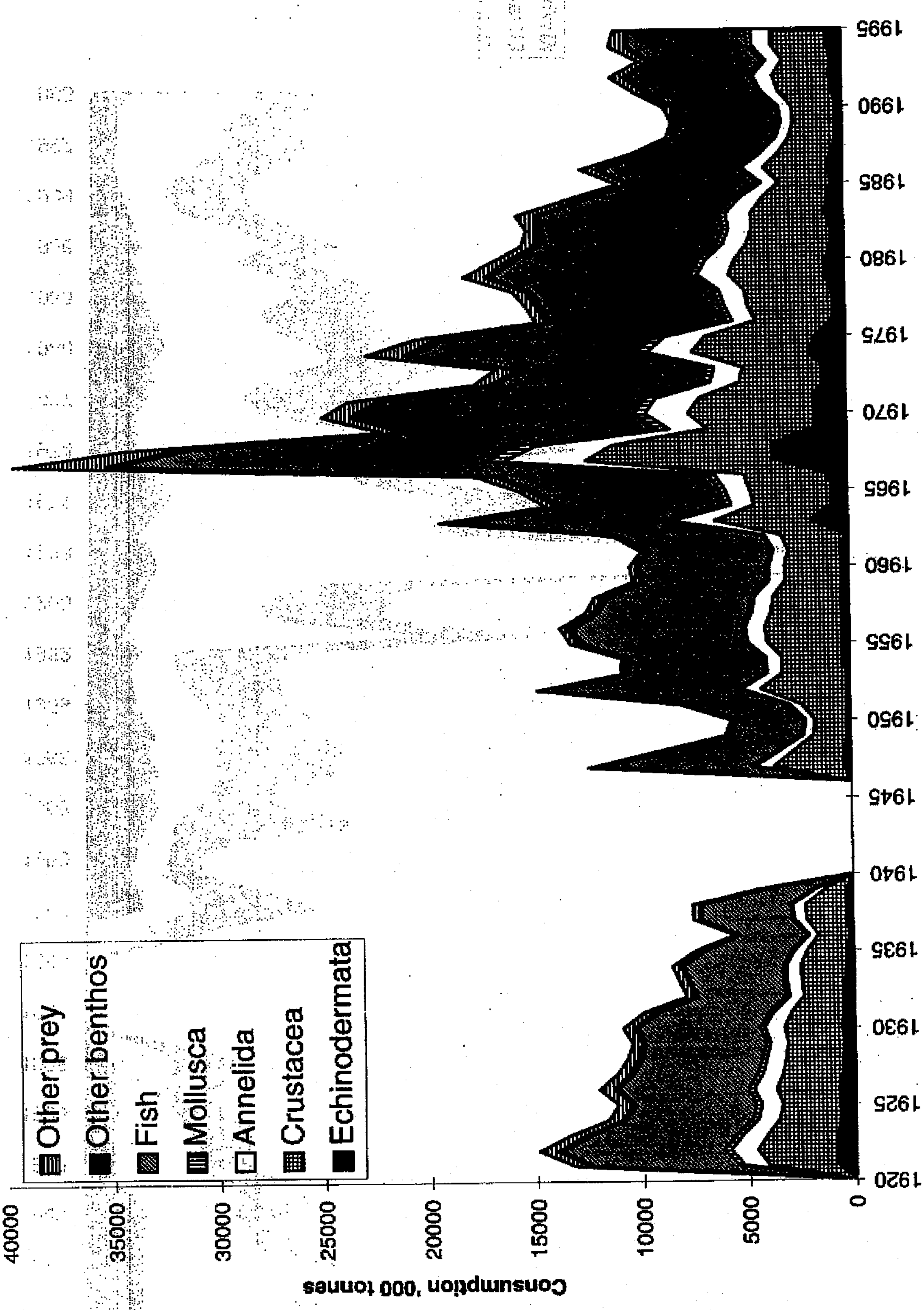


Figure 4.3.2.3.b. Estimated consumption of benthic food by gadoids from 1920–1995 by fish species.

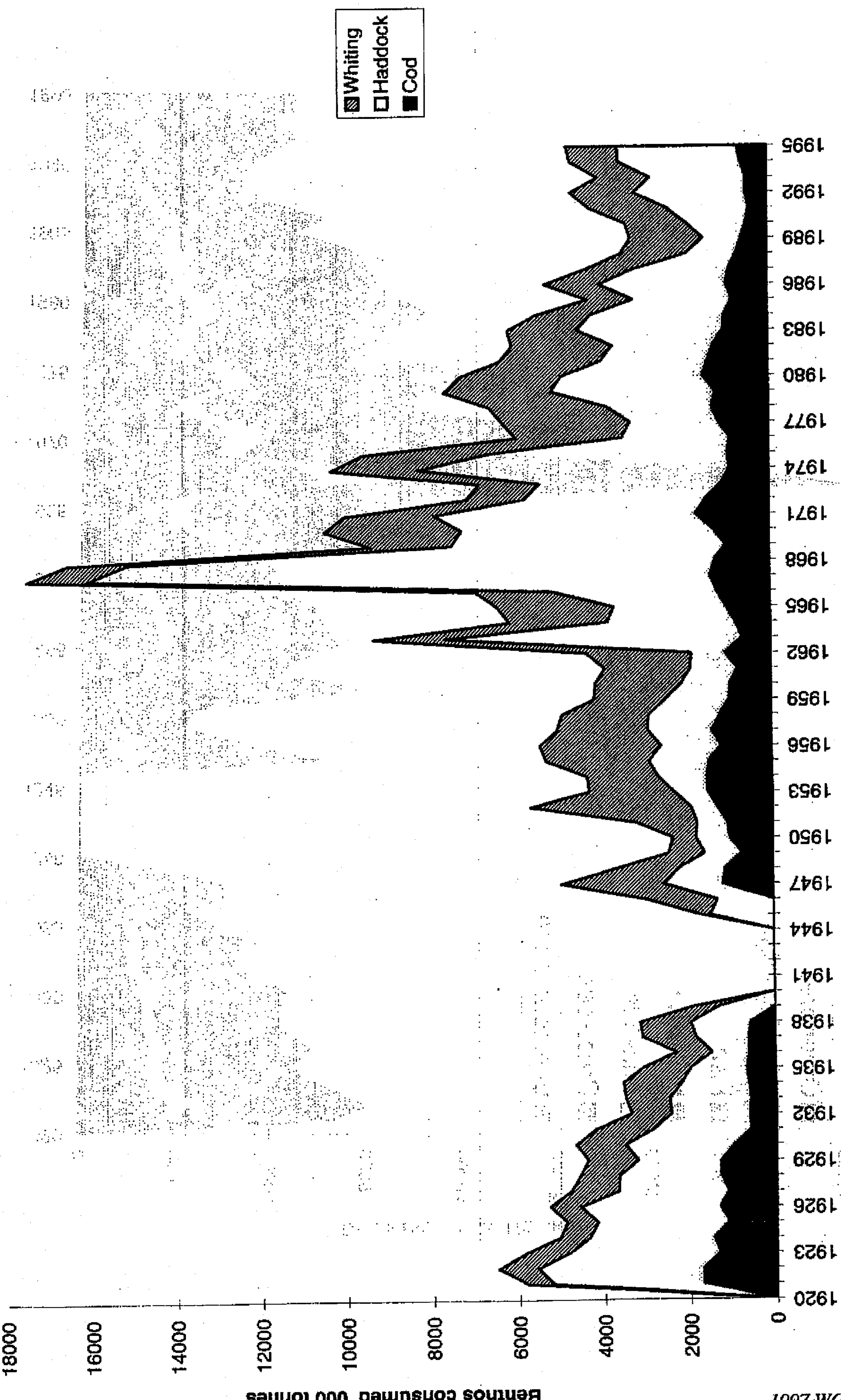
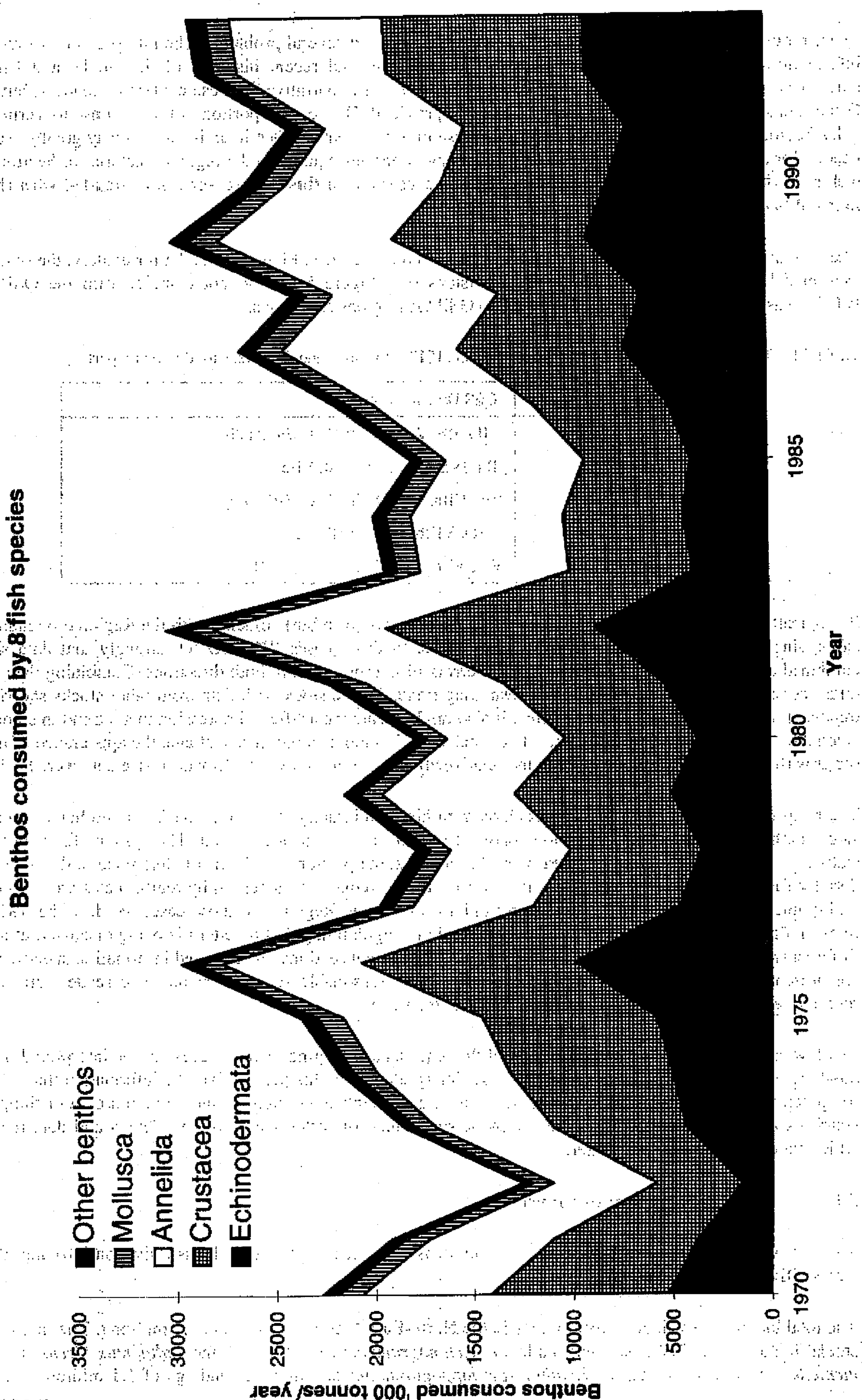


Figure 4.3.3. Estimated total consumption of benthic food by eight species of demersal feeding fish in the North Sea from 1970–1993.



5 INFORMATION ON IMPACT OF FISHING ON AGE/SIZE DISTRIBUTION AND SPATIAL DISTRIBUTION OF FIVE SPECIES FOR FIVE OSPAR AREAS: OPENING REMARKS (TORS C, D, E)

These requests from OSPAR appear straightforward, but posed several problems. The interpretation of most assembled information required some knowledge of the present state and recent histories of the stocks and fisheries in the corresponding OSPAR Region. OSPAR did not ask for specific narrative on these contextual matters, but the Working Group felt strongly that such information should be provided. Hence each portion of the response to Term of Reference c, by Region, begins with a section on content. Presenting this context once in an integrated way greatly reduces the text needed for all the following observations, as the context applies equally to the regional material in Sections 6 and 7, as well as in Section 5. **It is important that subsequent users keep this context section associated with the respective parts of Sections 6 and 7, as well as with Section 5.**

The boundaries of the OSPAR Regions are clearly differentiated (see Figure 5.1.1). Unfortunately, the boundaries ICES uses in delineating Areas, Sub-areas, and Divisions (see Figure 5.1.2) do not coincide with the OSPAR Regions. WGECO assigned ICES Areas and Divisions to OSPAR Regions as follows:

Table 5.1. The relationship between OSPAR regions and ICES Divisions and Sub-areas used in this report.

OSPAR Region	ICES Division/Sub-area
I	I, IIa, IIb, Va, Vb1, Vb2, XIVa, XIVb
II	IIIa, IVa, IVb, IVc, VIId, VIIe
III	VIa, VIIa, VIIb, VIIf, VIIg, VIIh, VIIj
IV	VIIIa, VIIIb, VIIIc, VIIId, IXa
V	VIb, VIIc, VIIk, VIIIe, IXb, X, XII

Biologically, the boundaries among fish stocks do not always (or often) coincide with the Regional boundaries. In some cases, single OSPAR Regions contain several stocks. In those cases WGECO felt strongly that data should not be combined across stocks due to the potential for errors of interpretation in both directions. Combining data across stocks might conceal important information, by swamping trends in one stock with data from other stocks showing dissimilar trends, or a similar trend at a different time. It also might create the artificial impression that a trend in some trait existed when there was no trend. For example, if two stocks in a Region differed in size at age, the appearance of a trend in size (or growth) could result from changes in the relative abundance of the two stocks whose size data were being combined.

The reciprocal problem also exists. Particularly with highly migratory stocks like mackerel and horse mackerel a single stock might be present for all or various parts of year in more than one Region. The systems for collecting fisheries statistics usually try to capture the area being fished, so some portions of Terms of Reference c, d, e can be addressed directly for these stocks. To address other portions of the request on a region-by-region basis would require making highly uncertain (sometimes fictitious) partitionings among the Regions. In those cases we describe and interpret the relevant data series fully in one of the OSPAR Regions, highlighting that a pattern is being discussed at a spatial scale different from the Regional boundaries. In each of the Regional sections, the data and information specific to the Region are presented and discussed as far as can be done with reasonable scientific rigour, and readers are pointed to the regional section which contains the full treatment of the issue.

In a few cases, WGECO found the wording of the requests challenging. Where necessary we interpreted the requests by applying a reasonable amount of common sense. Interpreted from this perspective, the information that the requests ask for generally makes sense as well. It may be surprising to clients how often the interpretation of something superficially simple is actually quite complex. It may also be surprising how often there are very few sound data regarding things which should be known much better.

5.1 OSPAR Region I: Introduction

This region includes the North-East Arctic (Sub-areas I and II), the Faeroe Islands (Division Vb) and North-Western Areas (Division Va and Sub-area XIV).

The total landings of fish and invertebrates in the North-East Arctic in 1996 were 2.5 million t. The major demersal fish stocks include cod *Gadus morhua*, haddock *Melanogrammus aeglefinus*, saithe *Pollachius virens*, redfish *Sebastes mentella*, and Greenland halibut *Reinhardtius hippoglossoides* and in 1995 landings of 1.1 million t were taken from these stocks. The main fleets exploiting the demersal species are factory and freezer trawlers, fresh fish trawlers, a fleet of vessels using conventional gears (gillnet, longline, handline, and Danish seine) and small purse seiners targeting saithe in coastal waters. The two last fleets account for approximately 30 % of the landings of demersal stocks. The major pelagic stocks are Norwegian spring-spawning herring *Clupea harengus* and capelin *Mallotus villosus*, and they are exploited by purse seine and pelagic trawl. In 1996 the landings of herring were 1.2 million t, while there is no fishing on capelin at the moment.

The major demersal stocks at the Faeroe Island are cod, haddock and saithe. The main gears used are bottom trawl, longline, and jigs. The total demersal catches decreased from 120,000 t in 1985 to 65,000 t in 1994 but have since increased again to 83,000 t in 1996.

In North-Western Areas the main demersal stocks are cod at Greenland and Iceland, saithe, and oceanic redfish. The demersal species are mainly exploited by stern trawlers, but considerable fisheries for cod are also carried out by longline, gillnetts, hand lines, and Danish seine. There is also a purse seine fishery on Icelandic summer-spawning herring and capelin in the Iceland/East Greenland/Jan Mayen area.

5.1.1 Cod

5.1.1.1 North-East Arctic Cod

5.1.1.1.1 Context - Pattern of numbers, biomass, landings and fishing mortality (ICES CM 1998/Assess:2)

From a level of about 900,000 t in the mid-1970s, landings (yield) declined steadily to around 300,000 t in 1983–1985 (see Figure 5.1.1.1.1.1). Landings increased to above 500,000 t in 1987 before dropping to 212,000 t in 1990, the lowest level recorded in the post-war period. The catches increased rapidly from 1991 onwards, and were stable around 750,000 t in 1994–1996, while the TAC for 1997 is 850,000 t. The average age 5–10 fishing mortality (F) increased almost continuously from a level about 0.2 in 1946 to 0.9 at the end of the 1970s. In the years 1981–1989 the average Fs were in the range 0.7 to 1.0. In 1990 fishing mortality dropped to 0.28 as a result of management measures brought into effect to control the amount of fishing effort. Age 5–10 F then increased, reaching 0.76 in 1994 but dropped again to 0.58 in 1996. The present level of exploitation is well above both the F_{max} and F_{med} levels of 0.26 and 0.46, respectively, which means that there is a potential for increased yields by lowering the fishing mortality. The MBAL for the spawning stock biomass (SSB) of Northeast Arctic cod is 500,000 t. Since 1991 the SSB has been above this level, and most of the year classes produced in the 1990s have been strong at the 0-group stage. With the present level of exploitation (F97 = 0.67), however, the SSB is expected to drop below 500,000 t in the year 2000. ICES recommended that the fishing mortality should be reduced to below F_{med}, corresponding to landings in 1998 of no more than 514,000 t. The agreed TAC, however, ended on 654,000 t.

5.1.1.1.2 Changes in size distribution and/or age composition

Current landings are dominated by the strong 1990 year class. In recent years landings have usually been concentrated on one or two strong year classes. The strong year-to-year variation in year class strength leads to changes in size and age distributions of such a magnitude that it, to some degree, will mask the effects of changes in total fishing effort or fishing mortality. Both the numbers at age and biomass (by age) distributions for the stock are shown in Figures 5.1.1.1.2.1 and 5.1.1.1.2.2. The distributions are clearly dominated by the large changes in recruitment. The effect of the fishery can somewhat be explained by looking at Figure 5.1.1.1.2.3. This graph shows the average age and biomass distributions for three different periods. The first period is 1946–1947 which represents the closest we get to an unexploited stock through this time series. This period is just after the Second World War when the fishing effort in the Barents Sea was negligible and the only fishery with any impact was the fishery in Lofoten during the spawning season. The period from 1974–1975 represents a time with very high fishing effort, while the last period (1992–1993) represents the stock just after a short period of reduced effort following the collapse in the late 1980s.

5.1.1.1.3 Changes in spatial distribution

The North-East Arctic cod is a highly migratory stock with both feeding and spawning migrations. Recent survey results have shown that the area occupied by the stock increases with increasing stock size (Jakobsen *et al.*, 1997). This increase in area should also be expected to be related to climatic conditions. That is, favourable climatic conditions (higher temperatures) seem to coincide with good recruitment, high growth, and increasing stock size. Due to the strong migrations, a sustainable fishery is not likely to affect the spatial distribution directly. However, if the spawning stock is largely reduced and fewer recruits are produced, the extension of both the spawning areas and nursery areas will be reduced. An attempt to illustrate the changes in distribution together with changes in abundance is presented in Figure 5.1.1.1.3.1. The yearly (February) abundance indices from the Norwegian bottom trawl survey for age groups 3 to 8 are shown together with the corresponding estimated percentages of the stock occupying the eastern part of the Barents Sea (the survey area east of 30°E). The impact of the strong 1983 and 1990 year classes dominates the abundance indices shown. The easterly distribution has an increasing trend from the late 1980s to the mid-1990s, with a tendency for local 'peaks' in 1989 and 1994. The data on percentages distributed in the east are also shown in Figures 5.1.1.1.3.2 and 5.1.1.1.3.3. The first figure shows the year class (cohort) effects while the second figure clearly demonstrates the age effect. That is, older fish are distributed further west than younger year classes. It should be noted that the distributional data presented represent only year-to-year effects between the February observations. Seasonal variations showing spawning and feeding migrations are not presented.

5.1.1.2 Icelandic Cod

Fisheries overview

The fleet fishing for cod at Iceland operates throughout the year (ICES, 1997). The gears used for catching cod are longlines, bottom trawls, gillnets, handlines, and Danish seines. The fishing vessels are of different sizes but can, however, be grouped into three main categories: trawlers (> 300 GRT), multi-gear boats (< 300 GRT), and small boats (< 20 GRT). The trawlers operate throughout the year outside the 12-mile limit. They follow the spawning and feeding migration patterns of cod and fish on spawning grounds off the southwestern and southern coasts during the spawning season, but move to feeding areas off the northwestern coast during the summer time. During the autumn, this fleet is more spread out. The multi-gear boats operate mainly using gillnets during the spawning season in winter and spring along the southwestern coasts, but in recent years this fleet has also used gillnets in late autumn. Part of this fleet uses longlines during autumn and early winter. During summer some of these boats trawl along the coast out to the 3-mile limit. Others fish with Danish seines close to the shore. Most of the smaller boats operate with handlines mainly in shallow waters during the summer and autumn periods.

5.1.1.2.1 Context - Pattern of numbers, biomass, landings and fishing mortality

In the period 1978–1981 landings of cod increased from 320,000 t to 469,000 t due to immigration of the strong 1973 year class from Greenland waters combined with an increase in fishing effort. Catches then declined rapidly to only 280,000 t in 1983. Although cod catches have been regulated by quotas since 1984, catches increased to 392,000 t in 1987 due to the recruitment of the 1983 and 1984 year classes to the fishable stock in those years. Since 1988, all year classes entering the fishable stock have been well below average, or even poor, resulting in a continuous decline in the landings. The 1995 catch of only 170,000 t is the lowest catch level since 1942. Effort on cod in 1994 decreased compared to 1993. This trend continued in 1995 and a marked reduction in effort against cod has taken place in the most recent years due to further reductions in quota and a diversion of the effort towards other stocks and areas. Due to an increase of the fishable stock biomass, the quota for the 1996/1997 fishing year was set at 186,000 t. Landings in 1996 increased accordingly to 182,000 t. This led to a slight increase in effort by the gillnet fleet, effort of the longliners declined compared to 1995, and effort of the trawlers was unchanged between these years.

The Icelandic cod stock reduced in numbers from about 600 million to 300 million individuals during the period 1977 to 1994 (Figure 5.1.1.2.1.1), but has increased to 400 million in 1996. The SSB was below 300,000 t during large parts of this period, but since 1993 the SSB has been growing and has now exceeded 300,000 t.

For the last twenty years fishing mortality has ranged between 0.43 and 0.96 (Figure 5.1.1.2.1.2), but since 1993 there has been a substantial reduction in the fishing mortality and in 1995 it was at $F = 0.52$ (Figure 5.1.1.2.1.3). The fishing mortality of the trawlers increased in 1996, which can be explained by increased catch rate for this fleet especially in 1996. The current estimate of F (0.45) is at the F_{med} level. In spite of poor recruitment in recent years, the spawning stock has shown the first signs of recovery from the historically low levels in most recent years. This is a result of recent catch restrictions (25 % of fishable stock size) combined with an increase in maturity at age.

5.1.1.2.2 Changes in size distribution and/or age composition

Fishing mortality by age (Figure 5.1.1.2.2.1) for the gillnetters and the Danish seiners shows that these fleets exploit mainly the oldest age-groups (8–12), whereas the longliners and especially the handliners exploit the younger ages. The average size of cod increased from 44 cm to 53 cm in 1985–1989, but a gradual decline in mean size occurred during 1989–1994 (53 and 44 cm, respectively). During the last few years, the mean length of Icelandic cod has increased again and was at 54 cm in 1997 (Figure 5.1.1.2.2.2b). This is reflected in the size composition of the cod stock, as the general trend in the proportional changes of cod > 40 cm showed an increase from 55 % to 75 % during 1985–1997 (Figure 5.1.1.2.2.2a).

5.1.1.2.3 References

Anon. 1997. State of marine stocks in Icelandic waters 1996/97. Prospects for the quota year 1997/98. Hafrannsoknastofnunin fjolrit nr. 56, Reykjavik 1997. 167 pp.

ICES. 1997. Report of the North-Western Working Group. ICES CM 1997/Assess: 17.

Jakobsen *et al.* 1997.

5.1.1.3 East Greenland cod

Tagging experiments show that an interrelationship exists among the cod stocks in East Greenlandic, West Greenlandic, and Icelandic waters. Due to migration effects, the offshore components of East and West Greenland were first assessed in 1996 as one stock unit and distinguished from the inshore populations.

Cod was mainly exploited by stern trawlers using otter-trawls, but considerable fisheries were also carried out using gillnets, longlines, and handlines (miscellaneous gears).

The officially reported data also include the inshore catches (ICES, 1998). The highest catches recorded since 1955 were reported in the 1960s. From 1968, the catches decreased sharply from about 450,000 t to about 50,000 t in the mid-1970s. Due to two recruiting medium-sized year classes of 1973 and 1984, the catches increased to 100,000 t and 130,000 t, respectively, but then decreased to 1000 t in 1996.

Before 1975, offshore catches dominated the total figures by more than 90 %. Thereafter, this proportion decreased to 40–50 % and the most recent yields have been dominated by inshore landings during the years after 1993 (Figure 5.1.1.3.1).

The directed cod fishery was given up in 1992 due to the severely depleted status of the offshore stock component. Since then, no adequate data have been available to update the analytical assessment. Therefore, the data series for the spawning stock biomass ended in 1992. Before 1992, Figure 5.1.1.3.2 shows a dramatic decrease in spawning stock biomass from 1.8 million t in 1955 to 20,000 t in 1977. After that, it varied within a range of 20,000 t to 100,000 t.

The dramatic collapse of the offshore component of the stock was associated with emigration, high fishing mortalities, and changes in environmental conditions. The interaction between the East Greenland and Irminger currents during the early 1970s and 1980s has apparently rendered climatic conditions unsuitable for offshore cod (ICES, 1997).

5.1.1.3.1 Changes in size distribution and/or age composition

As an analytical assessment has not been calculated since 1992, no information on stock in number per age is available. Age disaggregated abundance indices (for age groups 1–3, 4–6 and 7+), derived from the German groundfish survey (ICES, 1997) and showing the age composition from 1982 to 1996, are mainly related to the recruitment pattern and not to the fishery (Figure 5.1.1.3.1.1).

Rätz (1997) has analysed the structures and changes of the demersal fish assemblage off Greenland. During the period 1982–1996 he found fundamental shifts in species composition in coherence with dramatic changes in stock abundance, biomass, and size structure of ecologically and economically important species.

Figures 5.1.1.3.1.2 and 5.1.1.3.1.3 show these effects very clearly. In both areas, cod has nearly disappeared compared with the relatively high abundance values before 1990. The mean individual weight of cod off West Greenland has decreased from around 1.5 kg to nearly 0.3 kg, whereas off East Greenland the individual weight varies greatly from year to year around by about 2.5 kg but without a trend.

5.1.1.3.2 Changes in spatial distribution

Figure 5.1.1.3.1.2 also gives some indication of a geographical change in abundance. The losses in individuals were less pronounced off East Greenland (94 %) than off West Greenland (100 %). Changes in spatial distribution of cod within the area off East Greenland can hardly be investigated due to the present very low abundance.

5.1.1.3.3 References

ICES. 1998. Report of the Advisory Committee on Fishery Management. ICES Cooperative Research Report. In press.

ICES. 1997. Report of the North-Western Working Group. ICES CM 1997/Assess:13.

Hvingel, C., Siegstad, H., and Folmer, O. 1996a. The Greenland fishery for northern shrimp (*Pandalus borealis*) in Davis Strait in 1995 and January–October 1996. NAFO SCR Doc. 96/102, Ser. No. N2806, pp. 1-29.

Hvingel, C., Siegstad, H., and Folmer, O. 1996b. The Greenland fishery for northern shrimp (*Pandalus borealis*) in Denmark Strait in 1995 and January–October 1996. NAFO SCR Doc. 96/117, Ser. No. N2814, pp. 1–24.

Rätz, H.-J. 1997. Structures and Changes of the Demersal Fish Assemblage off Greenland and Trends in Near Bottom Temperature, 1982–1996. NAFO SCR Doc. 97/5, Serial No. N2830.

5.1.1.4 Faeroe Plateau cod

5.1.1.4.1 Context - Pattern of numbers, biomass, landings, and fishing mortality (ICES CM 1997/Assess:13)

Landings steadily decreased from 35,000 t in 1986 to 6000 t in 1993, the lowest catch on record (Figure 5.1.1.4.1.1). In 1995 the catches increased to 19,000 t and in 1996 to 40,000 t, the highest value during the 1961 to 1996 period. The average age 3–7 fishing mortality for the whole period is 0.48, with the lowest value in 1994 (0.20) and the highest value in 1996 (0.79). The present level of fishing mortality is more than twice the estimated F_{max} (0.31) and F_{med} (0.37). Due to poor recruitment from 1984 to 1991 and high fishing mortalities, the spawning stock biomass declined steadily from 1983 to 1992, when it was lowest on record at 20,000 t. Since then, it has increased sharply to almost 87,000 t in 1996. The SSB is expected to decrease in the medium term. No MBAL has been estimated for the SSB, but only one strong year class has been produced at SSBs lower than 70,000 t. ICES recommends that fishing mortality be reduced to sustainable levels (below F_{med} = 0.37).

5.1.1.4.2 Changes in size distribution and/or age composition

Historic landings are dominated by the catches of 3–5 year olds. The observed changes in size and age distribution are dominated by both changes in recruitment and in fishing effort. High effort shows a clear tendency to reduce the mean age in the stock (3 years and older) while a reduction in effort, as observed in 1991–1994, clearly produces an increase in mean age.

5.1.1.4.3 Changes in spatial distribution

There were no data available to analyse whether there have been any changes in spatial distribution.

5.1.1.5 Faeroe Bank cod

This is a very small stock and no detailed analytical assessment is made on it. The catches declined from 5000 t in 1973 to 330 t in 1992. Since then there has been a very strong regulation of the fishery. The predicted catch for 1997 is 2000 t and ICES recommends that the fishing effort in 1998 should not exceed the present level.

5.1.2 Herring

5.1.2.1 Norwegian spring-spawning herring

5.1.2.1.1 Context - pattern of numbers, biomass, landings, and fishing mortality (ICES CM 1997/Assess:14)

The catches decreased from 1.3 million t in 1951 to record low levels of around 10,000 t in the first part of the 1970s and then increased to 232,000 t in 1993 (Figure 5.1.2.1.1.1). The international fishery on herring recommenced in 1994 with catches of 479,000 t and the catches increased to 900,000 t in 1995, 1.2 million t in 1996, and probably 1.5 million t in 1997. The average age 5–13 weighted fishing mortality is presently at about 0.16. When the stock collapsed around 1970, the average F was about 1.5 (3.5 in 1968). The spawning stock biomass is estimated to have been about 14 million t in 1950, decreasing to below 10,000 t in the beginning of the 1970s, and then slowly recovering, passing 1 million t in 1987, and 5 million t in 1994. The SSB is presently estimated to be about 9 million t and is expected to increase further to 9.6 million t in 1998. From 1998 to 1999 the SSB will decrease for Fs above 0.06 in 1998 due to weak year classes after 1993. The current management strategy is based on F = 0.15, with a catch ceiling of 1.5 million t and a minimum SSB of 2.5 million t (MBAL). The ICES advice is to keep the SSB above MBAL and to adapt the catch control rule to this.

5.1.2.1.2 Changes in size distribution and/or age composition

Both the landings and the stock itself are dominated by very few strong year classes. The fishing mortality of 0-and 1-group herring was high in the 1950s and until the early 1970s, but has been close to negligible after the recovery of the stock in the late 1980s. As long as the stock is being harvested at the current levels and patterns of exploitation, the size and age distribution will be dominated by the varying recruitment, meaning that the mean age in the landings will tend to increase with close to one year for each new year until a strong new year class enters the fishery.

5.1.2.1.3 Changes in spatial distribution

Before the herring stock collapsed around 1970, the wintering took place in oceanic waters off East Iceland (Devold, 1963), and the Norwegian Sea was the main feeding area. Spawning has traditionally taken place at different grounds along the Norwegian coast. Since the herring stock recovered in 1988, the wintering area has been located in fjords of northern Norway and mainly in the Vestfjorden area (Slotte and Johannessen, 1997). In the early 1990s the herring reoccupied the Norwegian Sea as its main feeding area (Vilhjálmson *et al.*, 1997). Since the collapse of the stock was caused by too high fishing mortality, it is reasonable to believe that the fishing was also responsible for the total change in distribution from oceanic waters (the Norwegian Sea) to Norwegian coastal waters and fjords.

5.1.2.2 Icelandic summer-spawning herring

5.1.2.2.1 Context - pattern of numbers, biomass, landings, and fishing mortality

The herring is mainly fished by Danish seines during the autumn along the southeastern and eastern coasts of Iceland.

Total landings of Icelandic summer-spawning herring increased sharply from 16,000 t to 130,000 t in the period 1951–1963 (Figure 5.1.2.2.1.1). For a few years the annual catch remained at a high level, but was then followed by a total collapse in the herring fishery with only a few tonnes landed in 1972 and 1973. During the last twenty years, the Icelandic summer-spawning herring stock has been managed at levels corresponding fairly closely to fishing at $F_{0.1}$. During that period the annual landings have gradually increased from 250 t to the previous high levels of landings of 126,000 t. The stock size of Icelandic summer-spawning herring has increased from 1500 million individuals in 1977 to more than 3000 million in 1996 (Figure 5.1.2.2.1.2). During the same interval, the spawning stock biomass increased from 130,000 t to 600,000 t.

5.1.2.2.2 Changes in size distribution and/or age composition

In the years immediately after the fishery was reopened in 1975, the 1971 year class was the most abundant. During the period 1979–1982, the 1974 and 1975 year classes predominated in the catches. During the period 1983–1986, the fishery was dominated by the strong 1979 year class. On the other hand, the fishery in 1987 and 1988 was based on a number of year classes ranging from 4–10 years of age. In the period 1989–1991, the 1983 year class predominated in the catch. The 1988 year class was also well represented in the 1991 catches and predominated during the 1992 season.

In 1993 the age distribution was dominated by the strong 1989 year class although the 1988 year class was also well represented. In 1994/1995 the catches were distributed over four year classes, i.e., those of 1988–1991. The catch in numbers of 3-year-old herring has never been higher and yielded some 25 % of the total numbers in the 1994/1995 season.

During the period 1977–1996 few changes have taken place in stock numbers at age (Figure 5.1.2.2.2.1, Anon., 1997). In the 1995 landings, 50 % of the catch was 2–5 year-old herring. The proportional abundance of these age-classes has decreased from 80 % to 60 % during that period.

5.1.2.2.3 Changes in spatial distribution

The distribution of the Icelandic summer-spawning herring is mainly constricted to the southeastern and eastern coasts of Iceland. There is no evidence to suggest that the commercial exploitation of herring in this region has altered the spatial distribution of the species.

5.1.2.2.4 References

Anon. 1997. State of marine stocks in Icelandic waters 1996/1997. Prospects for the quota year 1997/1998. Hafrannsoknastofnunin fjolrit nr. 56, Reykjavik 1997. 167 pp.

ICES. 1997b. Report of the Northern Pelagic and Blue Whiting Fisheries Working Group. ICES CM 1997/Assess: 14.

Jakobsson, J. 1980. Exploitation of the Icelandic spring- and summer-spawning herring in relation to fisheries management, 1947–1997. Rapp. P.-v. Rèun. Cons. int. Explor. Mer., 177: 23–42.

Devold, F. 1963. The life history of Atlanto-Scandian herring. Rapports et Procès-verbaux des Rèunions du Conseil International pour l'Exploration de la Mer, 154: 98–108.

ICES. 1997a. Report of the North-Western Working Group. ICES CM 1997/Assess:13. 356 pp.

ICES. 1997b. Report of the Northern Pelagic and Blue Whiting Fisheries Working Group. ICES CM 1997/Assess:14. 188 pp.

ICES. 1998. Report of the Arctic Fisheries Working Group. ICES CM 1998/Assess:2. 366 pp.

Slotte, A., and Johannessen, A. 1997. Spawning of Norwegian spring-spawning herring (*Clupea harengus* L.) related to geographical location and population structure. ICES CM 1997/CC:17. 36 pp.

Vilhjálmson, H., Misund, O.A., Arrhenius, F., Holst, J.C., Gislason, A., Gudmundsdottir, A., Jacobsen, J.A., Krysov, A., Malmberg, S., and Reid, D. 1997. Report on surveys of the distribution, abundance and migrations of the Norwegian spring-spawning herring, other pelagic fish and the environment of the Norwegian Sea and adjacent waters in late winter, spring and summer of 1997. ICES CM 1997/Y:4. 32 pp.

Figure 5.1.1. Regions of the OSPAR maritime area (OSPAR, 1995).

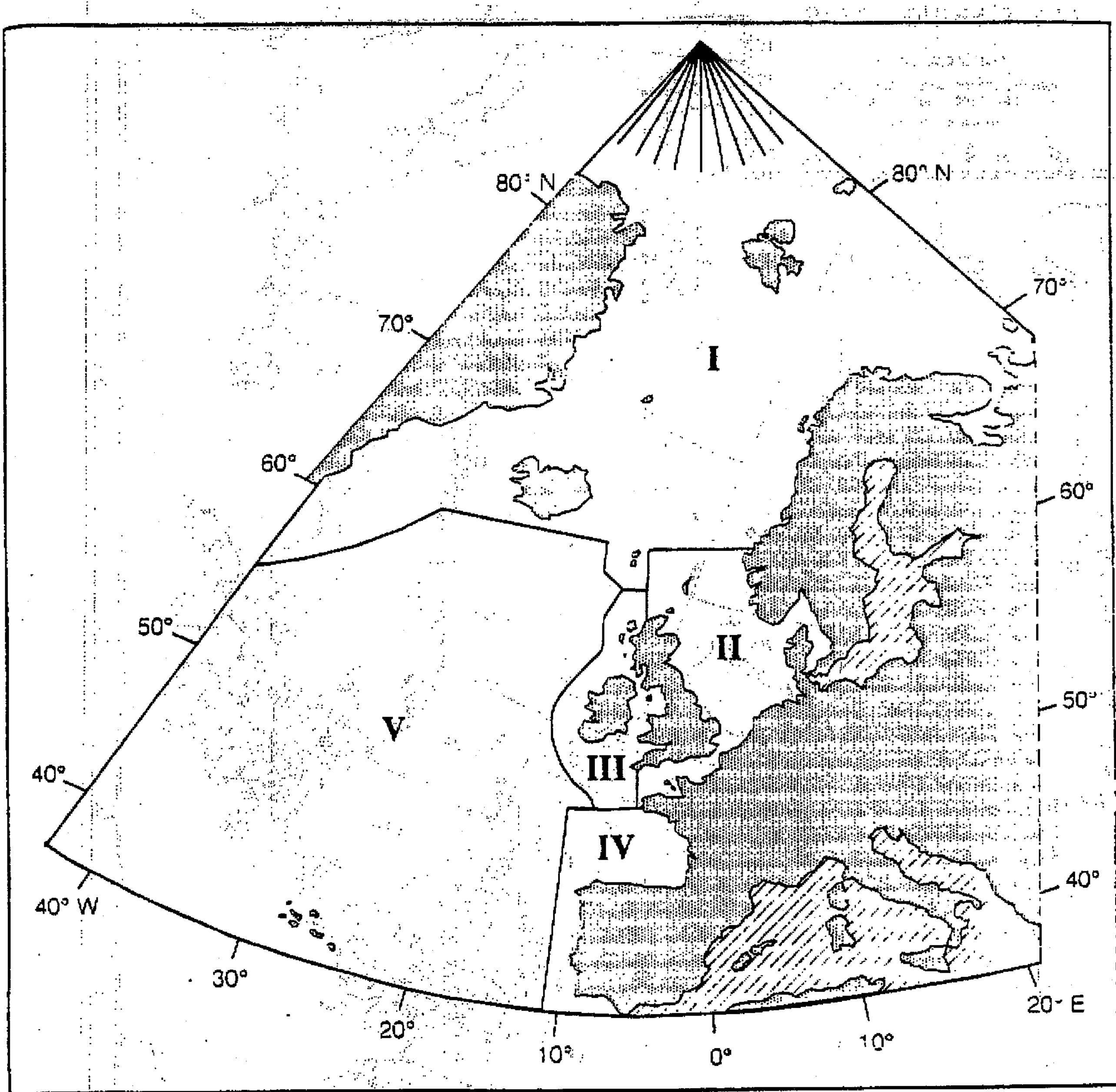


Figure 5.1.2. ICES Sub-areas and Divisions in the Northeast Atlantic.

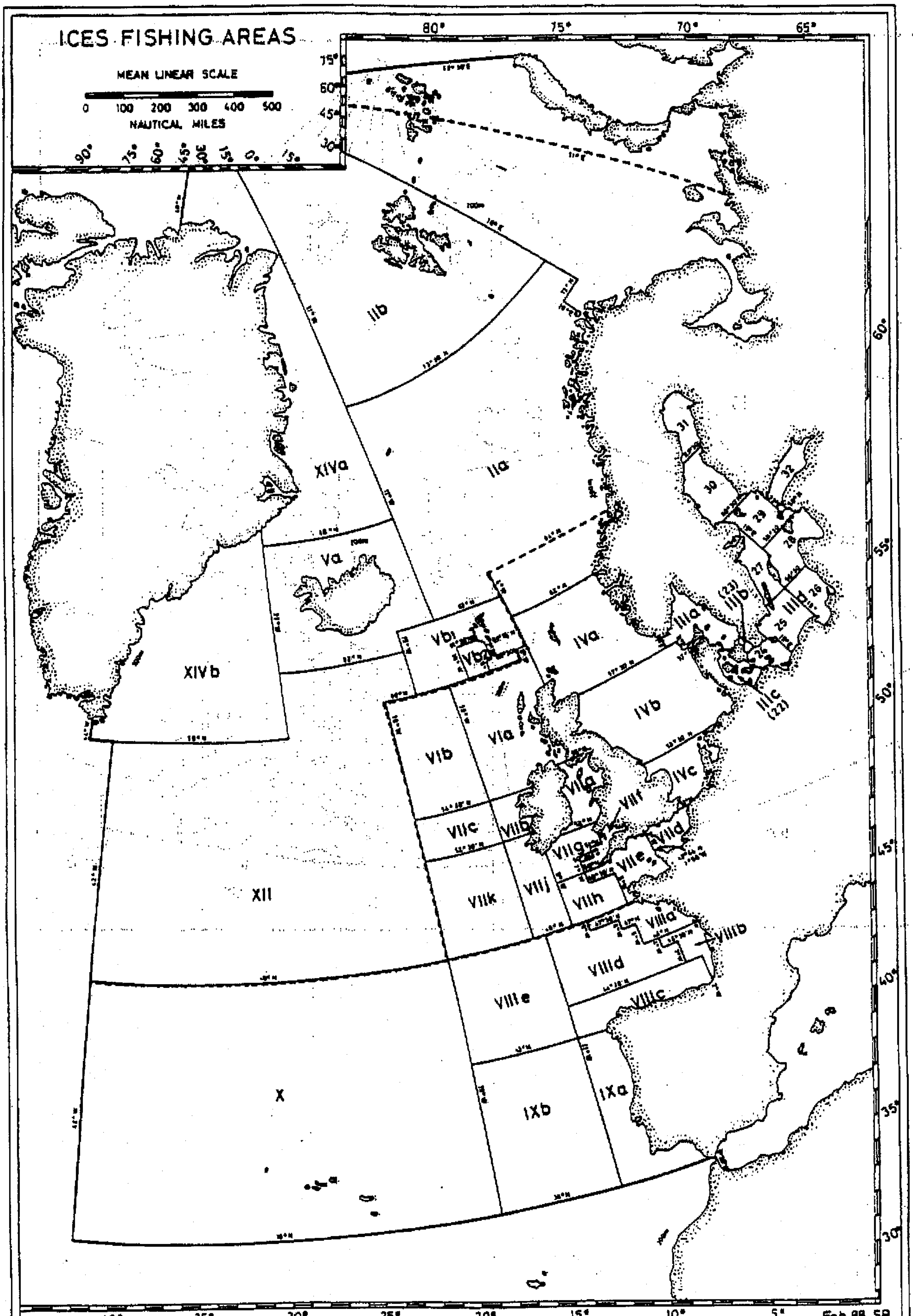


Figure 5.1.1.1.1.1. Landings (yield), fishing mortality, spawning stock biomass, and recruitment for cod in the Northeast Arctic from 1946–1996.

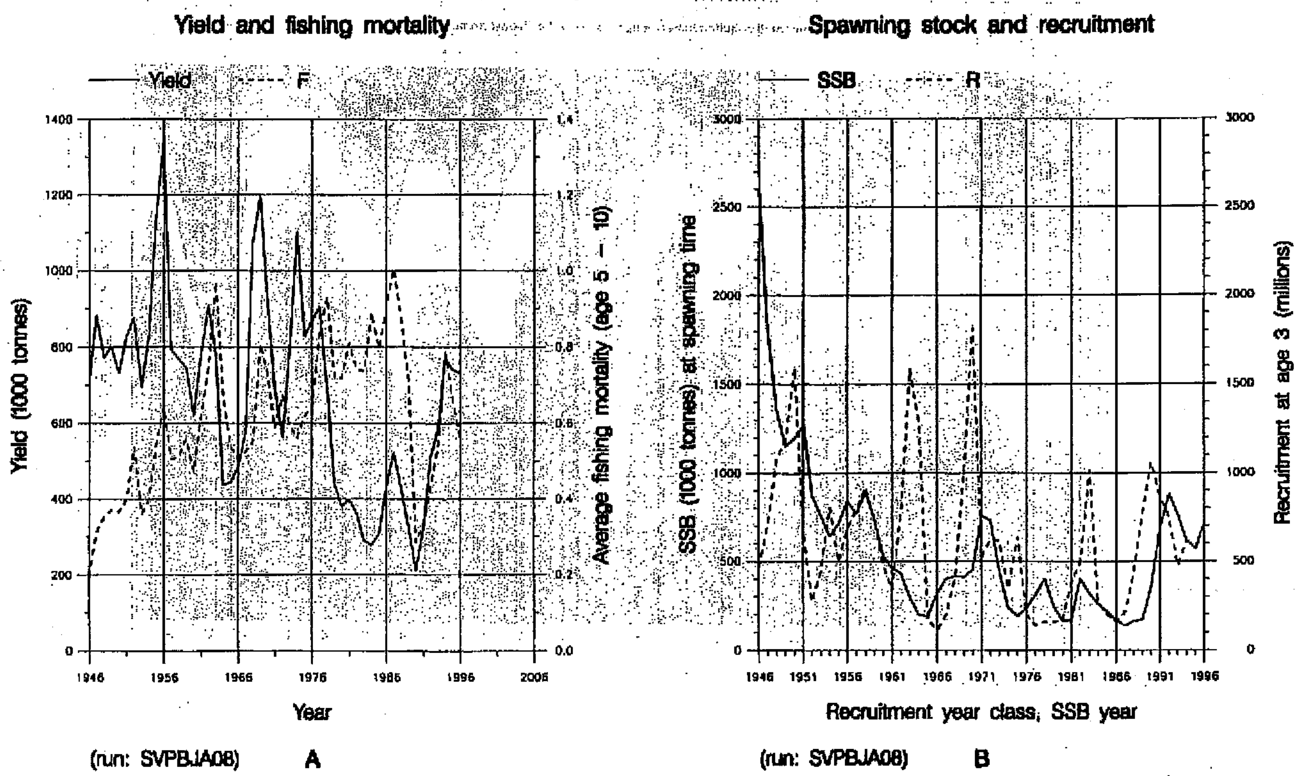


Figure 5.1.1.1.2.1. Observed age distribution Virtual Population Analysis (VPA) for ages 5 and up of Northeast Arctic cod.

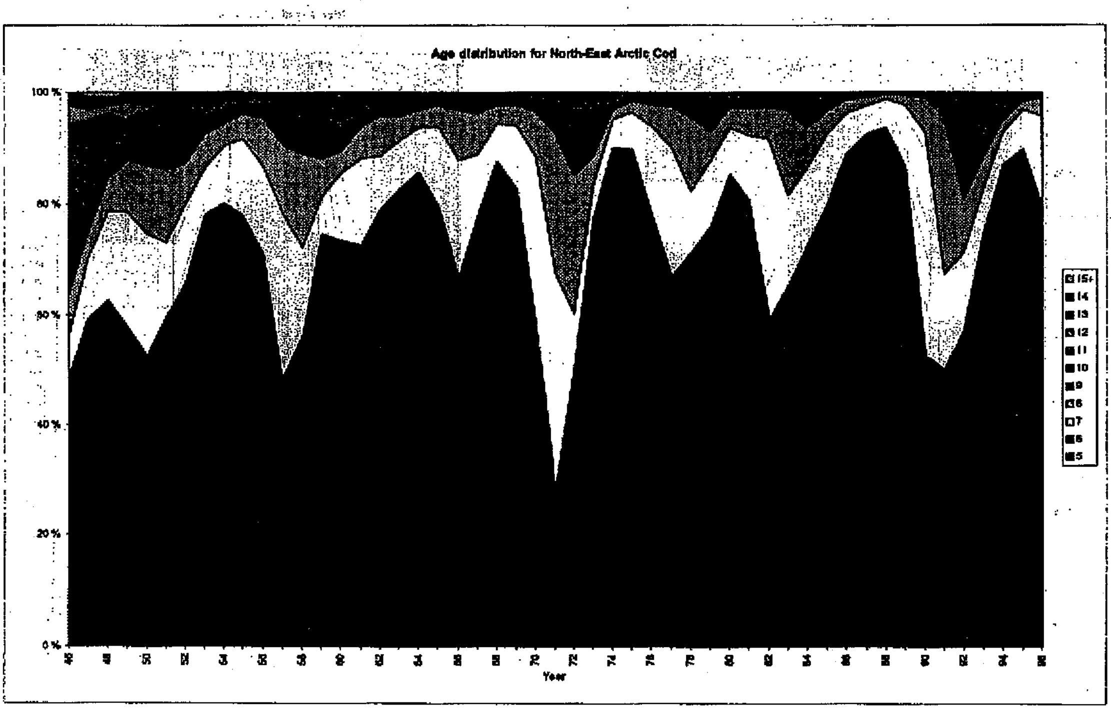


Figure 5.1.1.1.2.2. Observed biomass distribution Virtual Population Analysis (VPA) for ages 5 and up of Northeast Arctic cod.

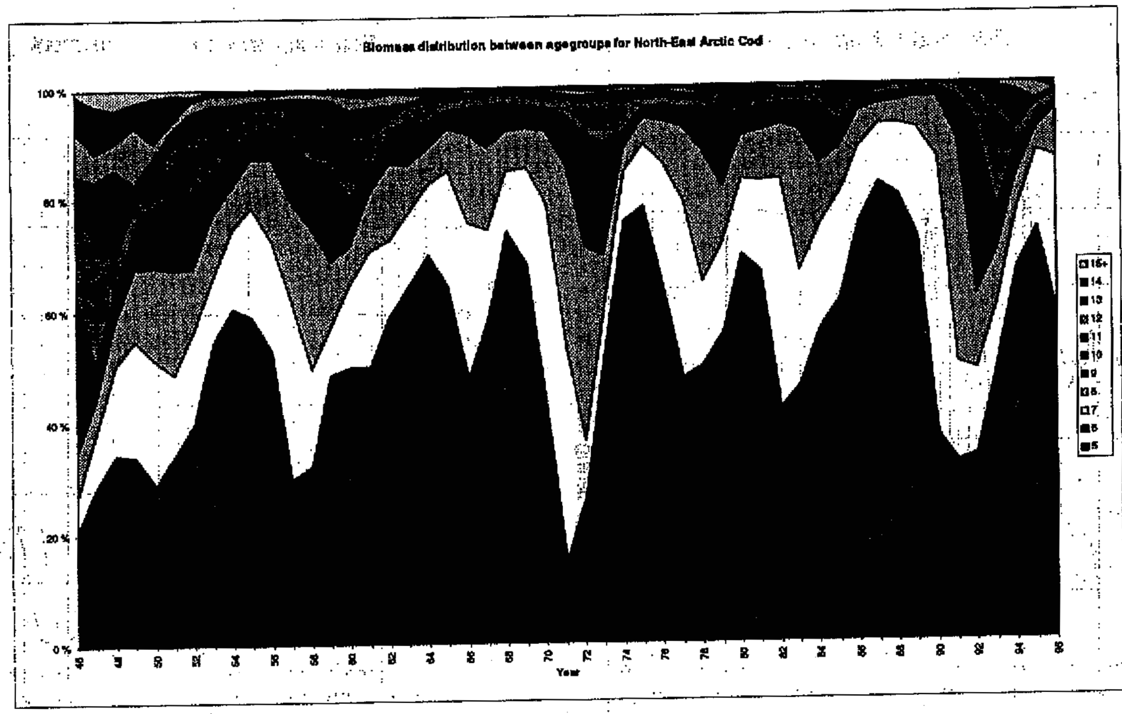


Figure 5.1.1.1.2.3. Average age and biomass distribution for the three periods 1946–1947, 1974–1975, and 1992–1993.

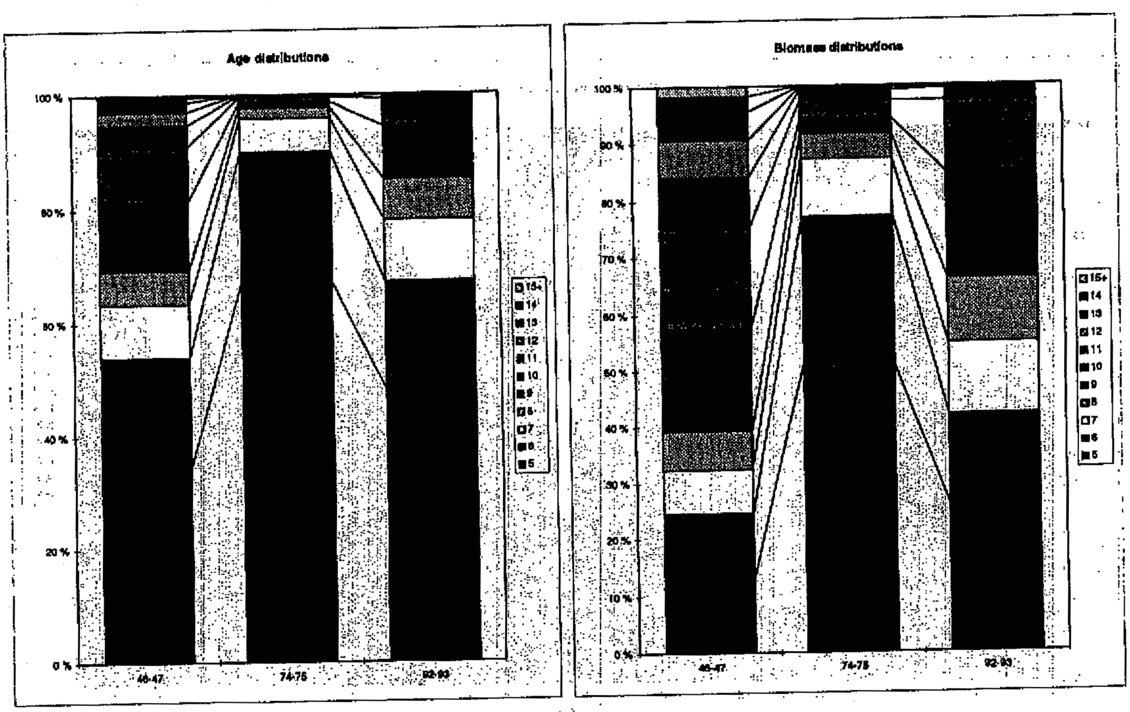


Figure 5.1.1.1.3.1. Abundance indices of Northeast Arctic cod (ages 3–8) together with the estimated percentage of the year class in the eastern part of the survey.

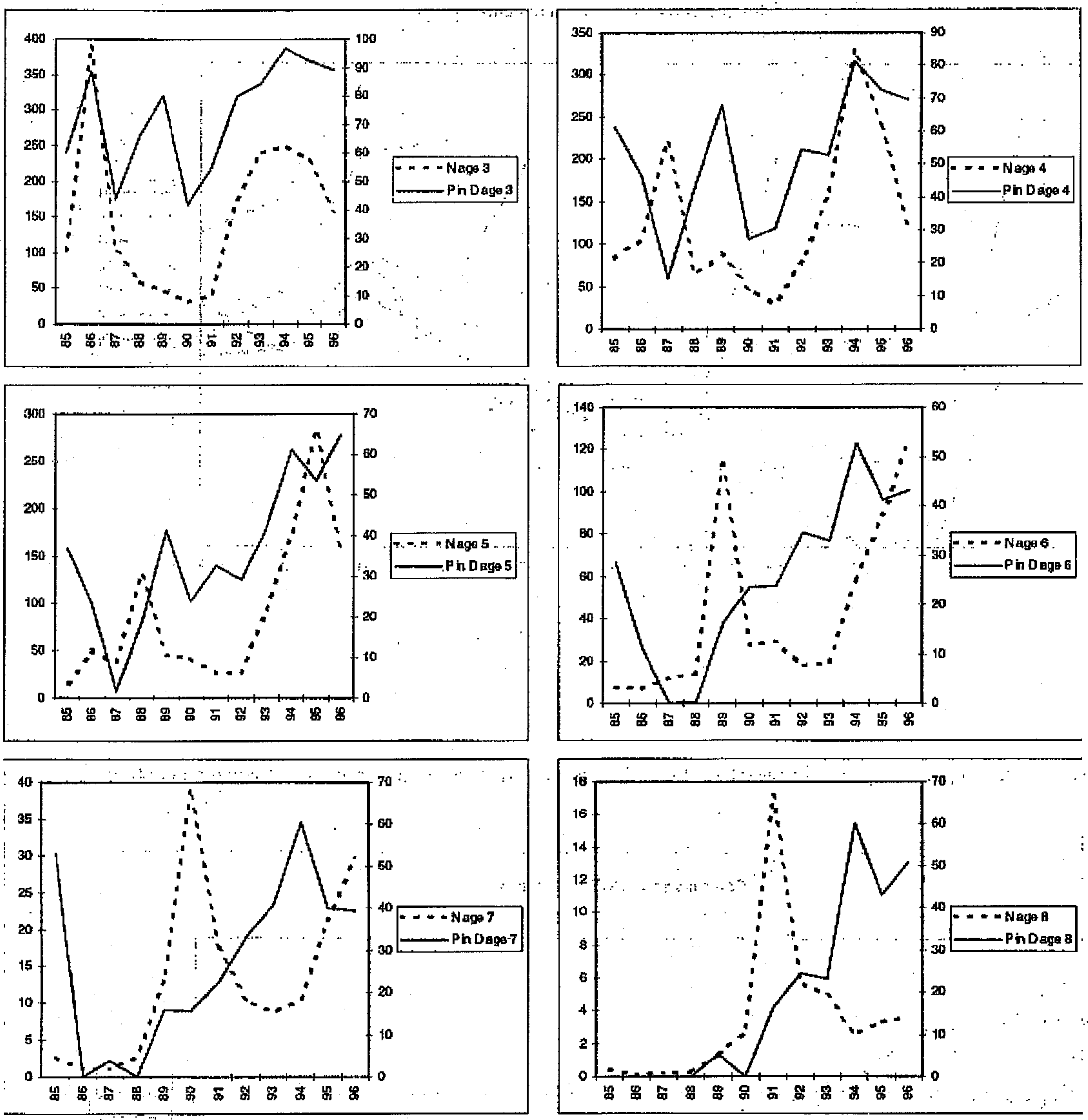


Figure 5.1.1.1.3.2. Estimated percentage of Northeast Arctic cod in the eastern part of the survey area by year and cohort.

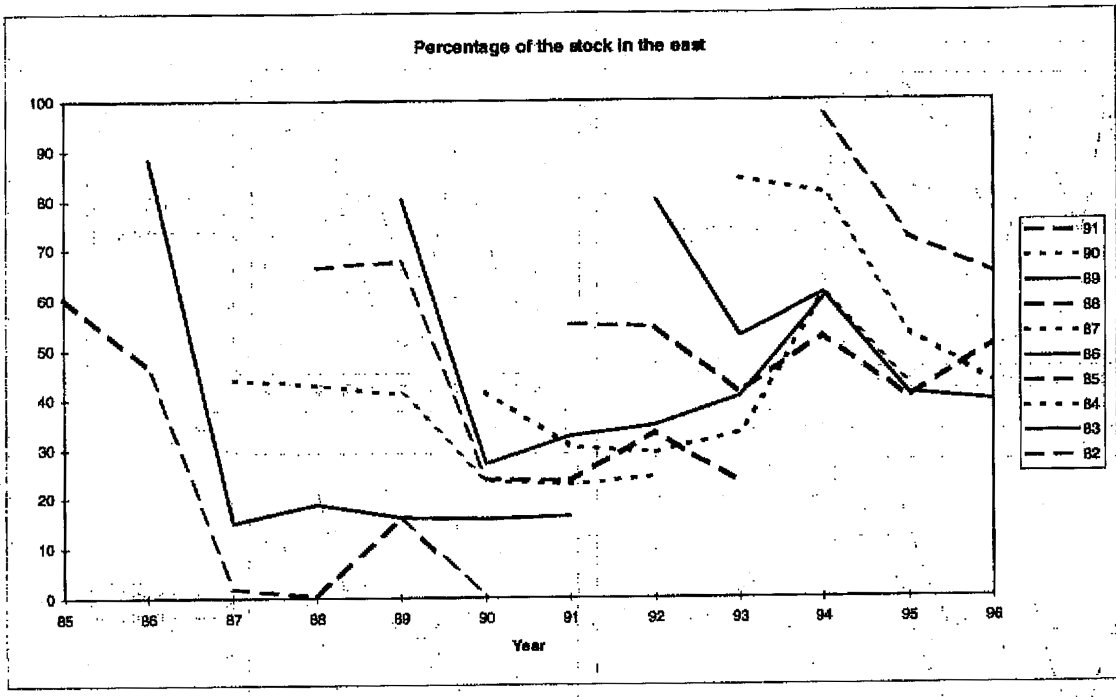


Figure 5.1.1.1.3.3. Estimated percentage of Northeast Arctic cod in the eastern part of the survey area by age and cohort.

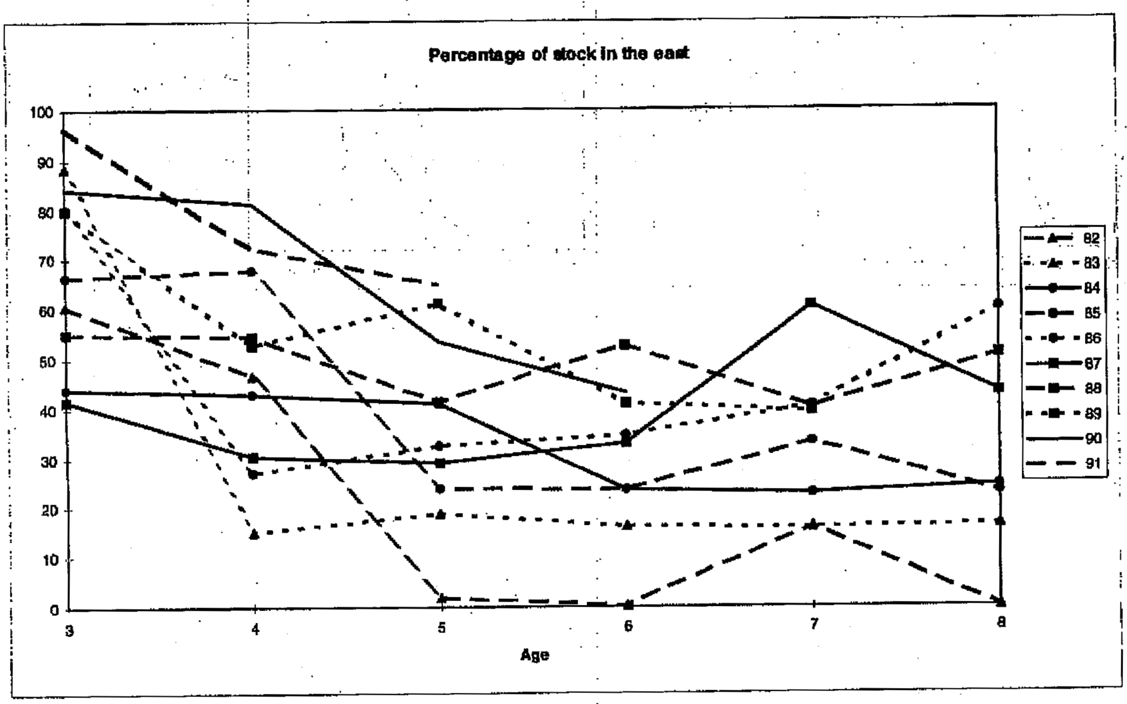


Figure 5.1.1.2.1.1. Cod at Iceland. Total stock in numbers (millions of individuals) and spawning stock biomass (thousands of tonnes). Data from ICES (1997).

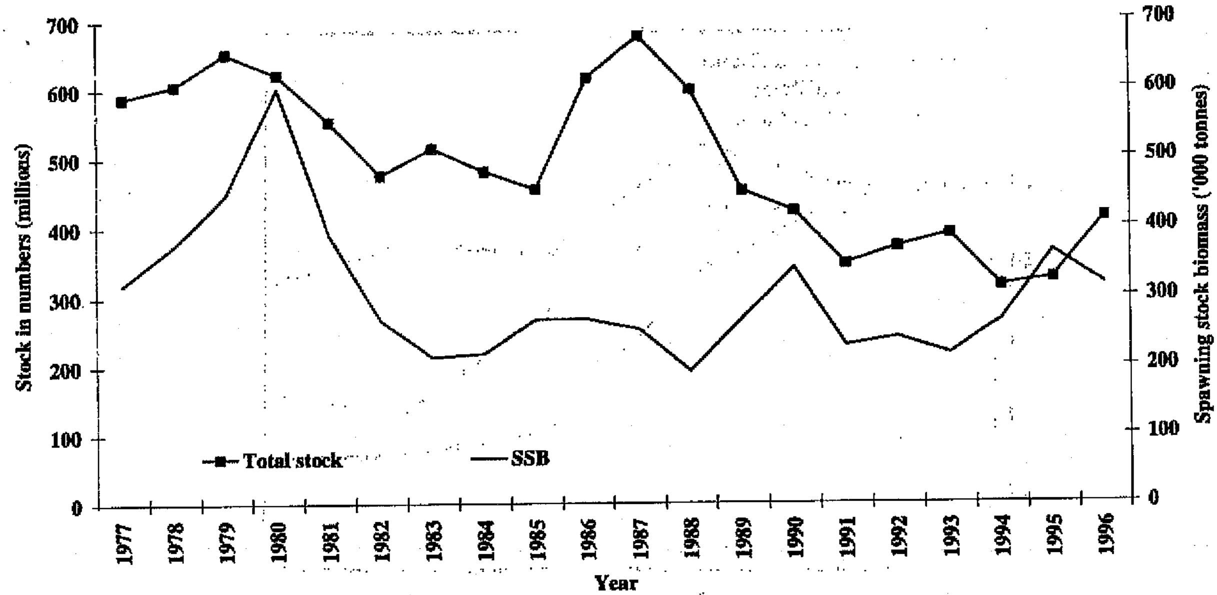


Figure 5.1.1.2.1.2. Cod at Iceland. Fishing mortality (unweighted average for age group 5–10 years). Data from ICES (1997).

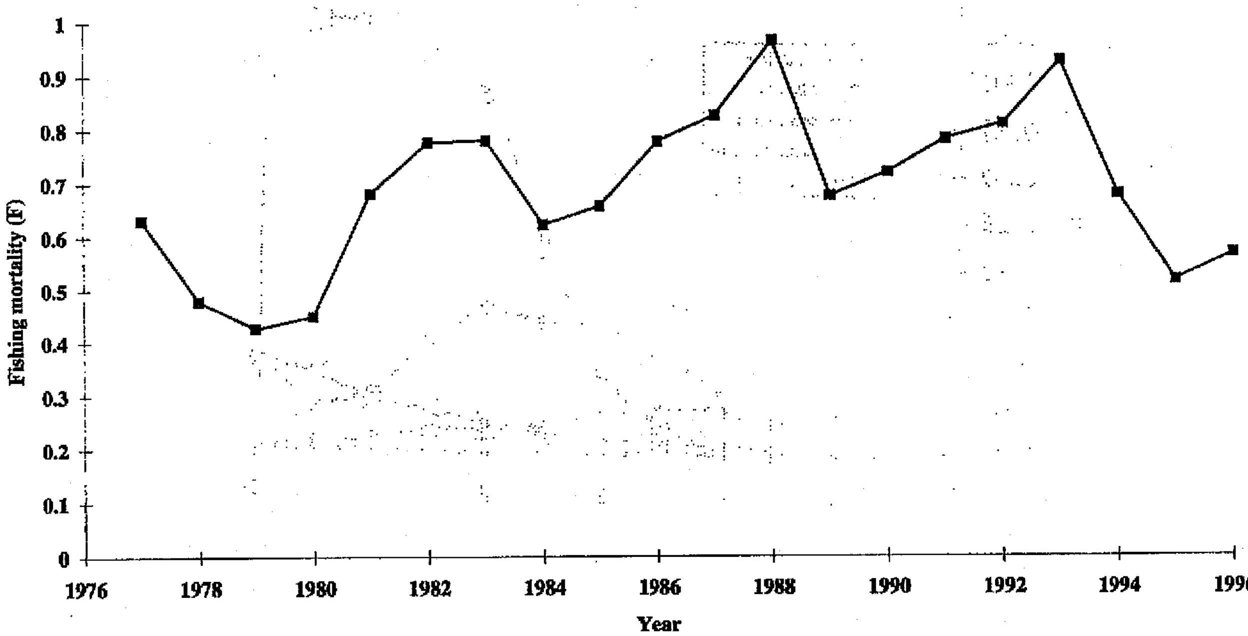


Figure 5.1.1.2.1.3. Trends in relative effort (1991 = 100) by fishing gear during 1991–1996 (Figure 2.1.3 in Anon., 1997).

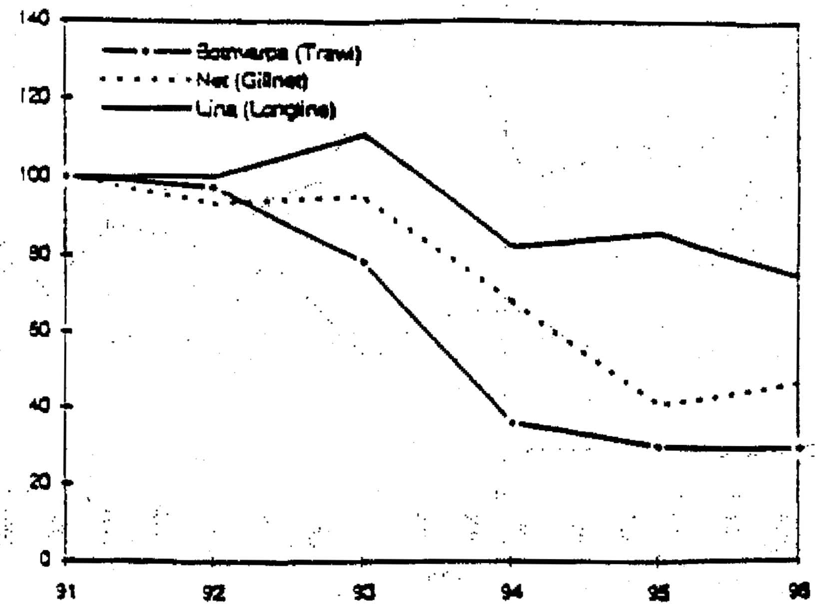


Figure 5.1.1.2.2.1. Fishing mortality by gear and age. Average over years 1992–1996 (Figure 3.3.2 in ICES, 1997).

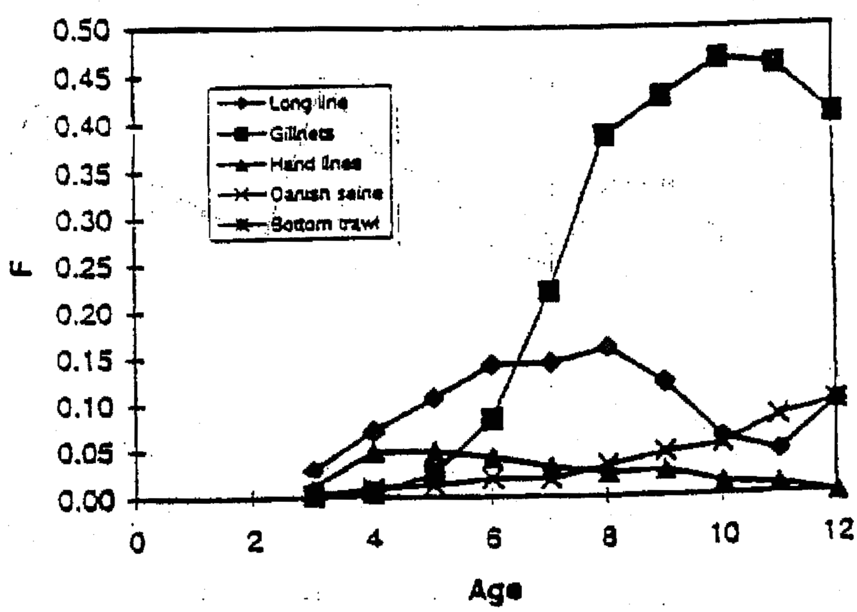


Figure 5.1.1.2.2.2.a. Relative size composition of the Icelandic cod stock 1985–1997. Size classes are 0–25 cm, 25–40 cm, 40–50 cm, 50–70 cm and > 70 cm total length.

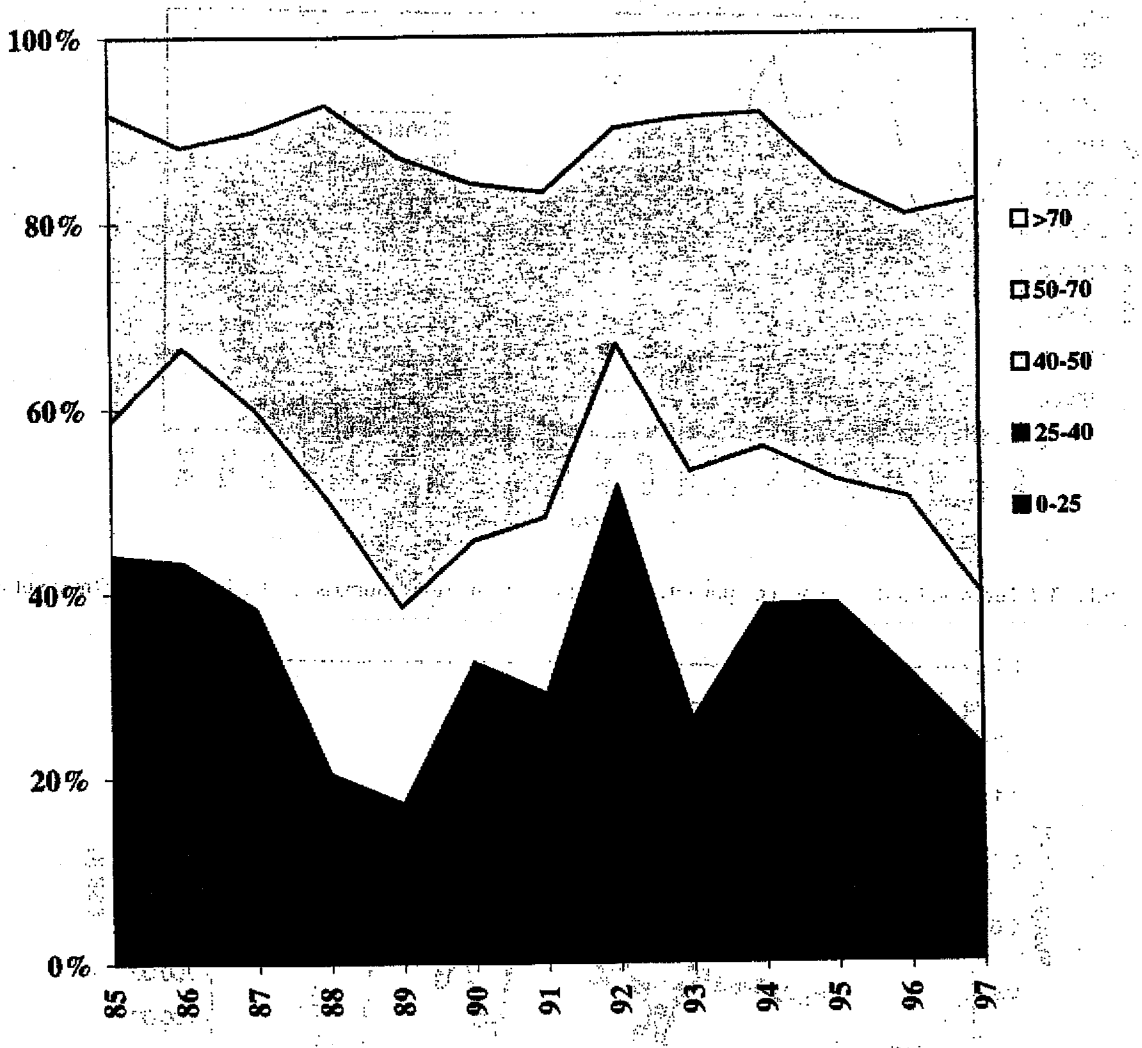


Figure 5.1.1.2.2.2.b. Average size of cod in Icelandic waters 1985–1997. Size classes are 0–25 cm, 25–40 cm, 40–50 cm, 50–70 cm.

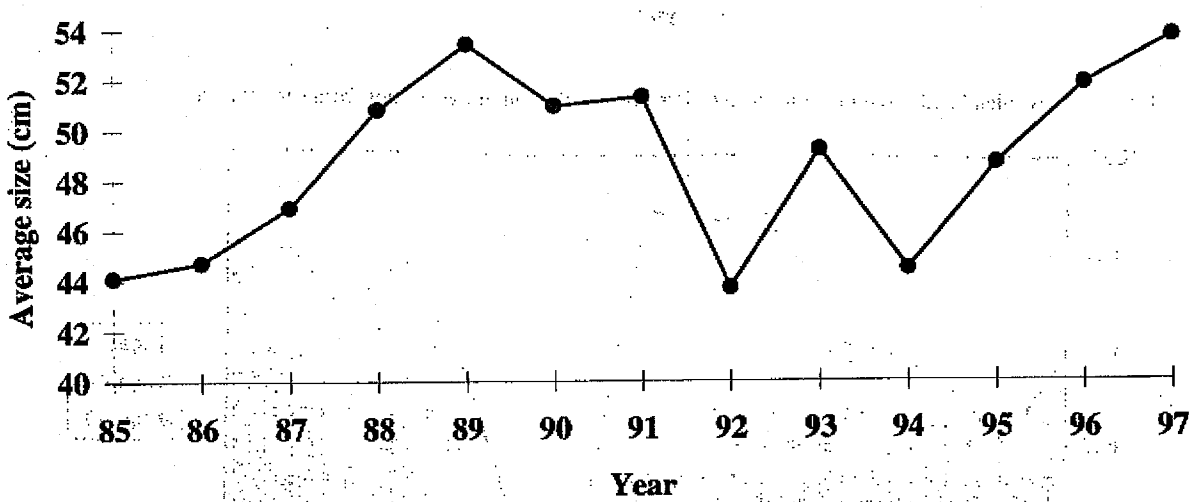


Figure 5.1.1.3.1. Cod catches (tonnes) from 1955–1995 off Greenland, divided into inshore and offshore components.

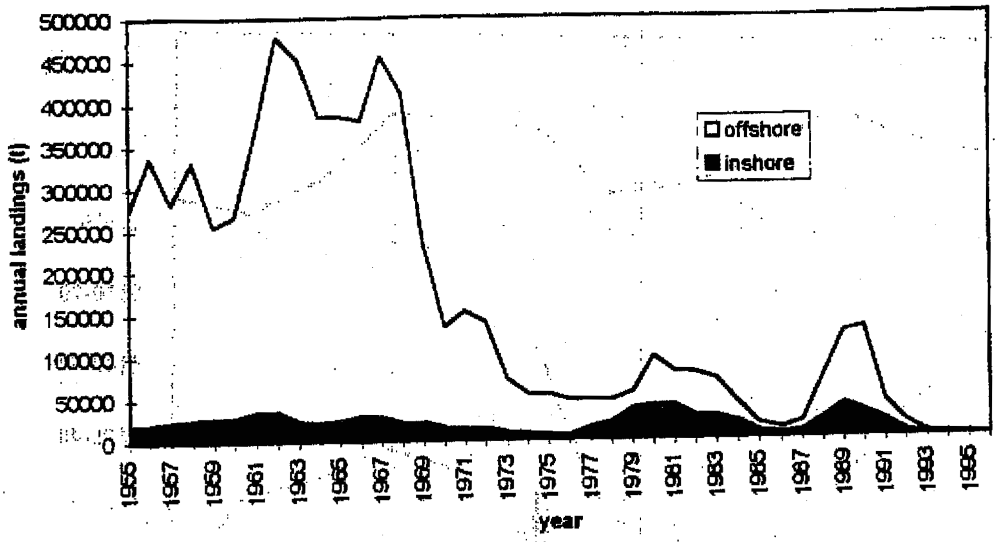


Figure 5.1.1.3.2. Greenland cod (offshore component). Trends in spawning stock biomass and fishing mortality (mean of age groups 5–8).

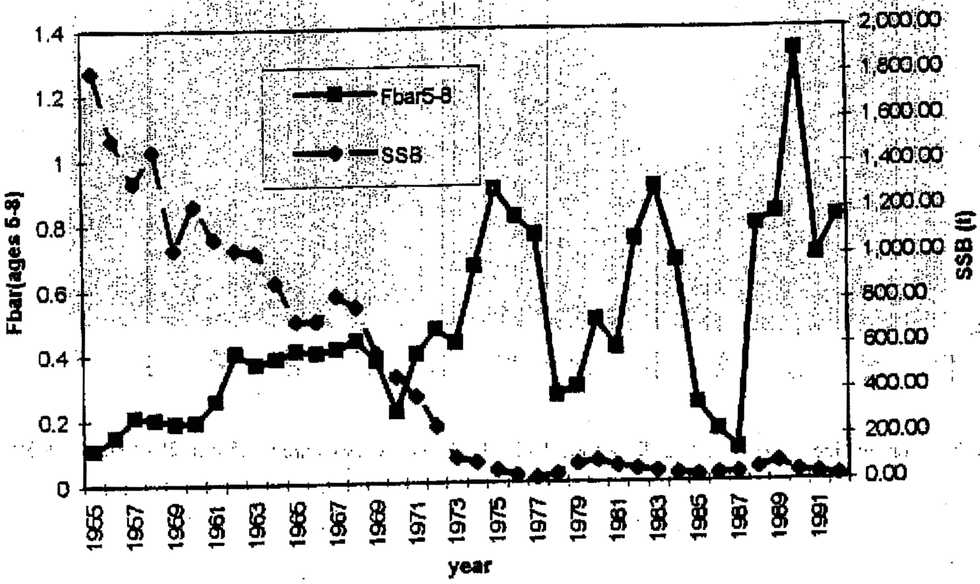


Figure 5.1.1.3.1.1. Greenland cod (offshore component). Percent distribution of age groups from survey data.

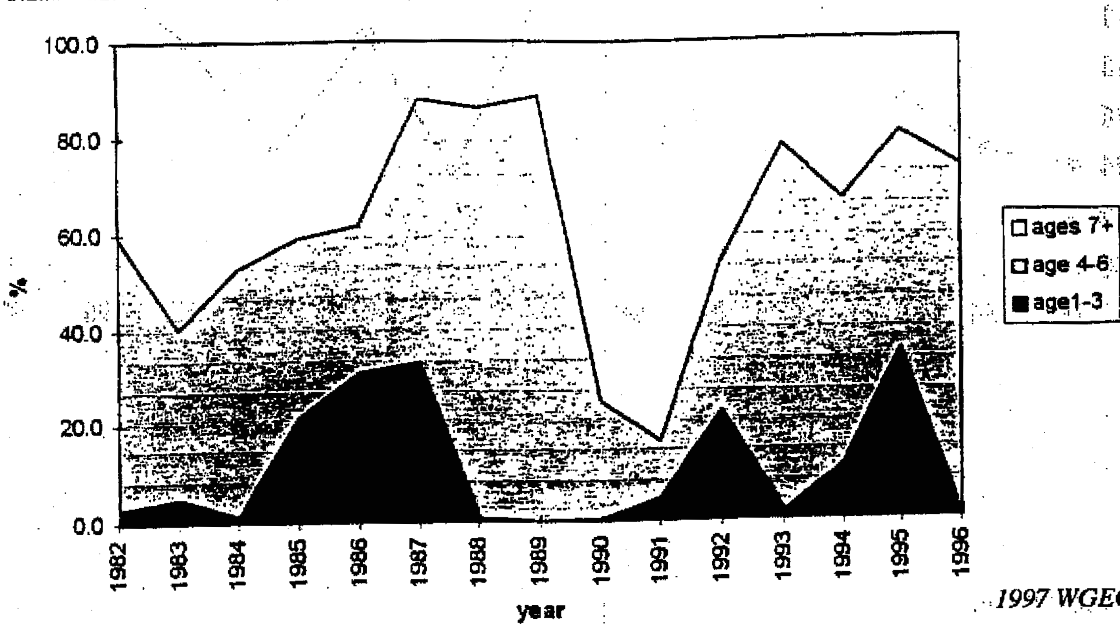


Figure 5.1.1.3.1.2. Abundance indices off West and East Greenland, and total for Atlantic cod.

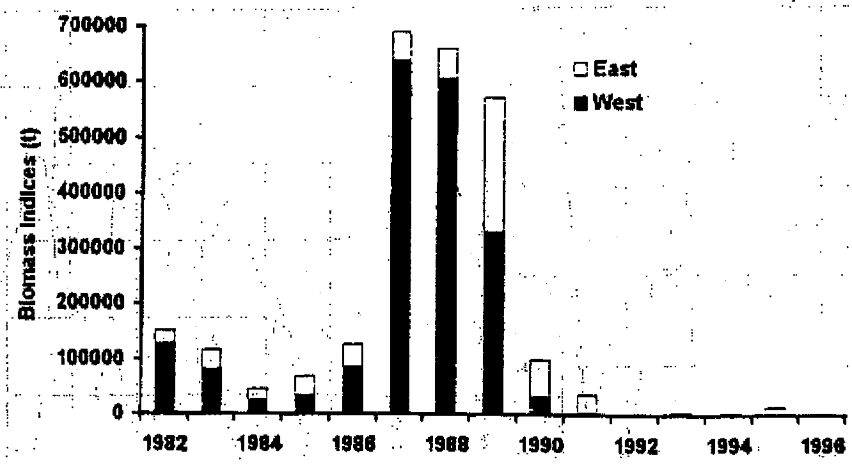


Figure 5.1.1.3.1.3. Mean individual weight of cod (kg) off West and East Greenland, and total for Atlantic cod.

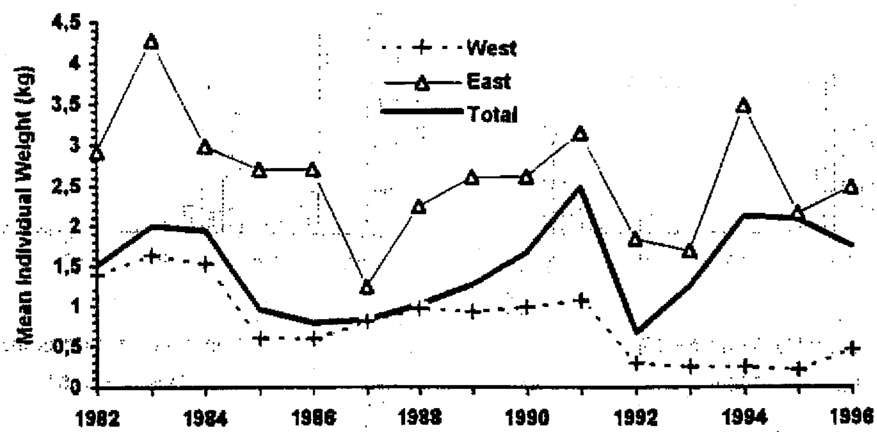


Figure 5.1.1.4.1.1. Landings (yield), fishing mortality, spawning stock biomass, and recruitment for cod in the Faeroe Plateau from 1961–1996.

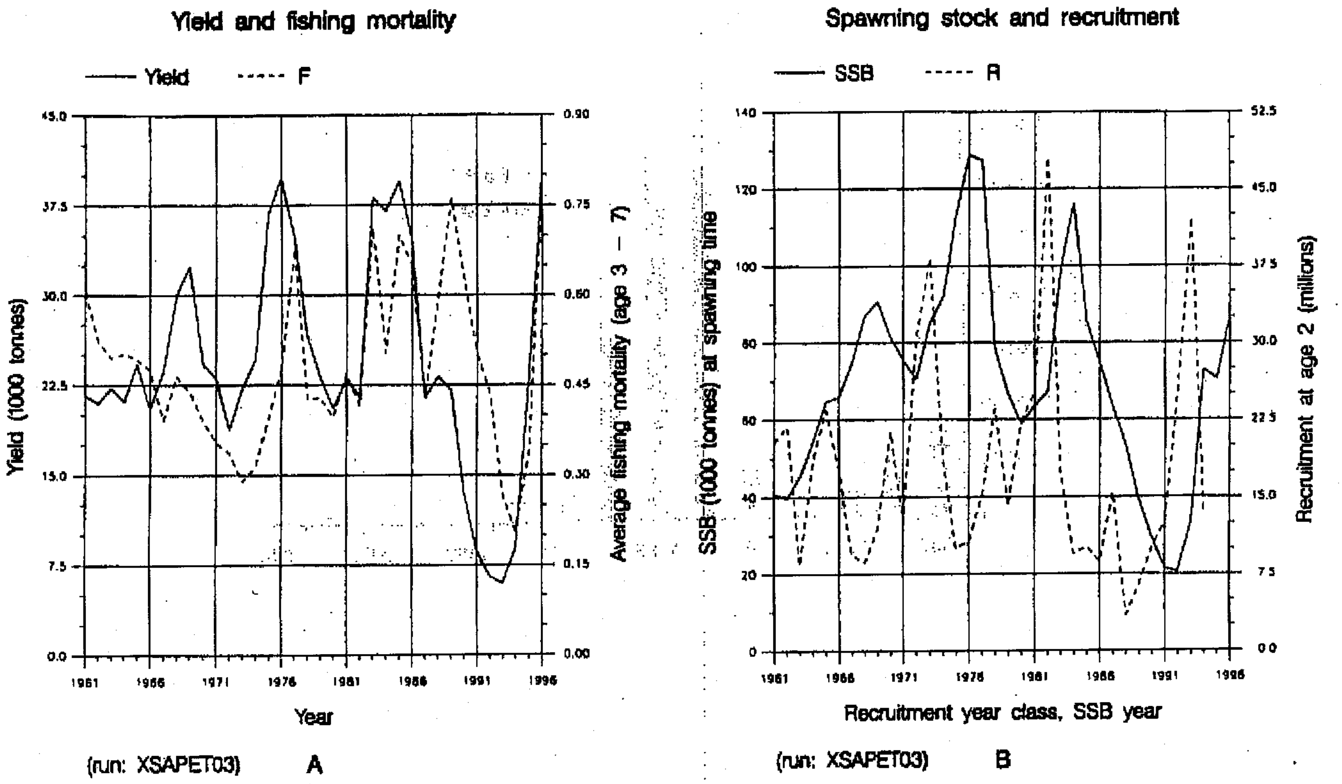


Figure 5.1.2.1.1.1. a) Landings and fishing mortality (ages 5–13), b) recruitment and spawning stock biomass of Norwegian spring-spawning herring from 1946–1995.

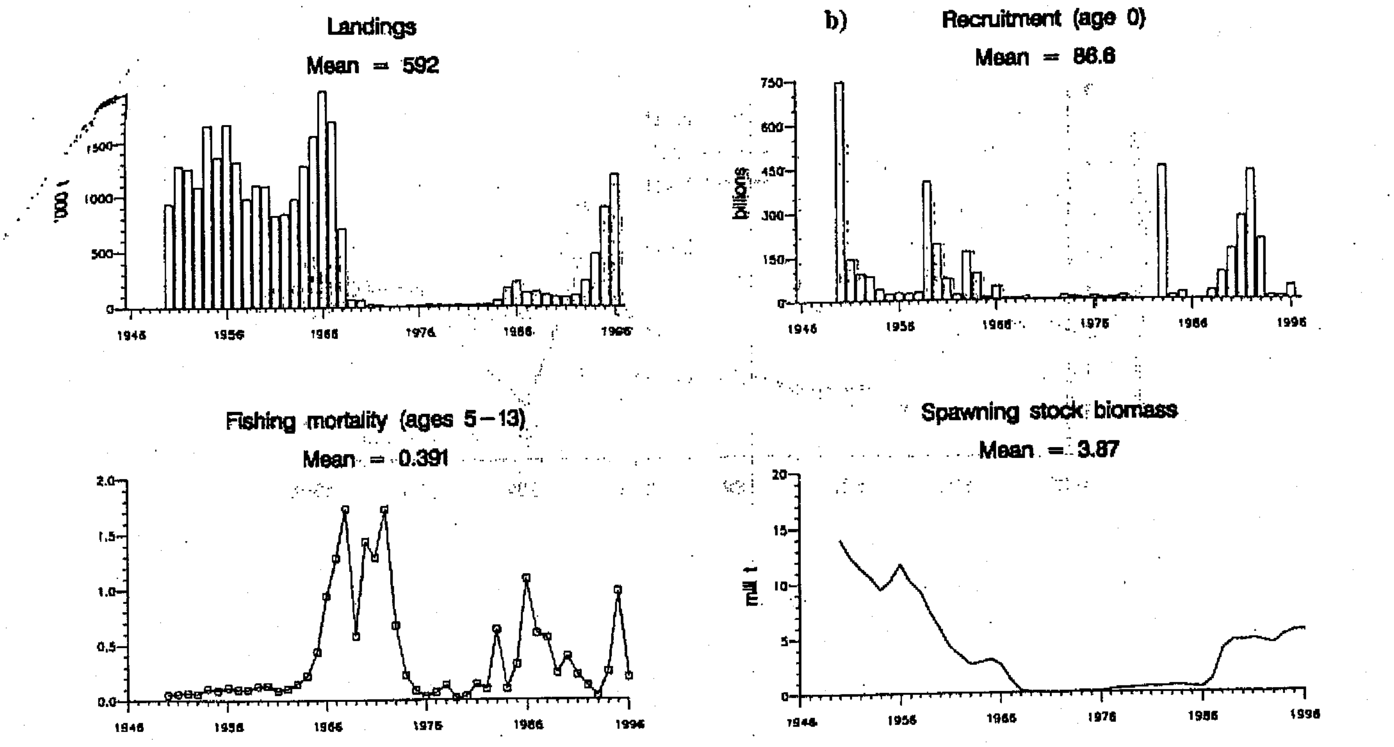


Figure 5.1.2.2.1.1. Total landings (thousands of tonnes) of Icelandic summer-spawning herring from 1951–1995 and fishing mortality (5+) during the period 1960–1995. Data from Jakobsson (1980) and ICES (1997b).

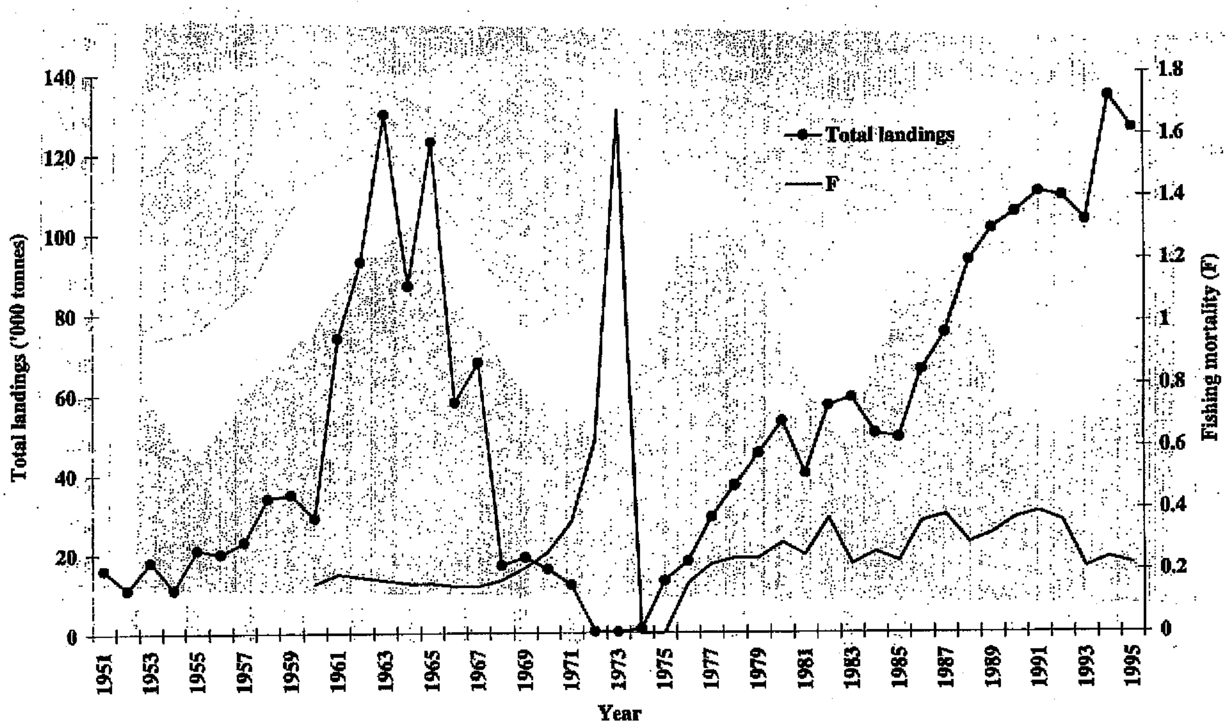


Figure 5.1.2.2.1.2. Total stock size of Icelandic summer-spawning herring (millions of individuals) and spawning stock biomass (thousands of tonnes).

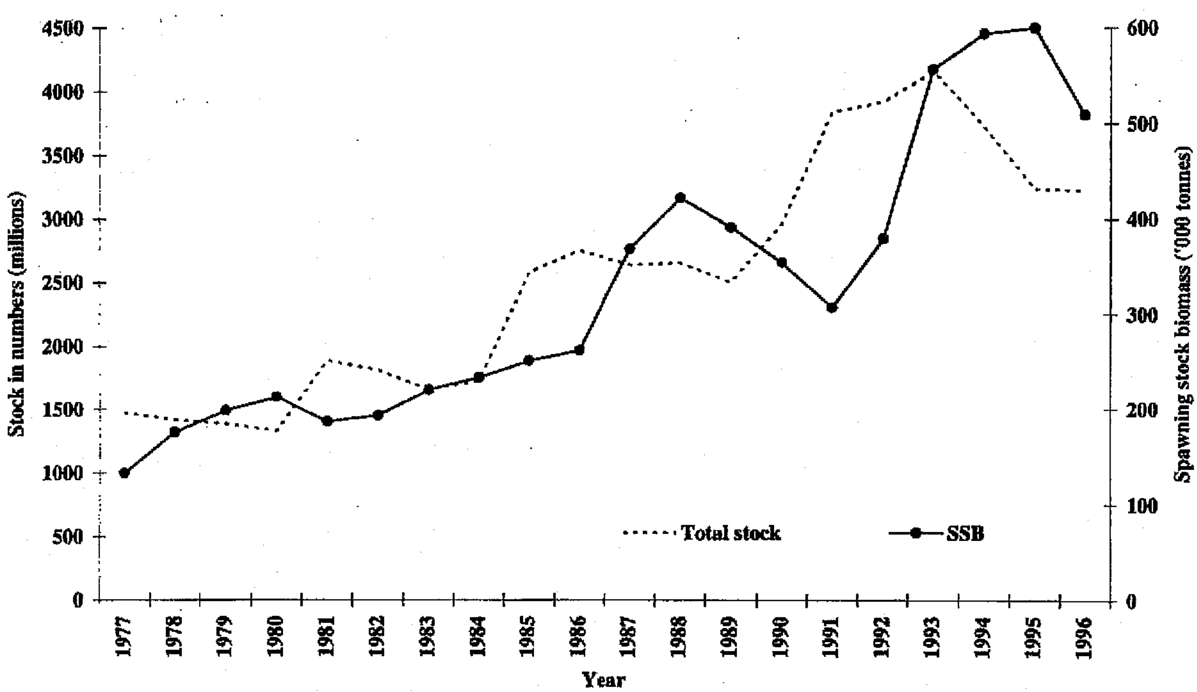
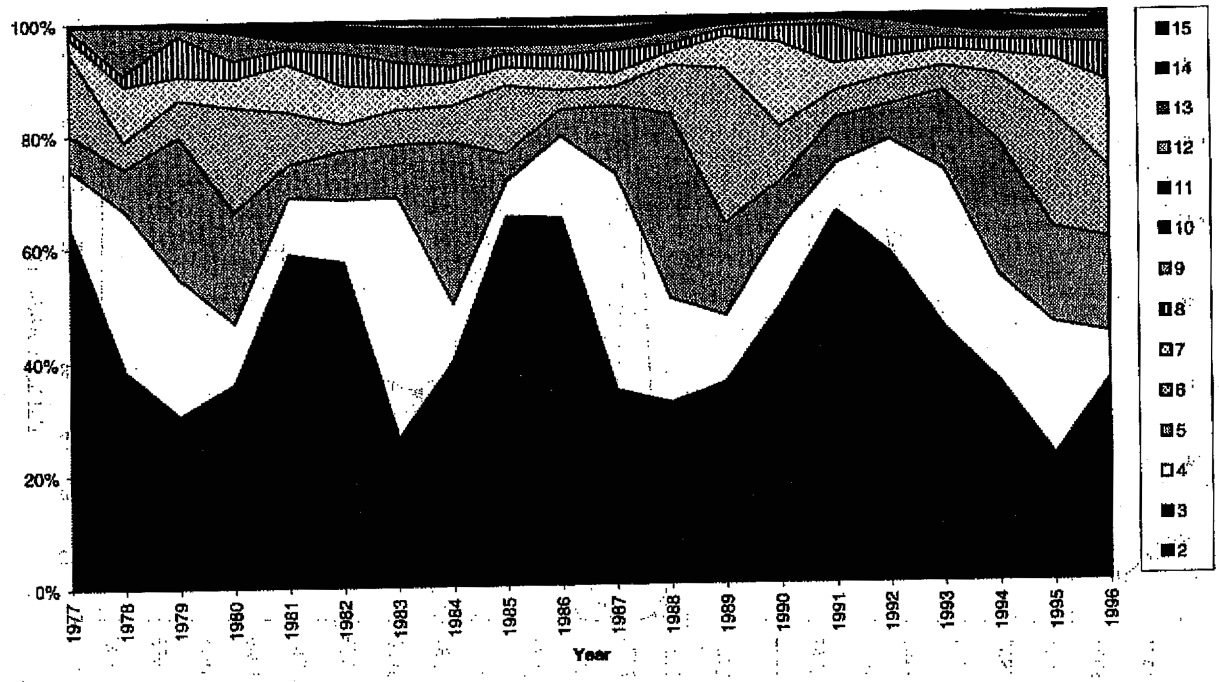


Figure 5.1.2.2.2.1. Relative stock number at age of Icelandic summer-spawning herring from 1977–1996.



5.2 OSPAR REGION II - Context of fisheries from ACFM

Demersal fisheries in the North Sea consist of human consumption fisheries and industrial fisheries. Human consumption fisheries target a mixture of roundfish species (cod *Gadus morhua*, haddock *Melanogrammus aeglefinus*, whiting *Merlangius merlangus*), or a mixture of flatfish species (plaice *Pleuronectes platessa* and sole *Solea solea*) with a by-catch of roundfish. The industrial fisheries use small mesh trawls and target mainly sandeels *Ammodytes* sp., Norway pout *Trisopterus esmarki*, and sprat *Sprattus sprattus*. The industrial catches also contain a by-catch of other species, including herring *Clupea harengus*, haddock, and whiting.

In the North Sea, all stocks of roundfish and flatfish species have been exposed to high levels of fishing mortality for a long period. For most of these stocks, their lowest observed spawning stock size has been seen in recent years. The present assessments for roundfish indicate a decline in fishing mortality in recent years for cod, haddock, and whiting. The reason for this decline is unclear. The decline is somewhat supported by a reduction in effort by some of the major fleets in the last few years and by a divergence of effort to *Nephrops* and anglerfish. However, this decline may also be artificial, since it appeared in the assessment for the first time. Fishing mortality on plaice and sole has been varying at a high level over a long period with no trend. Sandeel landings averaged around 600,000 t annually between 1976 and 1986, increasing to 800,000 t between 1987 and 1996, and have increased to record levels in the first half of 1997. The sandeel spawning stock has fluctuated around one million tonnes with no obvious trend. The 1996 sandeel year class appears to have been very strong. Norway pout landings decreased over the period 1974 to 1988 and thereafter fluctuated around 200,000 t. The spawning stock declined to its lowest level in 1986 and has since increased, partly due to a good year class in 1994.

The Skagerrak/Kattegat area is, to a large extent, a transition area between the North Sea and the Baltic Sea. Several of the stocks in the Skagerrak show close affinities to the North Sea stocks, including cod, haddock, whiting, saithe, hake, plaice, and Norway pout, in terms of population dynamics. Tagging experiments suggest extensive migration between the two areas. Both human consumption and industrial fisheries operate in the Skagerrak/Kattegat. Human consumption fleets include gillnetters and Danish seiners exploiting flatfish and cod as well as demersal trawlers targeting roundfish, flatfish, *Nephrops*, and *Pandalus*. Demersal trawling is also used in the industrial fisheries for Norway pout and sandeel. Pelagic trawlers exploit herring, mackerel, horse mackerel, and sprat. The main herring stocks exploited in the area are North Sea autumn-spawners and the stock of spring-spawners spawning in the western Baltic and the southern part of Division IIIa. Most of the stocks are assessed in conjunction with the stocks in the neighbouring areas: cod and autumn-spawning herring with the North Sea stocks, spring-spawning herring with the western Baltic stocks.

A large proportion of the Eastern Channel is in the coastal zone (12-mile limit) which is exploited by small-scale fisheries. The major fleets operating in this area are: a French inshore fleet consisting mainly of small vessels using a variety of gears; an English inshore fleet using fixed gear; English and Belgian offshore beam trawlers; and French offshore otter trawlers. The beam trawl fleets target sole, with a significant plaice by-catch. Sole is also taken by the two inshore fisheries using trammels and, in the case of the French inshore fleet, otter trawls. Plaice is targeted by the French offshore trawlers with sole taken as a by-catch, while the major part of the cod landings originate from French offshore trawlers and inshore gillnetters. Whiting is caught in mixed fisheries inshore and offshore by French trawlers. Effort directed at flatfish increased consistently and considerably in all fleets from 1975, reaching a peak in 1989–1990, after which it has stabilized. A pelagic fishery operates in winter during the herring spawning season.

Up to and including 1995, cod, whiting, plaice, and sole in Division VIId were assessed as separate stocks. Review of the stock identity of these species indicated that sole should continue to be considered as a separate stock, but that there were strong links between the Eastern Channel and the southern North Sea for cod, plaice, and whiting. The pelagic fish species herring (Downs herring), horse mackerel, mackerel, and sprat are subject to Total Allowable Catches (TACs) set over larger areas.

5.2.1 Cod

5.2.1.1 North Sea cod

5.2.1.1.1 Context - patterns of numbers, biomass, landings, and fishing mortality

Landings of cod increased from 108,000 t in 1963 to 341,000 t in 1972, fluctuated around the 200,000 t to 250,000 t level from 1972 to 1981, then declined steadily to 88,000 t in 1994 (Figure 5.2.1.1.1.1; Serchuk *et al.*, 1996). Over the last 30 years, fishing mortality has increased almost continuously, doubling over the period 1963–1989, until stabilizing at around F = 0.9. Since 1981 fishing mortality has been in excess of the biological reference point, F_{med}, indicating recruitment overfishing, and the landings have become dominated by 2-year-old immature fish. Furthermore, discards of one-year-old cod have been considerable in some years. Since 1983, the spawning stock biomass has been below MBAL, the level of spawning stock size below which the probability of poor recruitment increases as spawning stock size decreases (ICES, 1992), and year class strength was below average between 1987 and 1993. A good year class in 1993, however, resulted in an increase of spawning stock biomass to around 100,000 t in 1996 and landings have again increased to 126,000 t. Another good year class in 1996, together with an apparent drop in fishing mortality in recent years, are both favourable indicators for the status of the stock in the short term (ACFM report on Stocks in the Skagerrak and Kattegat, the North Sea, and the Eastern Channel, October 1997). WGECO notes, however, that in the recent past it has been widely reported that recruitment of strong year classes of gadoids to the fishery has been accompanied by high levels of discarding. This risk of losing the potential benefit of the strong 1996 year class, combined with the very precarious state of the stock, as shown by Cook *et al.* (1996), leaves the Working Group pessimistic about the likelihood of noteworthy improvement of the stock in the medium term.

5.2.1.1.2 Changes in size distribution and/or age composition

Figure 5.2.1.1.2.1 shows variation in the proportion of cod sampled in the International Beam Trawl Survey (IBTS) that belonged to different length classes. Clear trends are not obvious, though since 1976 smaller fish have tended to dominate the catches. The important point to note is that from 1976 larger cod are rarely encountered. The two points where cod (> 15 cm) again dominated the survey catches after 1976 probably reflect years of poor recruitment more than anything else. Plotting the trends in the numbers of cod at age as determined by the Single Species Virtual Population Analysis (SSVPA) reveals a clear shift towards younger fish (Figure 5.2.1.1.2.2). This trend is associated with the steady increase in fishing mortality over this period.

5.2.1.1.3 Changes in spatial distribution

Scottish August Groundfish Survey (AGFS) data were analysed to investigate changes in the distribution of cod in the northwestern North Sea. Data for two three-year periods, 1982–1984 and 1992–1994, were extracted from the database. The North Sea cod stock declined from around 130,000 t to 60,000 t over the intervening period. Mean catch rates in each statistical rectangle were determined, and any rectangle sampled less than twice in one of the three-year periods was excluded from the analysis. The data were gridded using a multiquadric radial basis function based on the mean trawl positions in each rectangle in each of the three-year periods. The distributions of cod in each period are shown in Figure 5.2.1.1.3.1. Although there is a superficial similarity between the two distributions, a plot of the difference between them indicates marked changes (Figure 5.2.1.1.3.2). Catch rates in areas of high density in 1982–1984 have decreased markedly, while catch rates in nearby low density areas have increased. The population distribution has shifted, but it is difficult to relate this to fishing activity in the absence of effort data. Certainly, no evidence of a retraction of the distribution associated with the decline in the North Sea cod biomass is indicated.

Cod is widely distributed over the North Sea. Data from the International Bottom Trawl Survey in February from 1971–1991 show that age group 1 is most abundant in the southern part of the North Sea, although in certain years most of the catch of this age group was taken in the central part. Two-year-old cod is more evenly distributed, and age three and older fish are mainly found in the northern North Sea (Heessen, 1993). Figure 5.2.1.1.3.3 shows the distribution of cod by age group from 1983–1987. Compared to earlier years, a smaller proportion of the juvenile cod is now found in the southern North Sea and German Bight. It is not known whether this is due to the low number of juveniles produced by the stock in these years or to changes in environmental parameters. With respect to temperature, previous analysis of changes in cod distribution does not suggest a temperature preference, at least not for the juveniles (Heessen and Daan, 1994).

More information regarding changes in cod distribution is given in Section 5.3.1, below.

5.2.1.2 Skagerrak and Kattegat

5.2.1.2.1 Context - patterns of numbers, biomass, landings, and fishing mortality

The state of cod in the Kattegat is uncertain, but indications are that it has been declining for two decades until recently. Landings in 1996 from the Skagerrak were 16,400 t (plus a by-catch of 900 t from the industrial fishery), compared with 12,100 t in 1994. Landings from the Kattegat were 6,100 t in 1996.

5.2.1.2.2 Changes in size distribution and/or age composition

No information on size distribution or age composition was available.

5.2.1.2.3 Changes in spatial distribution

No information on spatial distribution was available.

5.2.2 Herring

5.2.2.1 North Sea herring

5.2.2.1.1 Context - patterns of numbers, biomass, landings, and fishing mortality

Herring are taken in several different fisheries in this region including, for example, the directed herring fishery (mainly for human consumption) in the North Sea as well as the industrial fishery in the Kattegat. The overall development is shown in Figure 5.2.2.1.1.1 (Serchuk *et al.*, 1996). From 1951 to 1963 landings fluctuated between 600,000 t and 800,000 t, during which time fishing mortality was relatively stable at around F = 0.4. Landings then peaked briefly in 1965, at approximately 1.2 million t, then declined markedly to exceptionally low levels of less than 50,000 t in 1978. This coincided with a marked and sudden increase in fishing mortality, to values exceeding F = 1.0 for most of the 1968 to 1976 period, associated with the rapid expansion of the purse seine fishery. The fishery for herring in the North Sea was closed in 1977 to allow stocks to recover and, on reopening in the early 1980s (1981 southern North Sea, 1983 northern North Sea), landings increased quickly to nearly 900,000 t in the late 1980s before dropping back again. Landings in 1994 and 1995 were less that 600,000 t, but in 1996 they dropped to 264,000 t. Since 1986 fishing mortality has fluctuated around F = 0.6, although exploitation of juvenile herring in the small mesh sprat fishery has increased substantially in recent years, reducing the long-term yield of adult herring and diminishing the future reproductive potential of the stock. The herring spawning stock in the North Sea has been below the MBAL level of 800,000 t since 1992. In 1995 ACFM decided that the herring stock was outside safe biological limits and that if current levels of exploitation were maintained it would be unlikely that MBAL would be regained.

5.2.2.1.2 Changes in size distribution and/or age composition

The IBTS data suggest that the proportion of herring caught that belong to the smaller size classes has declined slightly over the 26-year period albeit with some marked year-to-year fluctuations. The mean size appears to have increased somewhat (Figure 5.2.2.1.2.1). This appears contradictory to the VPA proportion at age data, where the proportion of one-year old and older fish in the population declined steadily during the 1960s. Data for only every fifth year, and 1997, were analysed here to detect trends. Apart from an increase in one-year olds in 1975, this pattern has remained rather constant during the 1980s and 1990s (Figure 5.2.2.1.2.2).

5.2.2.1.3 Changes in spatial distribution

WGECO felt that this question would be better addressed by the Herring Assessment Working Group (HAWG).

5.2.3 Sole

5.2.3.1 North Sea sole

5.2.3.1.1 Context - patterns of numbers, biomass, landings, and fishing mortality

Sole are exploited in a mixed (with plaice) beam trawl fishery in the southern North Sea and in a directed gillnet fishery in coastal areas. The overall development of the fishery is shown in Figure 5.2.3.1.1.1 (Serchuk *et al.*, 1996). Landings doubled, from around 12,000 t to 26,000 t, from 1957 to 1962, coinciding with a peak in the spawning stock biomass of 150,000 t which was due to the recruitment of the outstanding 1958 year class. High natural mortality associated with the severe 1962/1963 winter weather conditions caused a marked drop in the spawning stock biomass in 1964 and 1965 and this was reflected in a sharp fall in landings to 10,000 t in 1964. However, another good year class in 1963 resulted in an increase in both spawning stock biomass and landings in 1967 to 100,000 t and over 30,000 t, respectively. Over this period, fishing mortality increased sharply and continuously, from F < 0.02 to F > 0.4. Over the next 15 years fishing mortality continued to increase steadily, reaching record levels in 1985 of F > 0.5. The spawning stock declined to around 40,000 t in 1974, while landings fell back to around 20,000 t in 1970 and both continued to fluctuate around these levels until 1989. A strong year class in 1987, coinciding with a slight drop in fishing mortality, back to F = 0.4, resulted in a marked increase in the spawning stock biomass to around 80,000 t in 1990, while landings increased to record high levels of approximately 35,000 t. A further good year class in 1991 sustained both spawning stock biomass and landings, while fishing mortality again increased to F > 0.5 in 1993. Fishing mortality has exceeded both F_{max} (0.23) and F_{med} (0.33) since the late 1960s, suggesting both growth and recruitment overfishing. Despite this, however, the sole stock was considered to be within safe biological limits and the spawning stock biomass remained above MBAL and was expected to remain so over the short to medium term. However, extra natural mortality in the 1995–1996 winter appears to have affected the stock size considerably. Because the mortality level could not be quantified, the present state of the stock is uncertain, but it is believed to be below an agreed MBAL of 35,000 t in 1997. Landings decreased in 1996 to 22,500 t.

5.2.3.1.2 Changes in size distribution and/or age composition

The proportion of the sole population in each length category in the IBTS data set has fluctuated markedly over time. Likewise, average length has also shown considerable variation (Figure 5.2.3.1.2.1). In neither representation of the data was any particular trend apparent over time. However, the VPA data suggest a clear decrease in the proportion of fish of six years of age and older in the population and this trend has been steady since the late 1960s (Figure 5.2.3.1.2.2). The proportion of two- and three-year old fish has increased. These trends are associated with the increases in fishing mortality discussed in the previous section.

5.2.3.1.3 Changes in spatial distribution

Changes in the distribution of sole are addressed in Section 5.3.3, below.

5.2.3.2 Skagerrak and Kattegat

5.2.3.2.1 Context - patterns of numbers, biomass, landings, and fishing mortality

The catches of sole in 1996 amounted to 1059 t. The stock size is not known precisely, but data from the fishery and surveys indicate that the stock was exceptionally high in the period 1988–1996. Recruitment now seems to be back to the pre-1988 level.

5.2.3.2.2 Changes in size distribution and/or age composition

No information on size distribution or age composition was available.

5.2.3.2.3 Changes in spatial distribution

No information on spatial distribution was available.

5.2.3.3 Eastern Channel

5.2.3.3.1 Context - patterns of numbers, biomass, landings, and fishing mortality

Fishing mortality has increased from 0.36 in 1995 to 0.48 in 1996. After an increase following strong recruitment in the period 1989–1991, the spawning stock has decreased for two years, but stays above the historical minimum of 7000 t. In recent years, TACs for sole have not been restrictive. However, at the current level of fishing mortality, there is a relatively high probability (65 %) of the spawning stock biomass falling below 7800 t.

5.2.3.3.2 Changes in size distribution and/or age composition

See Section 5.3.3, below.

5.2.3.3.3 Changes in spatial distribution

See Section 5.3.3, below.

5.2.4 Mackerel

Two mackerel stocks are exploited within this region, the Western and the North Sea stocks (Serchuk *et al.*, 1996). Mackerel, being a highly migratory species, are harvested at different times of the year in different OSPAR regions. The bulk of the catch of the combined stocks (North Sea, Western, and Southern) is taken in OSPAR Regions II and III. Changes in size/age composition and spatial distribution are considered in Sections 5.3.4 and 5.4.4, below.

5.2.4.1 North Sea stock

5.2.4.1.1 Context - patterns of numbers, biomass, landings, and fishing mortality

The development of this stock is shown in Figure 5.2.4.1.1.1. Before 1964, annual landings of the North Sea mackerel stock were less than 100,000 t. Spawning stock biomass at this time exceeded 3 million t (Jones, 1983). Development of purse seine technology in the early 1960s allowed an almost ten-fold increase in landings to more than 900,000 t in 1967. This was followed by a drastic decline in landings to less than 200,000 t in 1971, during which time spawning stock biomass fell by 80 %. Strong recruitment from the 1969 cohort resulted in a relatively small, and temporary, rise in spawning stock biomass and landings, but by 1980, spawning stock biomass had again fallen to less than 200,000 t and during 1979 to 1986 landings ranged between 25,000 t and 66,000 t. After 1985 the spawning stock biomass has been estimated at between 50,000 t and 100,000 t. From 1989 it has proved impossible to allocate catches of mackerel taken in the North Sea to either the North Sea or the Western stocks; catches from the North Sea stock have been assumed to be 10,000 t annually. The North Sea stock is considered to be outside safe biological limits and to require the maximum possible protection. Since 1980 ACFM have recommended that no catches be taken, however, this can only be achieved by closing all mackerel fisheries in areas where North Sea mackerel occur. Consequently, ACFM has recommended since 1991 that: no fishing for mackerel be allowed in Areas IIIa, IVb, and IVc at any time of the year; there should be no fishing for mackerel in Area IVa between 1 January and 31 July; and the minimum landing size of 30 cm and existing by-catch regulations in Division IIIa and Area IV be maintained. These regulations may encourage mis-reporting and discarding.

5.2.4.1.2 Changes in size distribution and/or age composition

Figure 5.2.4.1.2.1 shows variation in the proportion of mackerel at length in the IBTS data set. The proportion of small fish in the catches has been consistently high since 1986. Prior to this it was more variable. Average length was also variable up to 1985, then declined in 1986 and has since remained low. For further details, see Sections 5.3.4 and 5.4.4, below.

5.2.4.1.3 Changes in spatial distribution

See Sections 5.3.4 and 5.4.4, below.

5.2.4.2 Western Stock

5.2.4.2.1 Context - patterns of numbers, biomass, landings, and fishing mortality

Despite the collapse of the North Sea mackerel stock, landings of mackerel from the North Sea have increased from 50,000 t in 1985 to nearly 475,000 t in 1994 due to a shift in the annual migration pattern in the Western mackerel stock. Since 1986, landings of Western mackerel taken in the North Sea have accounted for over 50 % of the total landings from this stock; in earlier years this proportion was less than 10 % (Figure 5.2.4.2.1.1). Up to 1994 the Western mackerel stock was considered to be well within safe biological limits. However, the spawning stock biomass has since declined to a record low level, while fishing mortality levels have reached a record high. Landings in 1995 declined to 322,000 t. The combined mackerel stock (Southern, Western and North Sea, of which the Western stock is the dominant component) may now be outside safe biological limits. ACFM has determined that a significant reduction in fishing mortality in all areas where mackerel are caught, including international waters, is necessary to reverse the decline in spawning stock biomass; a 40 % reduction in fishing mortality is needed to prevent the spawning stock from further declining.

5.2.4.2.2 Changes in size distribution and/or age composition

See Sections 5.3.4 and 5.4.4, below.

5.2.4.2.3 Changes in spatial distribution

See Sections 5.3.4 and 5.4.4, below.

5.2.5 Hake

Hake are not targeted by the fisheries in this region.

5.2.6 References

Cook *et al.* 1996.

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ICES. 1992. The form of ACFM advice. ICES Cooperative Research Report No. 179, pp. 4–9.

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Serchuk, F.M., Kirkegaard, E., and Daan, N. 1996. Status and trends of the major roundfish, flatfish, and pelagic fish stocks in the North Sea: thirty-year overview. ICES Journal of Marine Science, 53: 1120–1129.

Figure 5.2.1.1.1.1. Trends in landings (——) and fishing mortality (– – –) (top) and spawning-stock biomass (——) and recruitment (– – –) (bottom) for North Sea cod, 1963–1994 (data from ICES, 1996c).

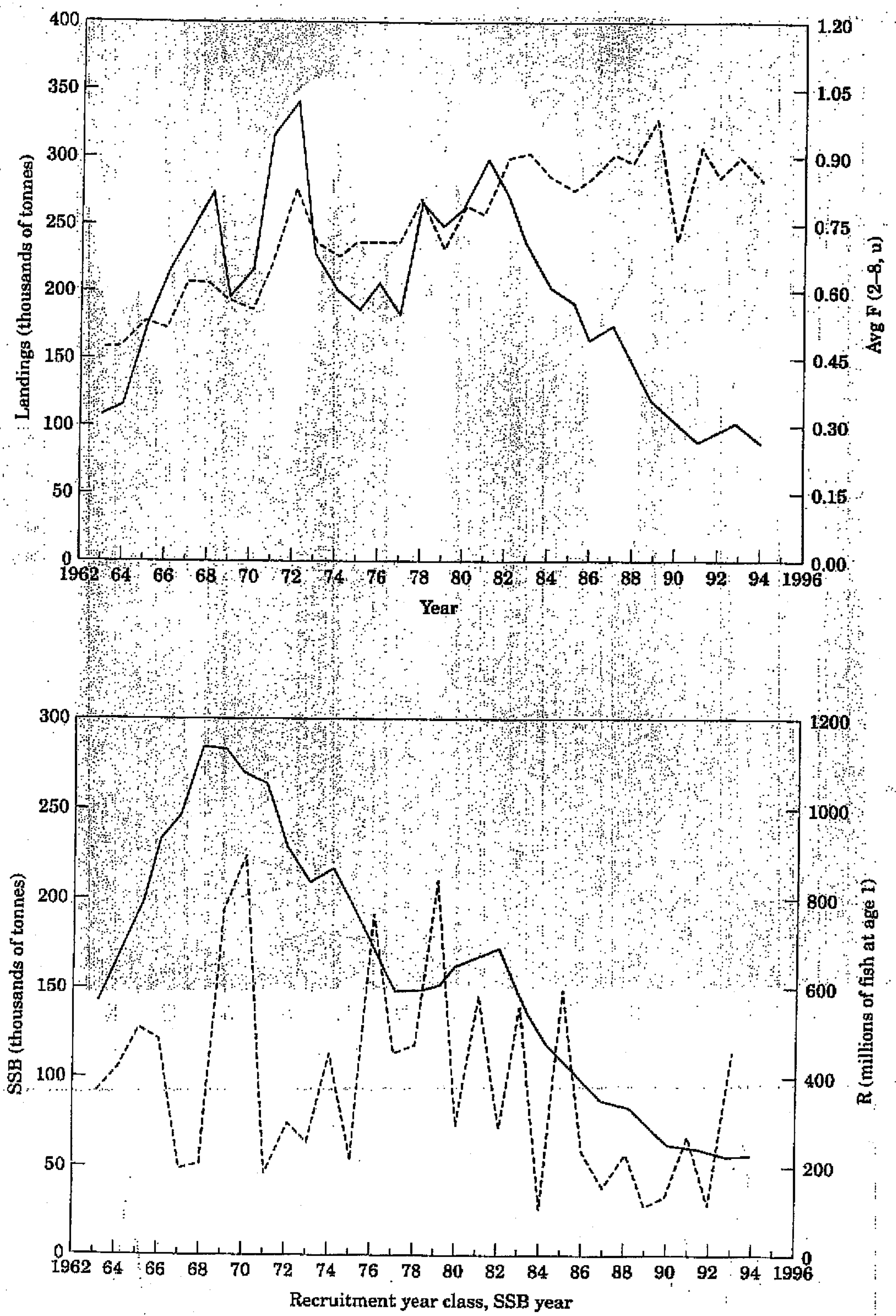


Figure 5.2.1.1.2.1. Changes in time of the proportion of cod at different length classes in the IBTS data set (top) and trend in average size (bottom).

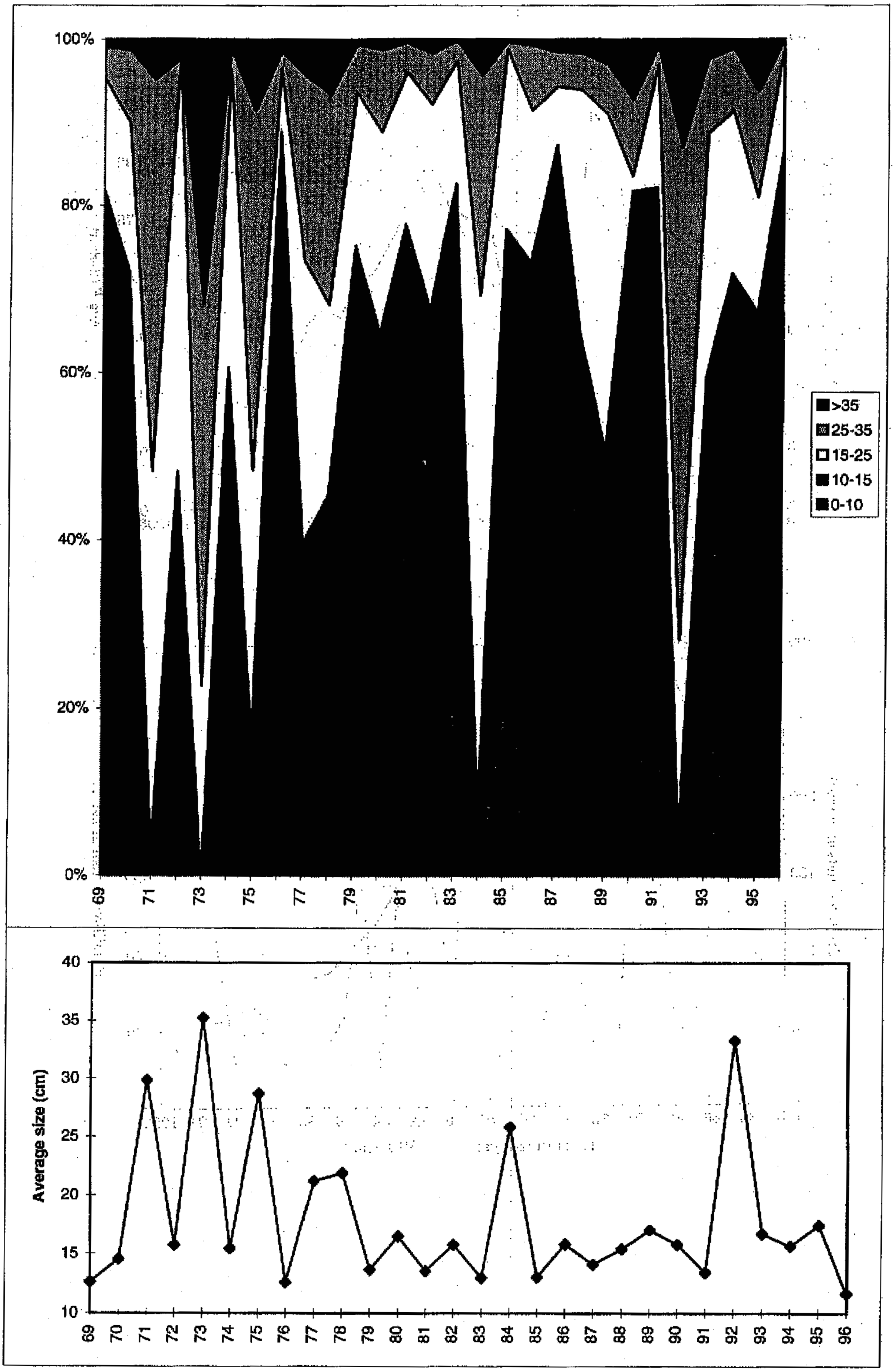


Figure 5.2.1.1.2.2. Variation in the proportion of cod at age over the duration of the VPA.

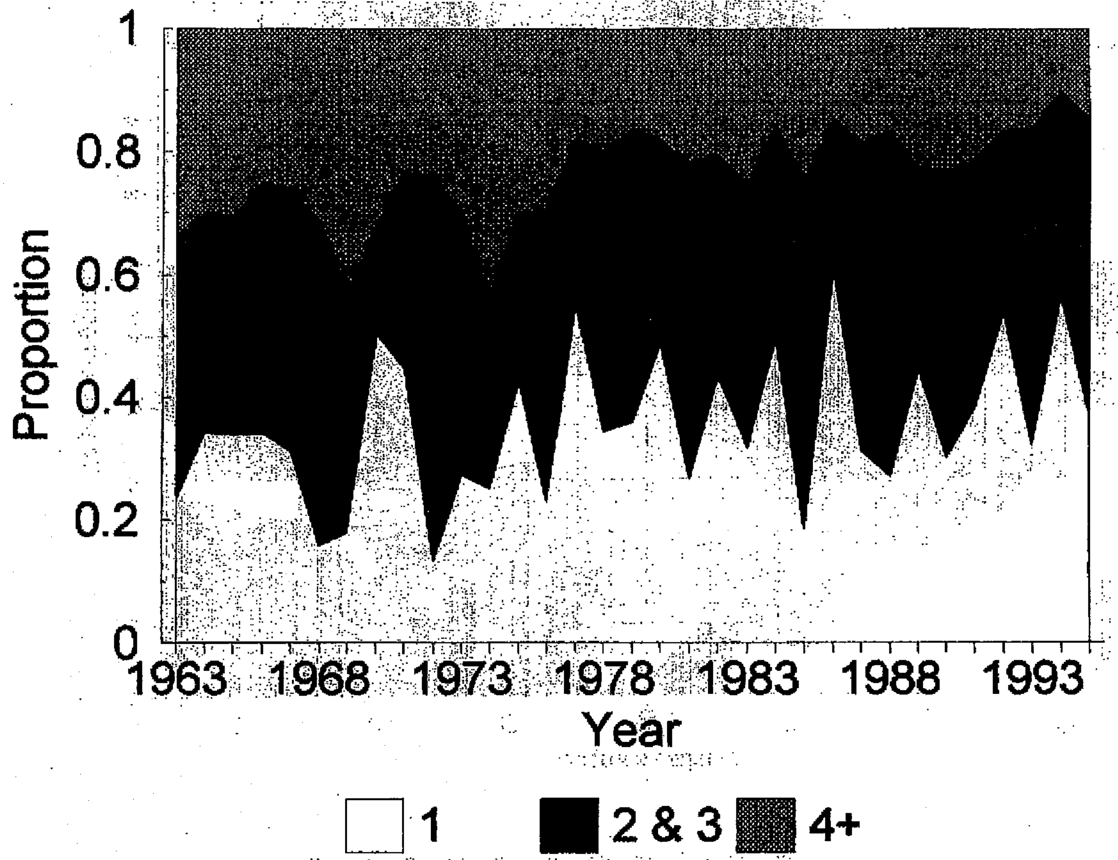


Figure 5.2.1.1.3.1. Distribution of cod sampled by the Scottish AGFS in 1982–1984 and 1992–1994.

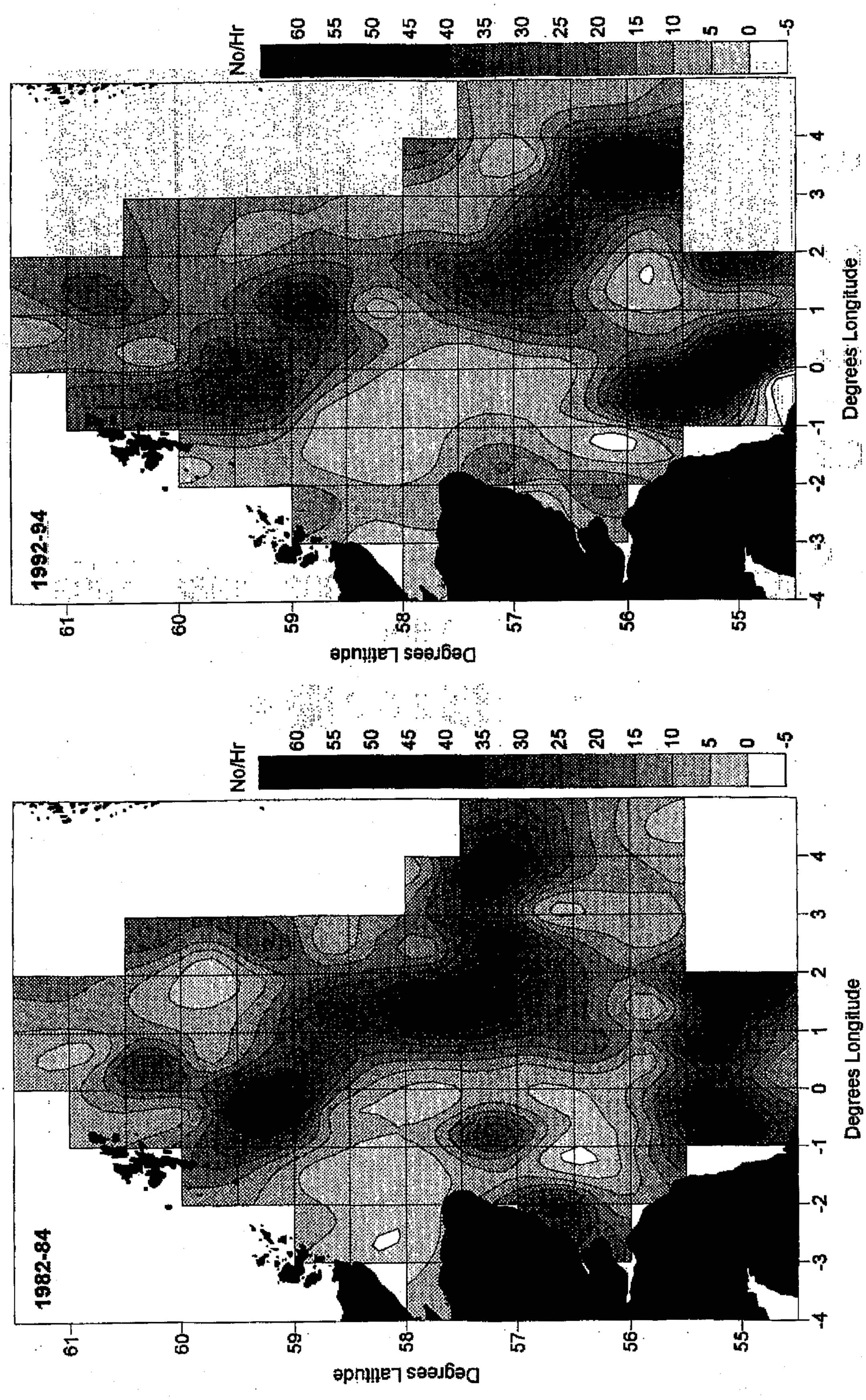


Figure 5.2.1.1.3.2. Chart showing the difference in abundance between the two time periods.

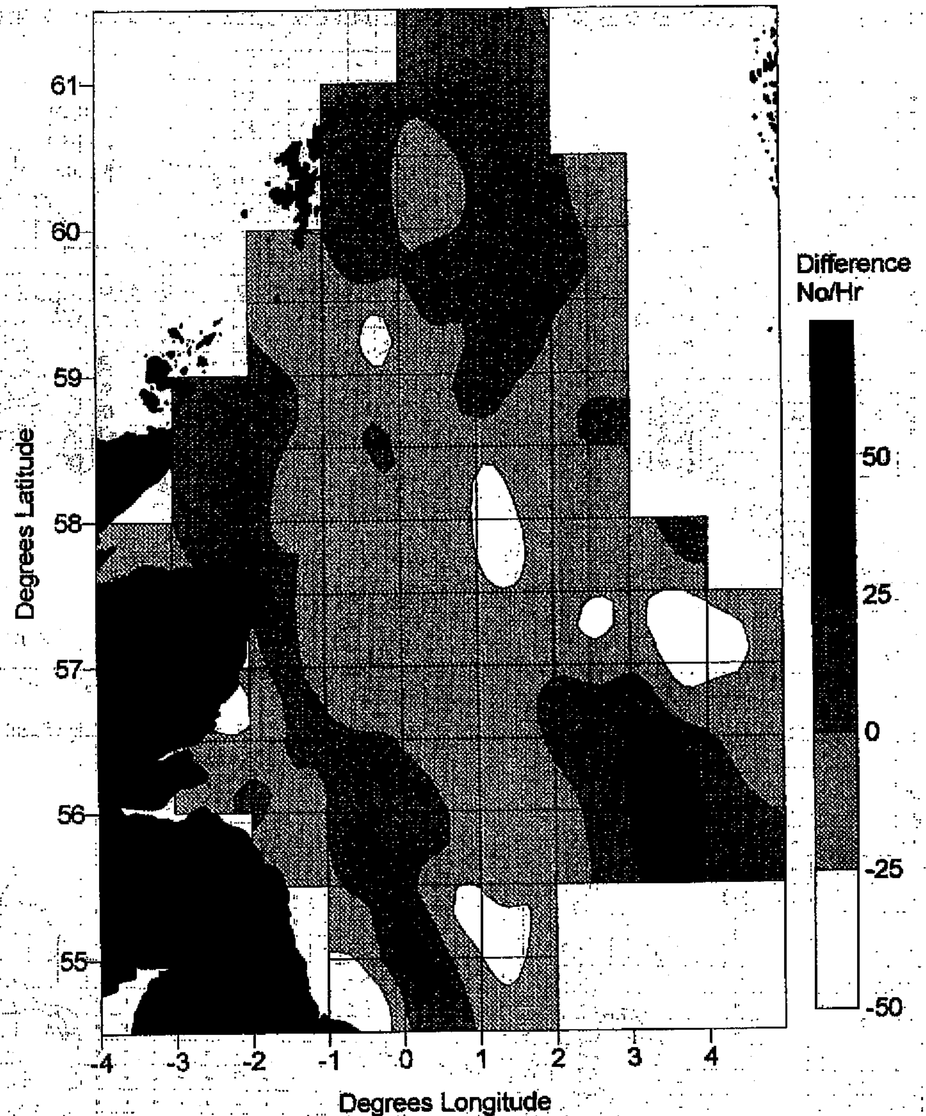


Figure 5.2.1.1.3.3. Average distribution of groups 1, 2, 3, and 4+ of cod for the period 1983–1987 (from Heesen, 1993).

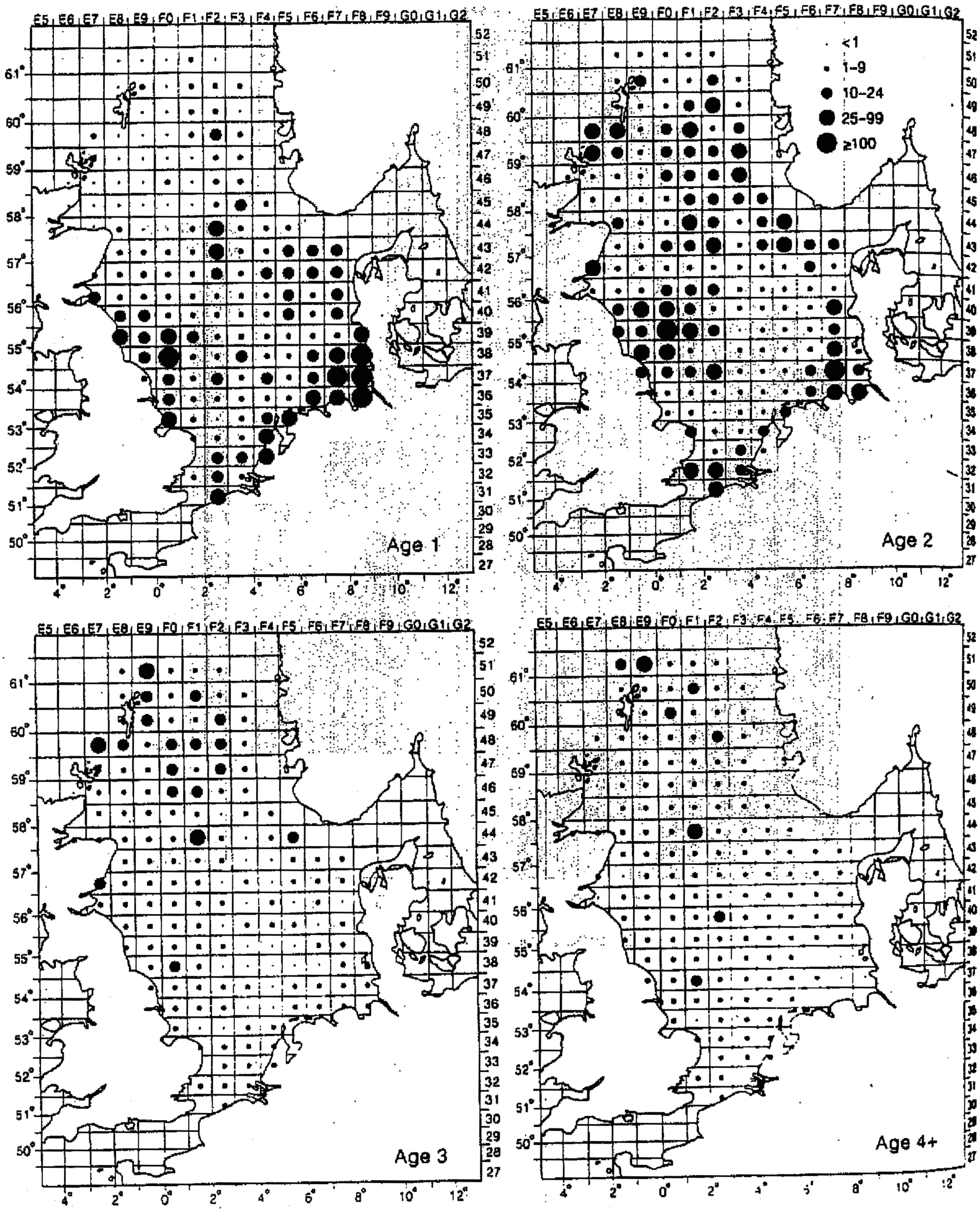


Figure 5.2.2.1.1.1. Trends in landings (——) and fishing mortality (– – – –) (top) and spawning-stock biomass (——) and recruitment (– – –) (bottom) for North Sea herring, 1947–1994. Landings include all North Sea autumn-spawning herring (data from ICES, 1996c).

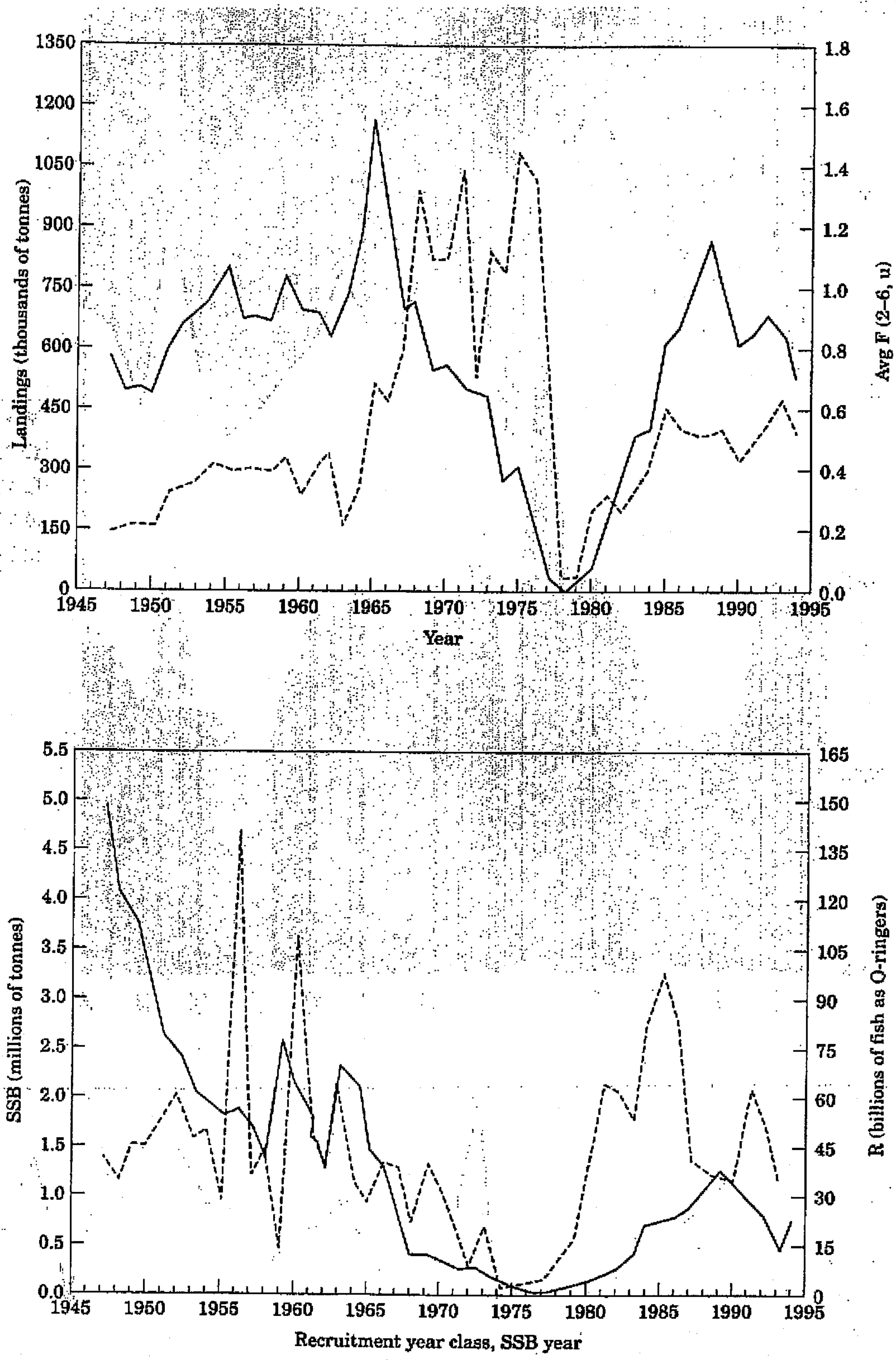


Figure 5.2.2.1.2.1. Changes over time in the proportion of herring at length in the IBTS (top) and trend in average size (bottom).

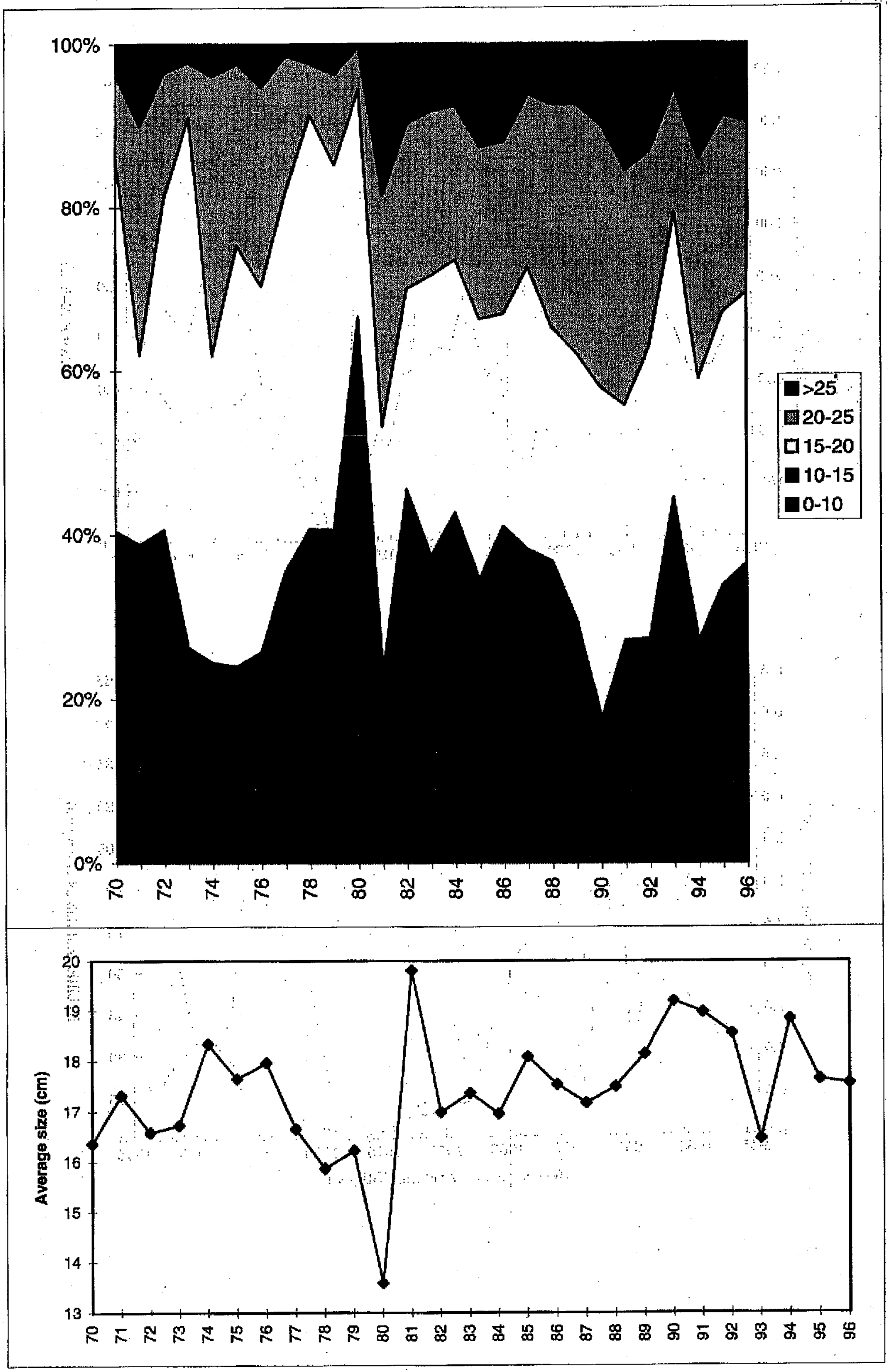


Figure 5.2.2.1.2.2. Variation in the proportion of herring at age over the duration of the VPA.

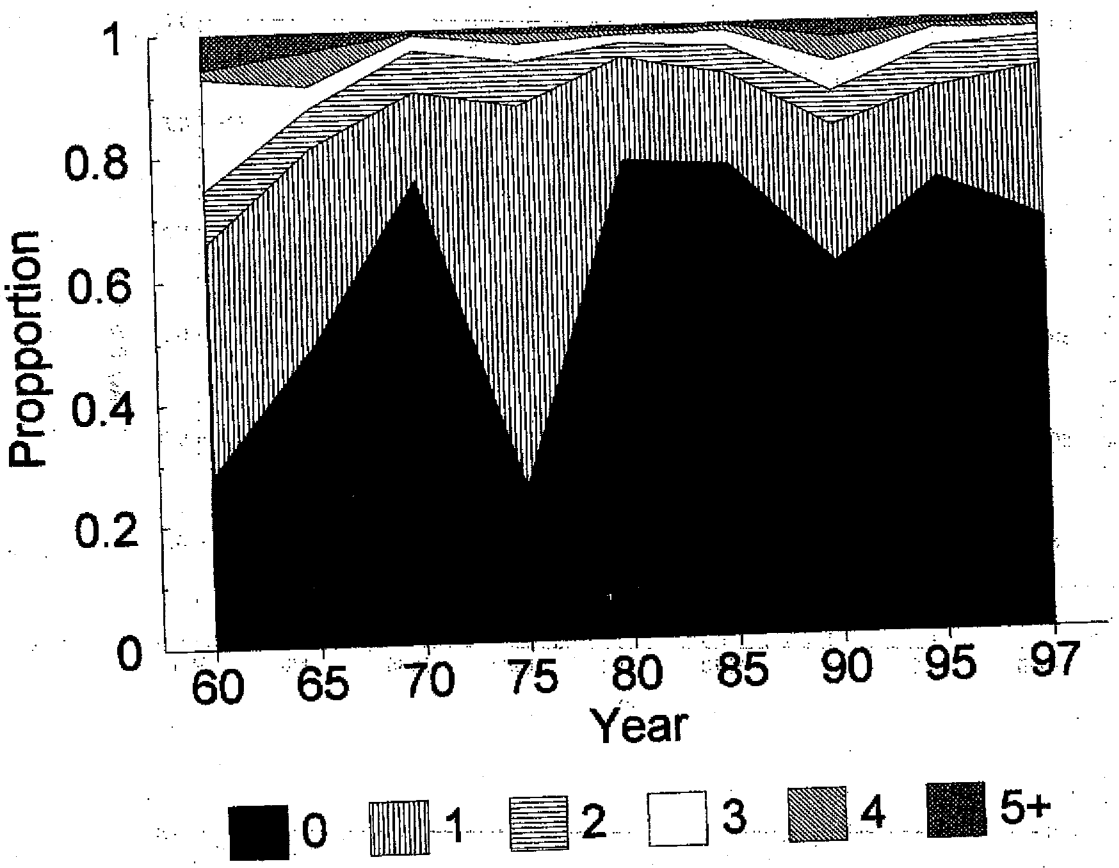


Figure 5.2.3.1.1.1. Trends in landings (——) and fishing mortality (– - –) (top) and spawning-stock biomass (——) and recruitment (– – –) (bottom) for North Sea sole, 1957–1994 (data from ICES, 1996c).

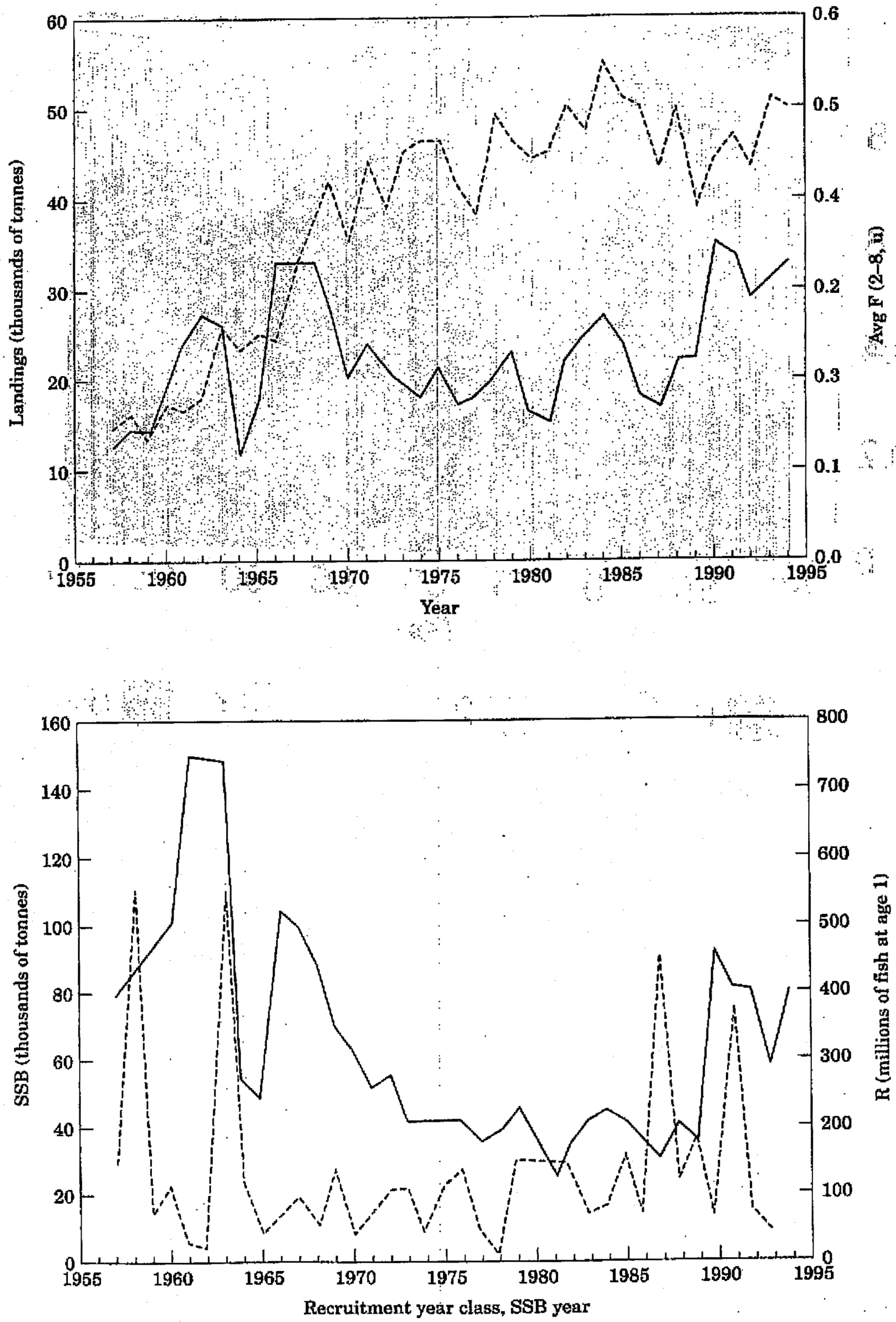
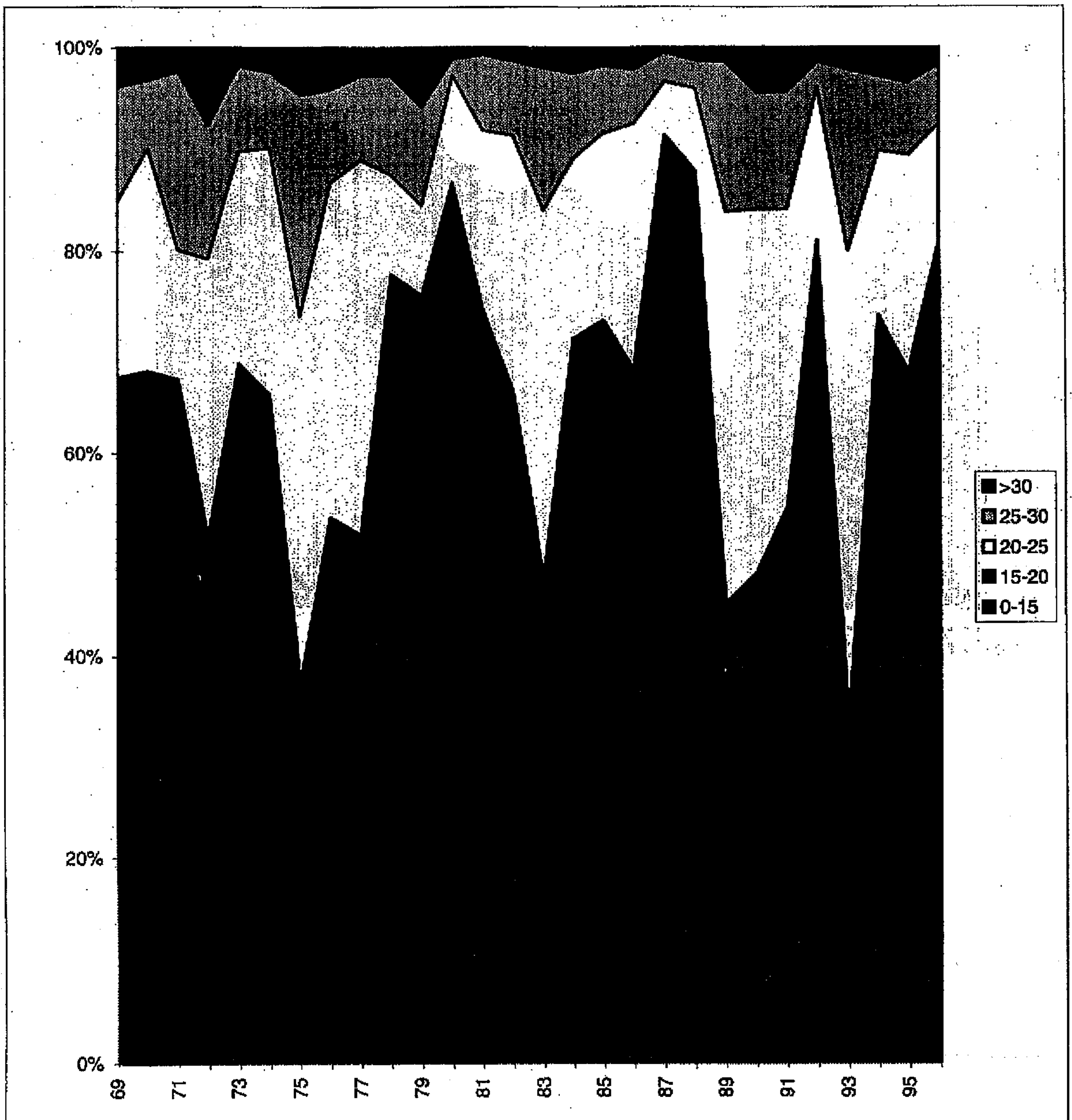


Figure 5.2.3.1.2.1. Changes over time in the proportion of sole at length in the IBTS (top) and trend in average size (bottom).



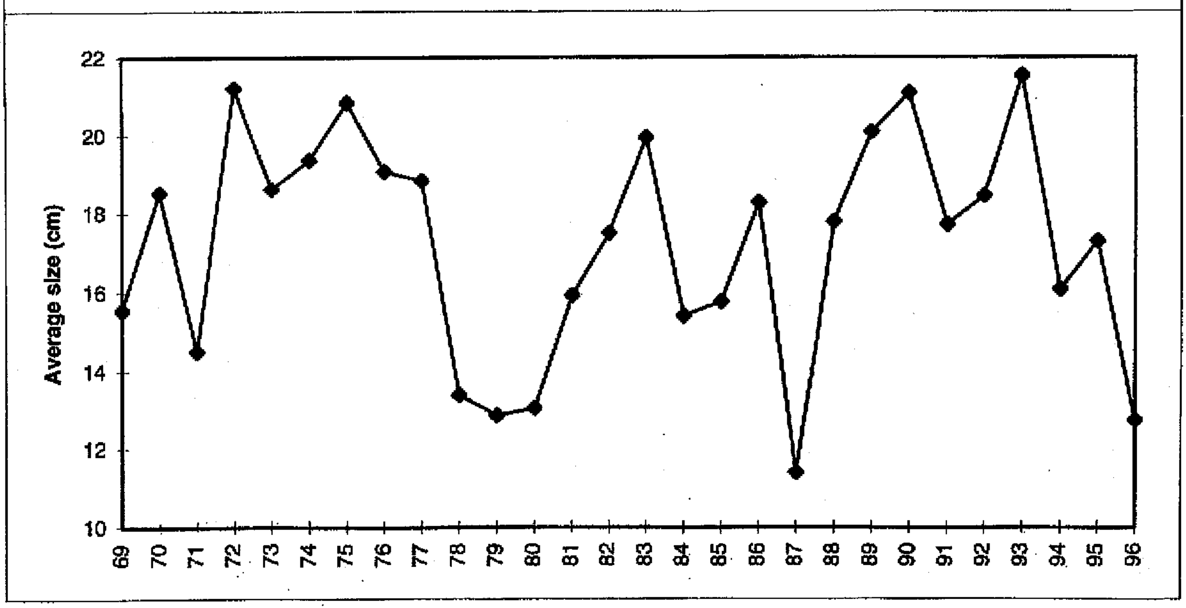


Figure 5.2.3.1.2.2. Variation in the proportion of herring at age over the duration of the VPA.

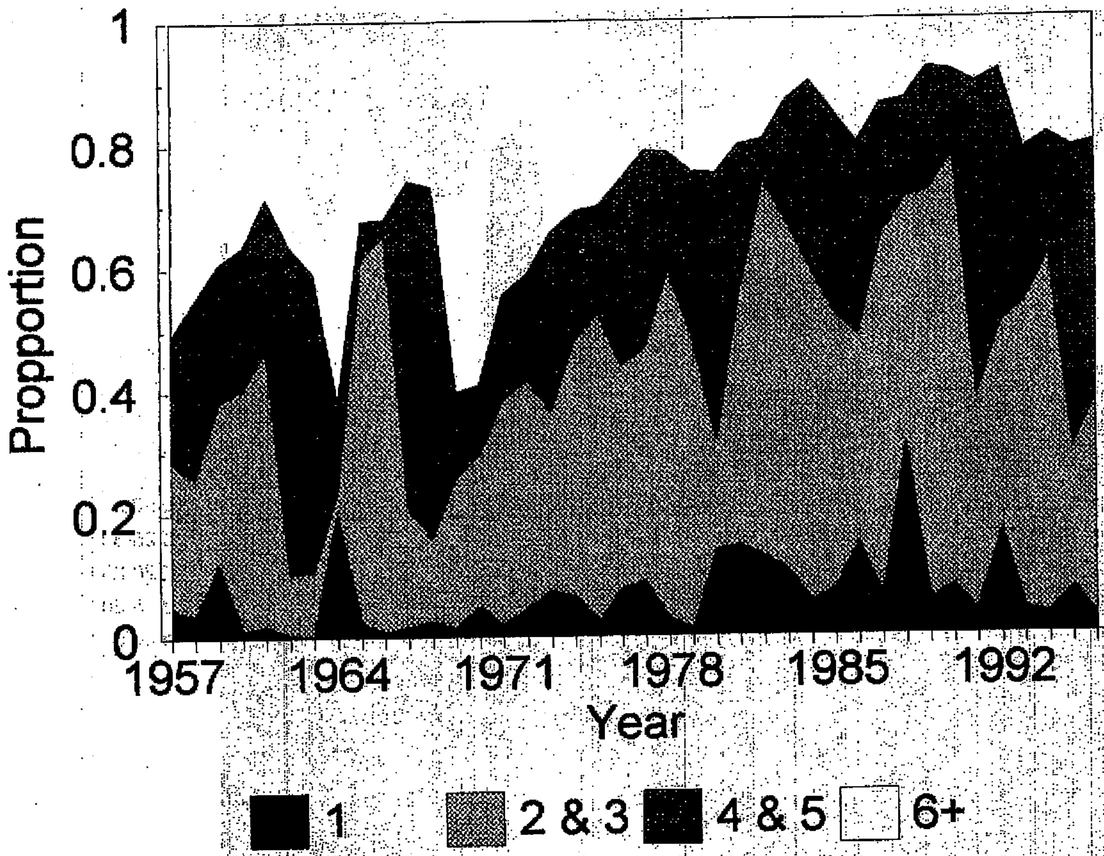


Figure 5.2.4.1.1.1. Trends in landings (——) and spawning-stock biomass (– – –) for North Sea mackerel, 1965–1990 (data from ICES, 1996c).

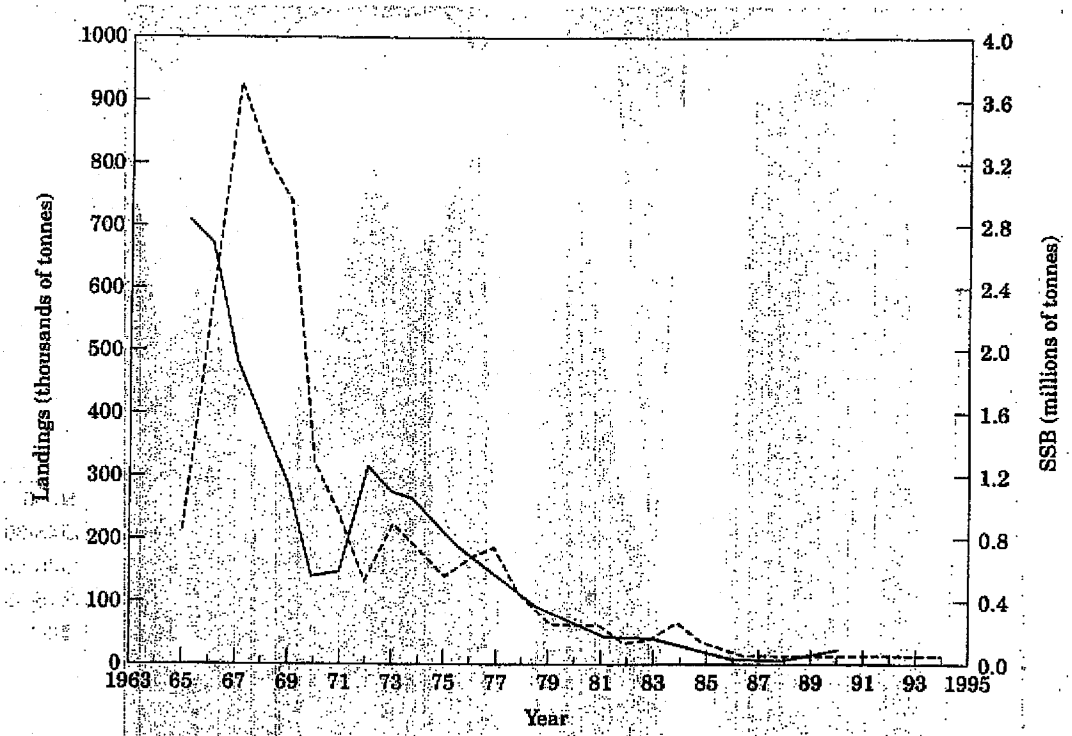


Figure 5.2.4.1.2.1. Changes over time in the proportion of mackerel at length in the IBTS (top) and trend in average size (bottom).

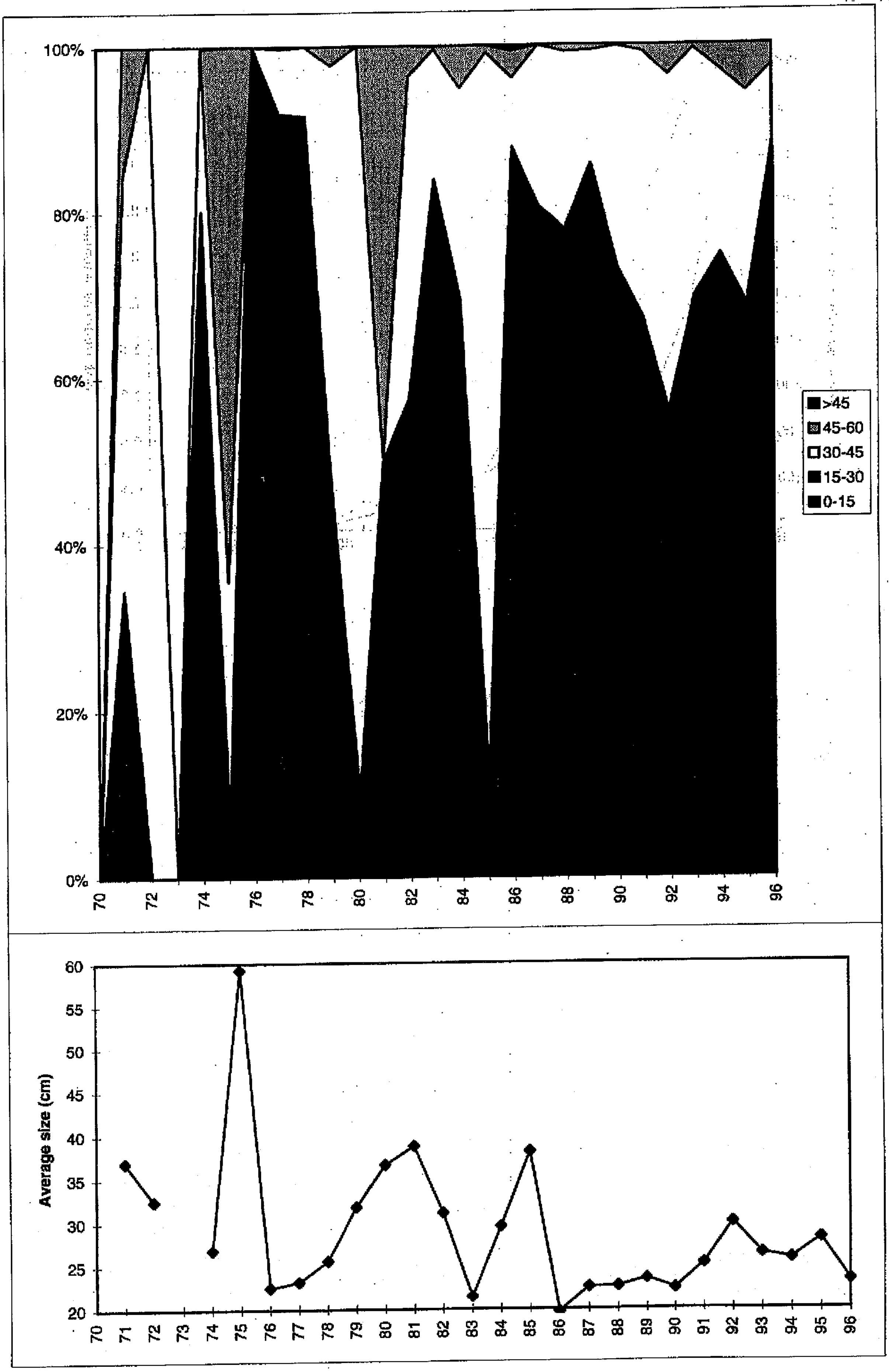
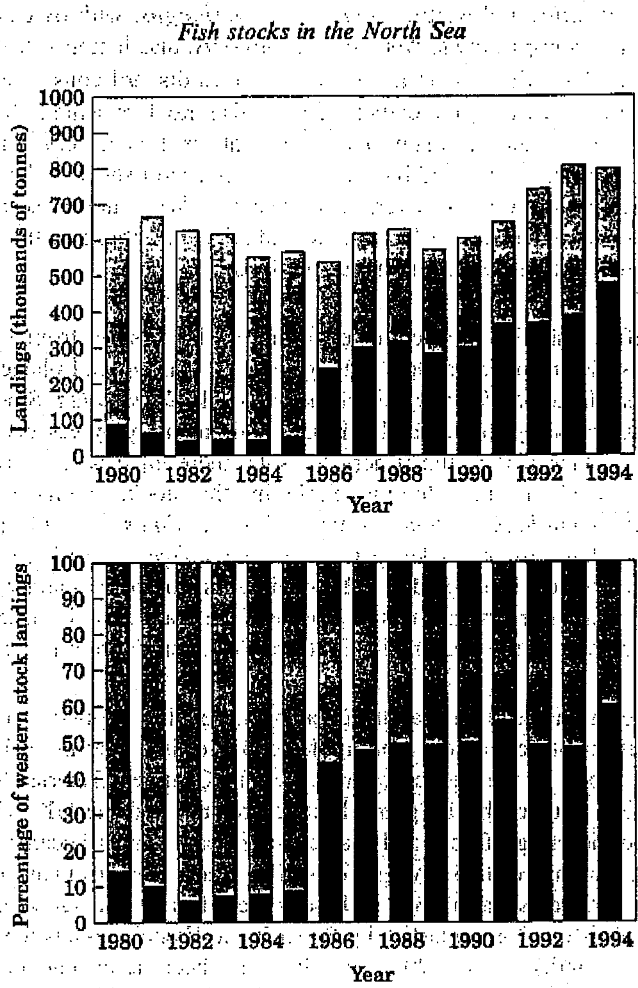


Figure 5.2.4.2.1.1. Total landings of Western mackerel (□) and landings of Western mackerel taken in the North Sea (■) (top), and percentage of total Western mackerel landings taken in the North Sea (■) (bottom), 1980–1994 (data from ICES, 1996c).



Total landings of Western mackerel (□) and landings of Western mackerel taken in the North Sea (■) (top), and percentage of total Western mackerel landings taken in the North Sea (■) (bottom), 1980–1994 (data from ICES, 1996c).

5.3 OSPAR Region III

Fisheries overview

The demersal fisheries in Division VIa are predominantly conducted by otter trawlers fishing for cod *Gadus morhua*, haddock *Melanogrammus aeglefinus*, and whiting *Merlangius merlangus*, with by-catches of saithe *Pollachius virens*, anglerfish *Lophius piscatorius*, megrim *Lepidorhombus whiffiagonis*, and lemon sole *Microstomus kitt*. These trawlers use mesh sizes of 80–100 mm depending on area and may at times discard considerable quantities of young haddock and whiting. The majority of these vessels are locally based Scottish trawlers using light trawls, but vessels from Ireland, Northern Ireland, England, France, and Germany also participate in this fishery. The pelagic fishery for herring is mainly operated by UK vessels in the north, and by Irish vessels in a roe fishery in the south. There is a directed fishery for blue whiting *Micromesistius poutassou*, mackerel *Scomber scombrus*, and horse mackerel *Trachurus trachurus* in the area. The industrial fisheries in Division VIa are much smaller than those in the North Sea and are based on the irregular Scottish sandeel fishery which peaked in the latter 1980s.

In the Irish Sea, the roundfish fisheries are conducted primarily by vessels from the bordering countries (UK and Ireland). The majority of vessels are otter-trawlers fishing for cod, whiting and plaice *Pleuronectes platessa*, with by-catches of haddock, anglerfish, hake *Merluccius merluccius*, and sole *Solea solea*. Since the early 1980s there has been a development of semi-pelagic trawling for cod and whiting, predominantly by vessels from Northern Ireland. Although some of the otter trawlers also take part in the fishery for sole, there have been a growing number of beam trawlers, particularly from southern England and from Belgium exploiting this stock. The most important by-catches of this fleet are plaice, rays *Raja* sp., brill *Scophthalmus rhombus*, turbot *Scophthalmus maximus*, and anglerfish. A fleet of vessels, primarily from Northern Ireland and Ireland, takes part in a targeted *Nephrops* fishery, and all boats take a considerable by-catch of whiting, most of which is discarded. These discards comprise mainly juveniles as the distribution of *Nephrops* coincides with the main nursery grounds of whiting. The main pelagic fishery in the Irish Sea is for herring *Clupea harengus*, although the size of the fleet, mainly pair trawlers from Northern Irleand, has declined in recent years.

Most of the demersal fisheries in this area have a mixed catch. Although it is possible to associate specific target species with particular fleets, various quantities of cod, whiting, hake, anglerfish, megrim, sole, plaice and *Nephrops* are taken together, depending on gear types. In the Celtic Sea and Western Channel, fisheries for demersal species, mainly cod, whiting, sole and plaice, are conducted by Belgium, France, Ireland, and the UK. The principal gears used are otter trawls and beam trawls. The targeting of sole and plaice using beam trawls became prevalent during the mid-1970s, leading to an increase in the landings of these two species. The gradual replacement of otter trawls by beam trawls has occurred in the Belgian and UK fleets. In the Bay of Biscay there has been a substantial increase in the coastal gillnet fishery targeting sole. A trawl fishery for anglerfish by Spanish and French vessels developed in the Celtic Sea and Bay of Biscay in the 1970s and expanded until 1990. The fishery has become dependent on small juvenile fish for which there is no minimum landing size. In addition, a gillnet fishery has developed in the Celtic Sea in the last decade.

Nephrops is an important component of the fisheries in this area. These fisheries developed in the 1970s and 1980s and effort increased continuously until recent years. Landings increased initially as effort increased, but they have tended to stabilize or decline at continuing high effort levels. The mesh size when fishing for *Nephrops* can lead to a significant by-catch of juvenile fish, notably hake.

There are separate trawl fisheries targeting herring in the Celtic Sea and mackerel and horse mackerel in the whole area. The herring fishery is principally a 'roe' fishery and discard rates have at times reached very high levels. There is also a small directed fishery for sprat *Sprattus sprattus* in the Channel.

5.3.1 Cod

5.3.1.1 Division VIa

5.3.1.1.1 Context - patterns of numbers, biomass, landings, and fishing mortality

Over the last 20 years, fishing effort in ICES Division VIa (West of Scotland) has generally increased in the Scottish light trawl fleet and *Nephrops* trawl fleet, although the latter has shown some reductions in very recent years, and effort has also declined for the Scottish seine fleet. In 1994, Scottish trawl effort declined to a particularly low level, but has since risen to levels of the late 1980s and early 1990s. In addition to Scotland, French trawlers have also been important in this area, and landed over 1,500 t in 1996. With estimates of misreporting since 1992 included in the assessment, the spawning biomass in 1996 is estimated to be 15,600 t, well below the long-term mean of the series (26,200 t). Mean fishing mortality in 1996 (0.86) is also well above the long-term average (0.77) and exceeds both F_{max} (0.274) and F_{med} (0.586) (Figure 5.3.1.1.1.1) (ICES, 1998a).

5.3.1.1.2 Changes in size distribution/age composition

Highest levels of F are on 3- and 4-year old fish (1.03 and 1.07, respectively) and on all ages over 2, F is above 0.84. For Division VIa cod, the strong 1996 year class is thought to contribute ca. 40 % to landings in 1998, and 35 % to SSB (Figure 5.3.1.1.2.1). For this stock the maturity ogive suggests that only 52 % of 2-year old fish, and 86 % of 3-year old fish are considered to be mature.

5.3.1.1.3 Changes in spatial distribution

Cod is a species which generally shows a northern Arctic/boreal distribution, and is abundant in the northern part of this OSPAR region. The majority of landings are from the coastal waters of west coast of Scotland close to the Hebrides, and beyond the western edge of the continental shelf, and shelf waters of the west of Ireland. The 1996 landings from Division VIa were 9331 t and from Division VIb were 327 t. There is no information to suggest that the distribution of cod in this region has been altered as a direct result of commercial fishing activity (ICES, 1998a).

5.3.1.2 Division VIIa

5.3.1.2.1 Context - patterns of numbers, biomass, landings and fishing mortality

The catch per unit effort (CPUE) of the spring Northern Ireland fishery exploited by otter and pelagic trawl fleets in the western Irish Sea has declined continuously since 1986, although there was a sharp increase in CPUE of this fleet in 1996. The England and Wales otter trawl CPUE is still at a level of approximately half that of 1989. Fishing effort in all three main fleets has shown a marked decline since 1989, and as a result landings have declined from a peak of 14,000 t in 1988 to present levels of 48,000 t, (Figure 5.3.1.2.1.1). The SSB reached its lowest point in 1995 and, due to the poor state of this stock, the TAC was reduced substantially from 11,000 t in 1993 to 5800 t in 1995. The current estimate of F for this stock (0.58) exceeds F_{max} but lies below F_{med}, and sensitivity analysis suggests that the probability of the spawning biomass falling below B_{loss} in 1999 is negligible. B_{loss} is defined as the biomass where models suggest that the ability of the stock to recover is jeopardised. Nevertheless, cod are taken in a mixed fishery with haddock, whiting and plaice, and the implications of increased effort directed on the large haddock year class entering the Irish Sea fishery should be considered (ICES, 1998a).

5.3.1.2.2 Changes in size distribution/age composition

Landings of cod are predominantly of 2- to 4-year old fish, as landings of 1-year old cod have declined since 1991 and are the second lowest of the time series, and landings of 5-year old fish have also been the poorest on record. The 1995 and 1996 year classes (2- and 3-year olds) contribute 65 % to the 1998 landings (Figure 5.3.1.2.2.1). Maturity at age for this stock has been revised as a result of recent studies and data used in the assessments now indicate that 38 % of 2-year old fish, and 100 % of 3-year old fish are mature (ICES, 1998a).

5.3.1.2.3 Changes in spatial distribution

Cod are found throughout the Irish Sea but occur in the greatest abundance in the coastal and offshore waters of Ireland and in Liverpool Bay (Figure 5.3.1.2.3.1.a). There is insufficient evidence to confirm that the commercial exploitation of cod in this region has altered the spatial distribution of the species, as results from the beam trawl survey show generally comparable distributions in western waters between 1990 and 1996 (Figure 5.3.1.2.3.1.a,b).

5.3.1.3 Celtic Sea stocks

5.3.1.3.1 Context - patterns of numbers, biomass, landings and fishing mortality

Western Channel cod are now assessed with stocks in the Celtic Sea, so this description includes cod from Divisions VIIe, VIIf, VIIg, VIIh, VIIj, and VIIk. Only Divisions VIIe and VIIk are considered to lie outside OSPAR Region III.

The very strong 1986 year class resulted in high landings in 1988–1990, but since 1991, landings have returned to the levels recorded in the period 1981–1987, and for 1996 were 11,900 t for the entire region. The majority of these landings (80 %) were taken by France, but England, Wales, Belgium and Ireland also landed significant quantities of fish from this fishery. The spawning stock biomass of Celtic Sea cod reached a peak of 24,000 t in 1989 and subsequently decreased sharply to 6600 t in 1992 due to high fishing mortality and poor recruitment (Figure 5.3.1.3.1.1). With recruitment of the relatively good 1990 and 1991 year classes, SSB increased to 13,700 t in 1994 and 15,000 t in 1996, which is currently above the mean of the longest time series. After reaching the highest value of fishing mortality of over 1.0 in 1991, F decreased slightly until 1995 (F = 0.73) and is estimated at 0.75 in 1996, which is above the mean, and slightly below F_{med} (ICES, 1998b).

5.3.1.3.2 Changes in size distribution/age composition

Cod is a fast growing and early maturing fish, and the predicted SSB in these assessments is heavily dependent on the assumed values of recruitment for the incoming year classes, and landings from this OSPAR region are increasingly dependent on strong year classes entering the fishery. Fishing mortalities are high and, at such levels, the contribution of good year classes to the SSB is very transitory.

5.3.1.3.3 Changes in spatial distribution

Cod is found throughout the region but at a reduced abundance relative to the unexploited stock. There are insufficient data on cod in this region to identify changes in the spatial distribution of the species resulting from commercial exploitation.

5.3.2 Herring

5.3.2.1 Division VIa (North)

5.3.2.1.1 Context - patterns of numbers, biomass, landings, and fishing mortality

Continued difficulties with catch reporting exist for this stock and misreporting is thought to be approximately 68 % of the total catch (ICES, 1997). The problem is particularly acute during the peak months of the herring fishery around Shetland (August to October). Acoustic surveys have been used to estimate SSB at 370,000 t, which is lower than values derived in 1993, which was an exceptionally high stock estimate thought to have been affected by an influx of populations from other regions. Assessment of this stock suggests that it is lightly exploited, with little risk of a stock decline at current levels of exploitation.

5.3.2.1.2 Changes in size distribution/age composition

Current estimates of stock size are still influenced by poor sampling of weight at age and additional weight information needs to be collected from offshore regions for future assessments. Until more complete data are available, it is unclear how the continued exploitation of this stock has affected the age composition.

5.3.2.1.3 Changes in spatial distribution

The herring fishery in the northern part of Division VIa takes place in two main areas: certain vessels fish inshore for small, younger herring, while other vessels fish offshore in deeper waters where the fish are larger and older. The distribution of herring in quarter 4 in this region and other parts of OSPAR Region III is shown in Figure 5.3.2.1.3.1. There are no data which can be used to identify how the spatial distribution of the species has altered as a result of commercial exploitation.

5.3.2.2 Clyde herring

5.3.2.2.1 Context - patterns of numbers, biomass, landings, and fishing mortality

Management of this stock is complicated by the presence of two virtually indistinguishable stocks: a resident spring-spawning population and the immigrant autumn-spawning component from Division VIa. In recent years, management has been directed at rebuilding the highly depleted spring-spawning component to historical levels, using closed areas and seasons. Historically this spring-spawning stock supported a fishery with catches of up to 15,000 t per year in the 1960s and landings generally began to decline through the 1970s and 1980s until, at present, the landings have fluctuated at below 1000 t. As there are no fishery independent surveys and no stock separation of the catches, nothing is known about the current state of the spring-spawning stock (ICES, 1997).

5.3.2.2.2 Changes in size distribution/age composition

No data are available which identify changes in size or age composition as a result of exploitation.

5.3.2.2.3 Changes in spatial distribution

Spatial changes in the distribution of the autumn-spawning component can be explained by environmental factors affecting the distribution of migrating species, but there are no data to suggest that exploitation of the spring-spawning component has affected the spatial distribution. Herring spawn on coarse sand and gravel sediments and the location of these substrates influences spawning distribution.

5.3.2.3 Divisions VIa (South) VIIb, and VIIc

5.3.2.3.1 Context - patterns of numbers, biomass, landings, and fishing mortality

There have been no recent analytical assessments of this stock, but there is some evidence that the stock has declined in recent years and is now at a comparatively low level. There has been no substantial recruitment to the stock in recent years and the very strong 1985 year class has now passed through the fishery (ICES, 1997).

5.3.2.3.2 Changes in size distribution/age composition

No data are available which identify changes in size or age composition as a result of exploitation.

5.3.2.3.3 Changes in spatial distribution

The scarcity of herring in this region may be due to a combination of a decline in the stock, accentuated by a more northerly distribution of the stock in recent years resulting from environmental factors.

5.3.3 Sole

5.3.3.1 Division VIIa

5.3.3.1.1 Context - patterns of numbers, biomass, landings, and fishing mortality

Sole is at the northern limit of its distribution in the Irish Sea and the major fisheries for the species are in the Irish Sea, the Bristol Channel and Celtic Sea, where it is mainly taken in a beam trawl fishery with plaice as a by-catch. In the Irish Sea, sole spawning stock biomass is currently (spawning time in 1996) at its historically lowest level and is only 60 % of the average observed over the period 1970–1996 (Figure 5.3.3.1.1.1). Since the good 1989 year class, there have been five consecutive below-average recruitments. The prediction for spawning stock biomass in 1997 is 17 % below B_{loss}, and sensitivity analysis would suggest that the probability of spawning stock biomass remaining below B_{loss} in 1999 is about 45 %. Whilst this stock is considered to be outside safe biological limits, the population contains a broader distribution of year classes than cod, for example, and would be expected to decline slowly. Recent studies of sole maturity have suggested that for western sole stocks a maturity rate of about 70 % applies to 3-year old fish, and fish are considered fully mature (98 %) at age 5 (ICES, 1998a).

5.3.3.1.2 Changes in size distribution/age composition

Although no specific studies are known of changes in the age composition of sole in this Division, data from Working Group Assessments show that the 1994 and 1995 year classes (2- and 3-year olds) will form ca. 20 % of the catch in 1997 and ca. 50 % of the catch in 1998. This confirms that the age structure of Irish Sea sole is dominated by young fish and the fishery is heavily dependent on them (Figure 5.3.3.1.2.1).

5.3.3.1.3 Changes in spatial distribution

Flatfish abundance data from the Irish Sea have been collected since 1998 by the UK as part of a programme to monitor variation in recruitment of sole stocks on the southern and western coasts of England. The International Bottom Trawl Surveys collates these data with those of other surveys in the North Sea, and data for 1990 and 1996 are shown in Figure 5.3.3.1.3.1 (ICES, 1994; Rogers *et al.*, Working Paper 19). Although the spatial distribution of the surveys varies slightly during the seven-year period, most notably with the inclusion of the central North Sea in recent years, the centres of peak abundance of sole in Liverpool Bay and Solway Firth remain constant. Data collected from a longer time period would be required to compare the distribution of sole at a time when fishing activity was lower, but the requirements of this species for specific fine and productive sand/mud sediments suggest that changes in distribution would be limited and are governed by environmental factors (Symonds and Rogers, 1995).

5.3.3.2 Celtic Sea

5.3.3.2.1 Context - patterns of numbers, biomass, landings, and fishing mortality

Sole stocks in Divisions VIIf and VIIg (Bristol Channel) are assessed by ICES. Total international landings were 994 t in 1996, and the largest proportion of these landings are taken by Belgian beam trawlers. Assessments have shown that fishing mortality has increased from around 0.29 in the 1970s to a peak of 0.65 in 1990, and F is currently at 0.48 (Figure 5.3.3.2.1.1). This value of F is 41 % above F_{med} and 2 % above F_{high}. At the current F there is a greater than 50 % probability that SSB will fall below Bloss in 1999, and that a reduction of 25 % in fishing mortality is required in order to ensure that SSB remains above Bloss in 1999. A slight decline in F during the early 1990s has been explained by the greater time spent by beam trawl fleets in other fishing areas. Recruitment has fluctuated without any trend, however, and the 1989 year class was outstanding and equivalent only to that of 1970. Sole is taken mainly in a beam trawl fishery, with plaice as a by-catch, and to a lesser extent in the otter trawl fisheries, so management advice needs to take into account measures proposed for plaice (ICES, 1998b).

5.3.3.2.2 Changes in size distribution/age composition

Although no specific studies are known of changes in the age composition of plaice in the Celtic Sea, data from Working Group Assessments suggest that young fish are becoming an increasingly larger proportion of the landings from this fishery.

5.3.3.2.3 Changes in spatial distribution

This region is close to the Irish Sea and many of the comments in Section 5.3.3.1.3, above, also apply to the Celtic Sea stock. For those parts that have been surveyed, which represent the main centres of abundance of sole in the Celtic Sea, the population is largely within the Bristol Channel, where it occupies sheltered substrates in the bays of the area (Figure 5.3.3.1.3.1). An analysis of the spatial distribution of sole age groups in the Bristol Channel by Symonds and Rogers (1995) showed that juveniles (0–2 group fish) remained generally within the shallow 0–20 m depth contours, but recruit fish become more widely dispersed in the outer Bristol Channel and the Celtic Sea. This species has particular requirements for nursery, feeding, and spawning grounds, which are governed by local hydrography and substrate types.

5.3.4 References

ICES. 1994. Report of the Study Group on Beam Trawl Surveys in the North Sea and Eastern Channel. ICES CM 1990/G:59.

ICES. 1997. Report of the Herring Assessment Working Group for the Area South of 62 N. ICES CM 1997/Assess:8.

ICES. 1998a. Report of the Working Group on the Assessment of Northern Shelf Demersal Stocks. ICES CM 1998/Assess:1.

ICES. 1998b. Report of the Working Group on the Assessment of Southern Shelf Demersal Stocks. ICES CM 1998/Assess:4.

Symonds, D.J., and Rogers, S.I. 1995. The influence of spawning and nursery grounds on the distribution of sole *Solea solea* L. in the Irish Sea, Bristol Channel and other areas. Journal of Experimental Marine Biology and Ecology, 190: 243–261.

Figure 5.3.1.1.1.1. Trends in landings and recruitment (top), fishing mortality and spawning stock biomass (bottom) for cod in Division VIa from 1966–1996.

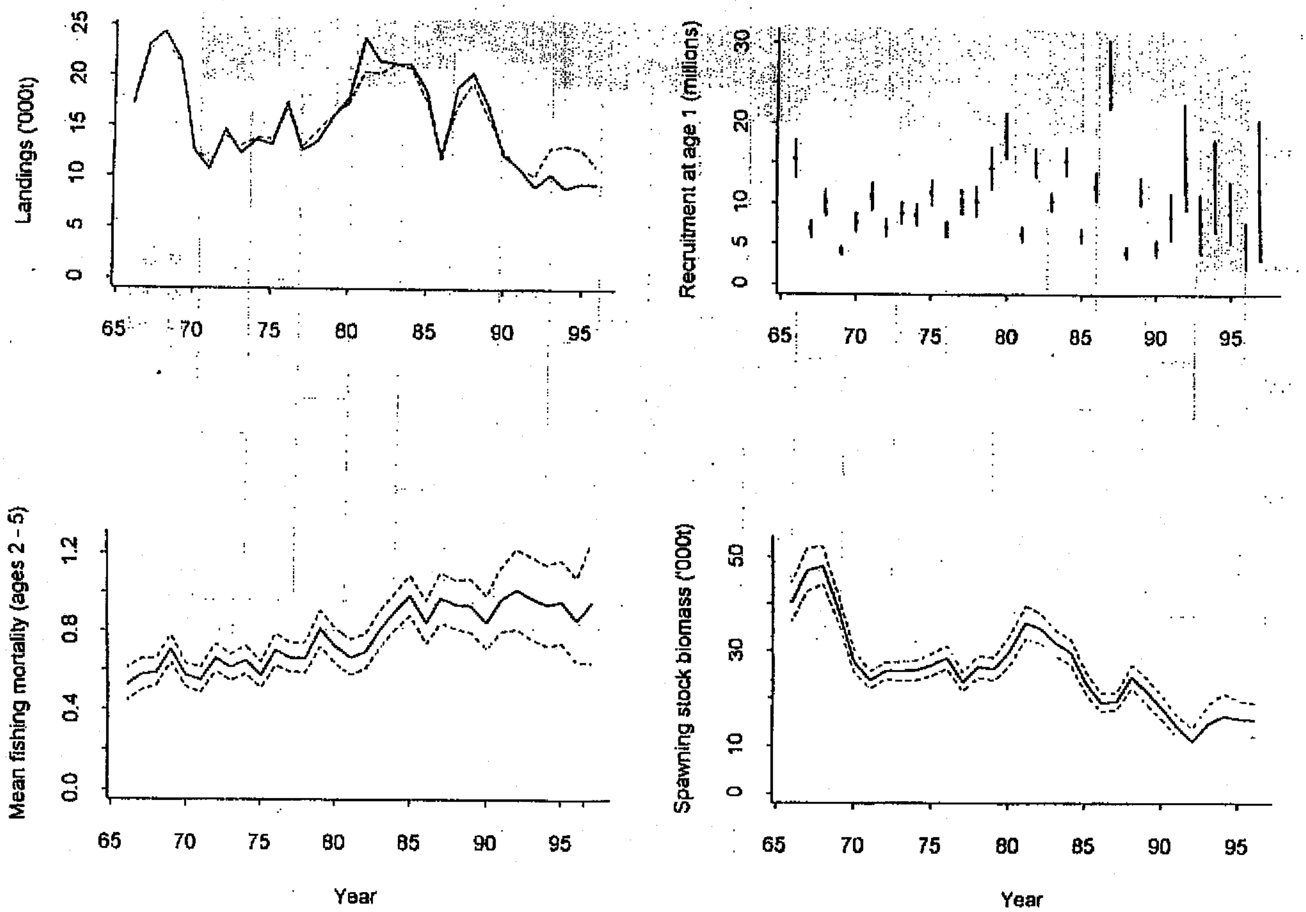


Figure 5.3.1.1.2.1. Cod in Division VIa - age population (years) composition by number from 1966–1985.

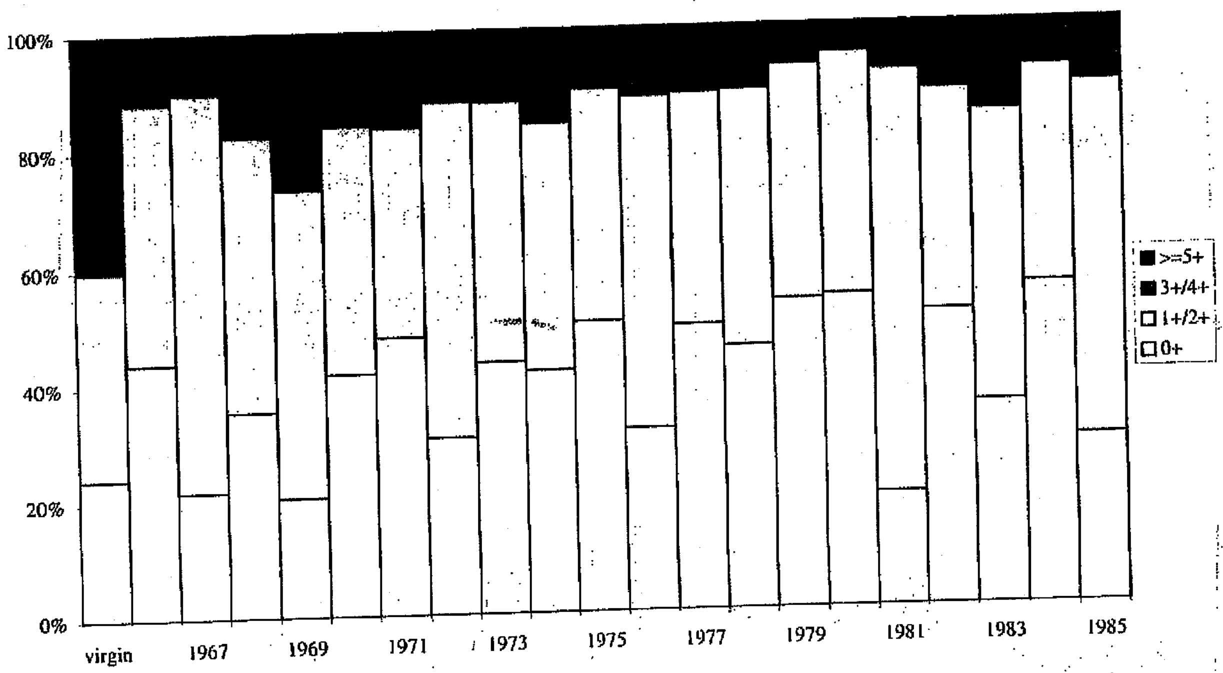


Figure 5.3.1.2.1.1. Trends in landings and recruitment (top), fishing mortality and spawning stock biomass (bottom) for cod in the Irish Sea (Division VIIa) from 1969–1996.

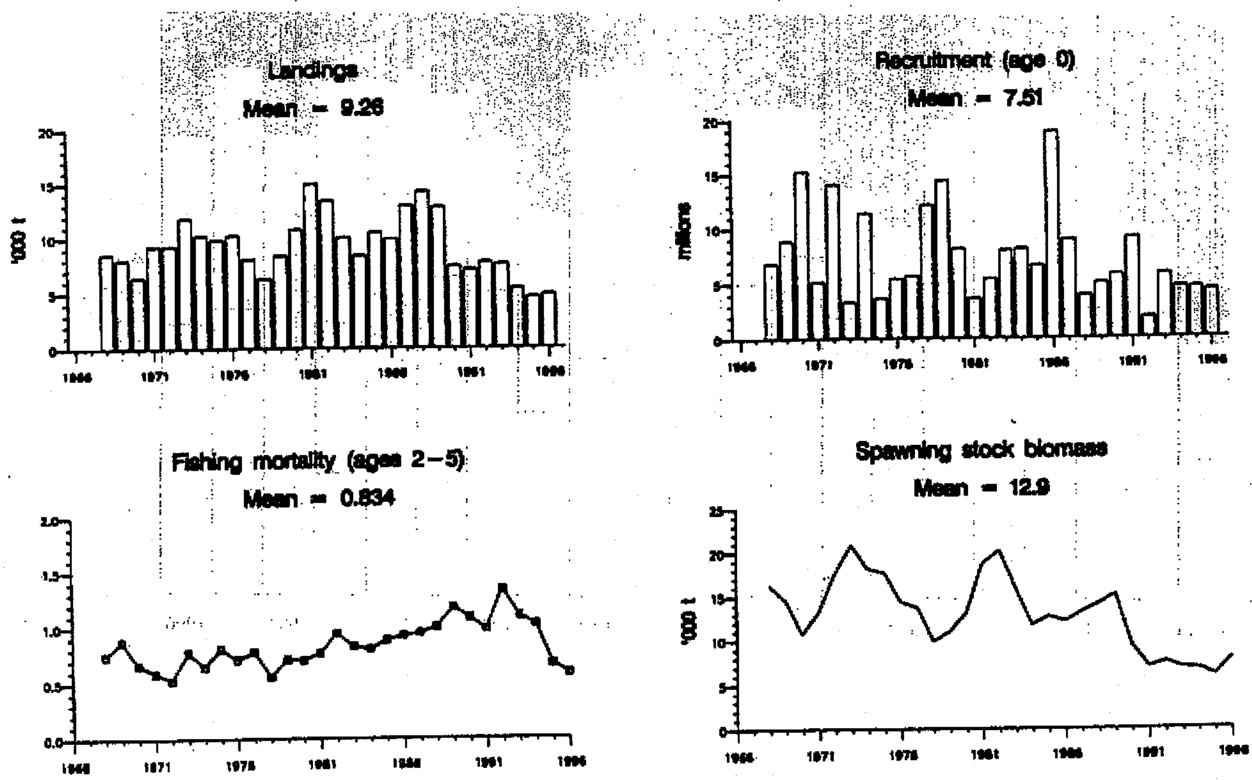


Figure 5.3.1.2.2.1. Cod in the Irish Sea - population age (years) composition by number from 1977–1996.

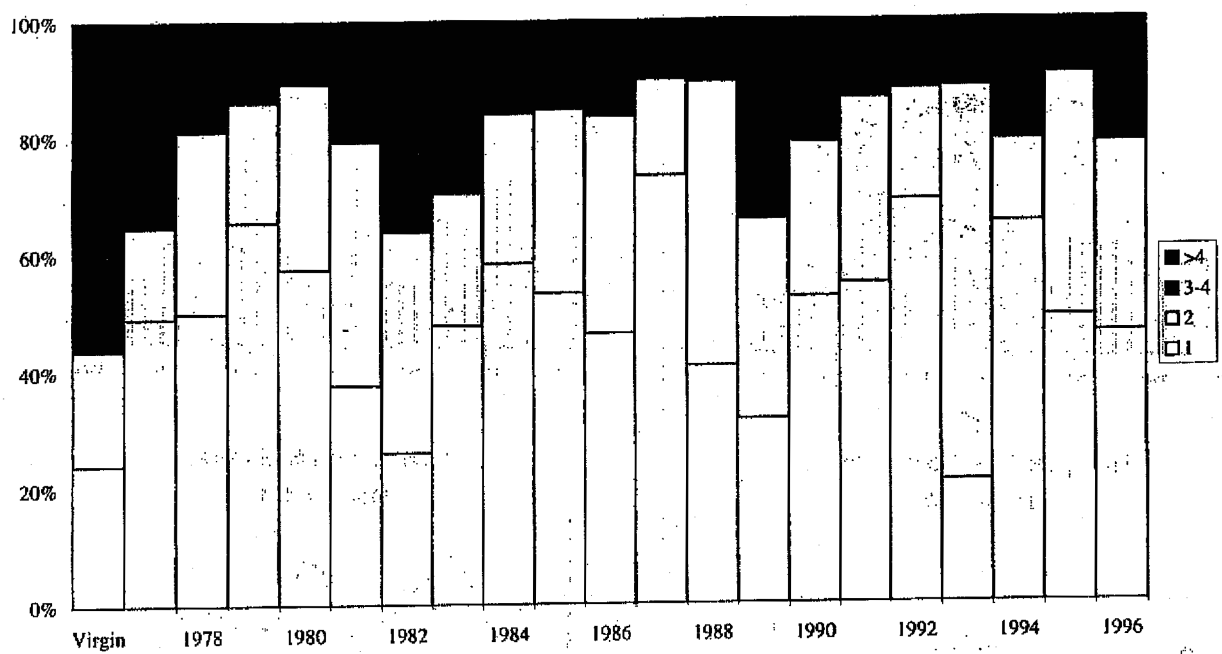


Figure 5.3.1.2.3.1.a. Distribution of cod, as determined by the International Beam Trawl Surveys, in the third quarter of 1990.

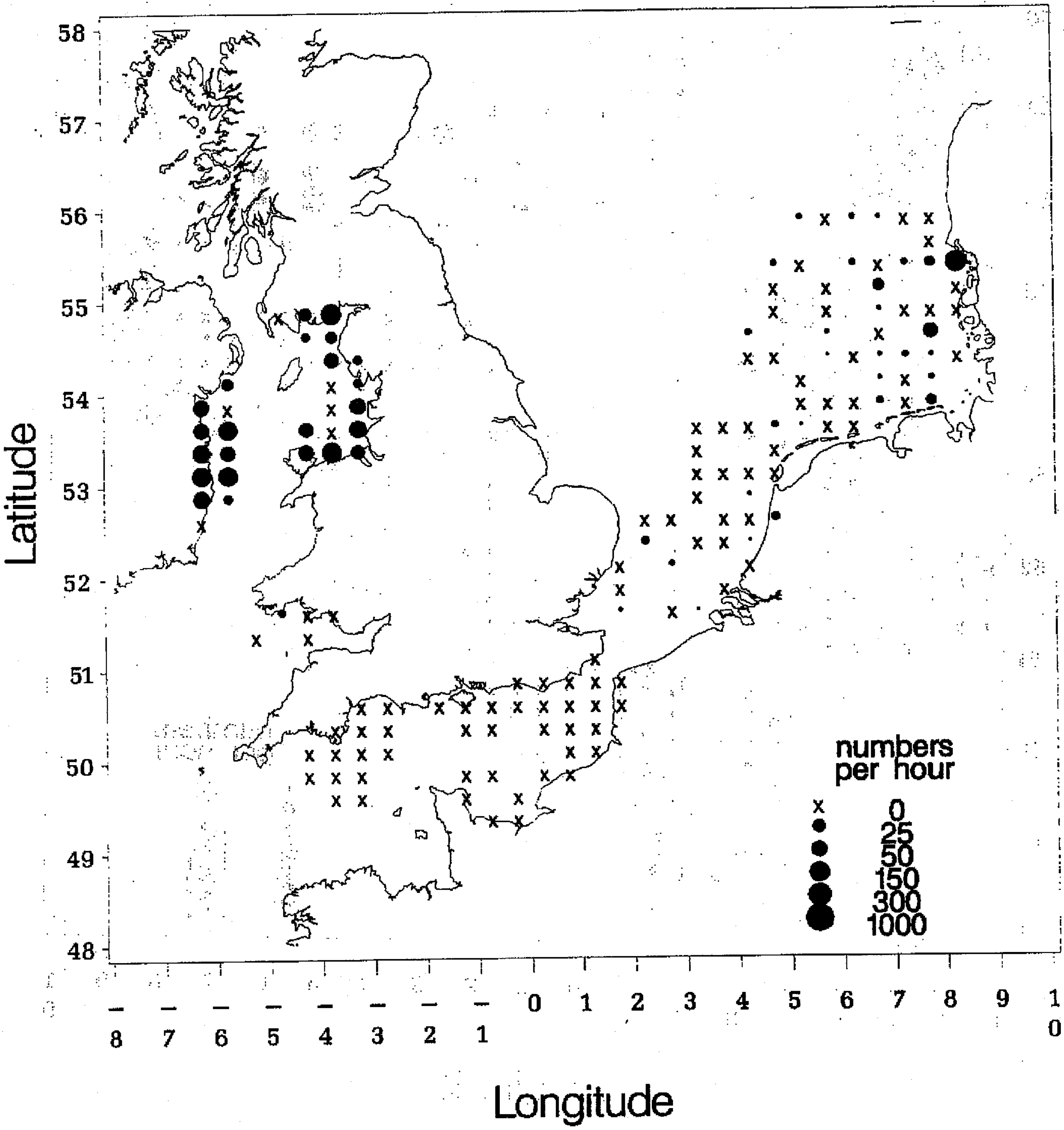


Figure 5.3.1.2.3.1.b. Distribution of cod, as determined by the International Beam Trawl Surveys, in the third quarter of 1996.

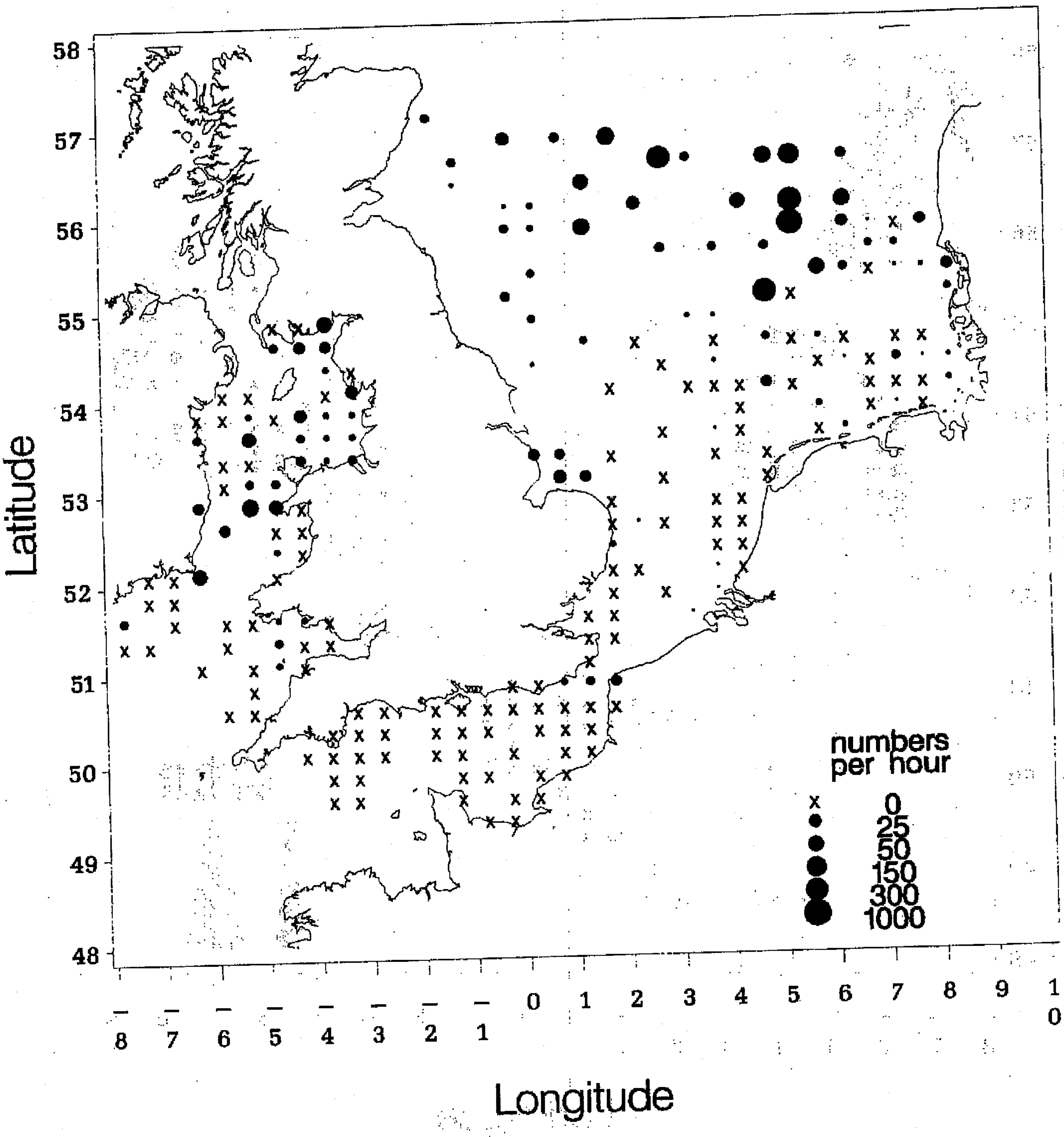


Figure 5.3.1.3.1.1. Trends in landings and recruitment (top), fishing mortality and spawning stock biomass (bottom) for Celtic Sea and Western Channel cod from 1971–1995.

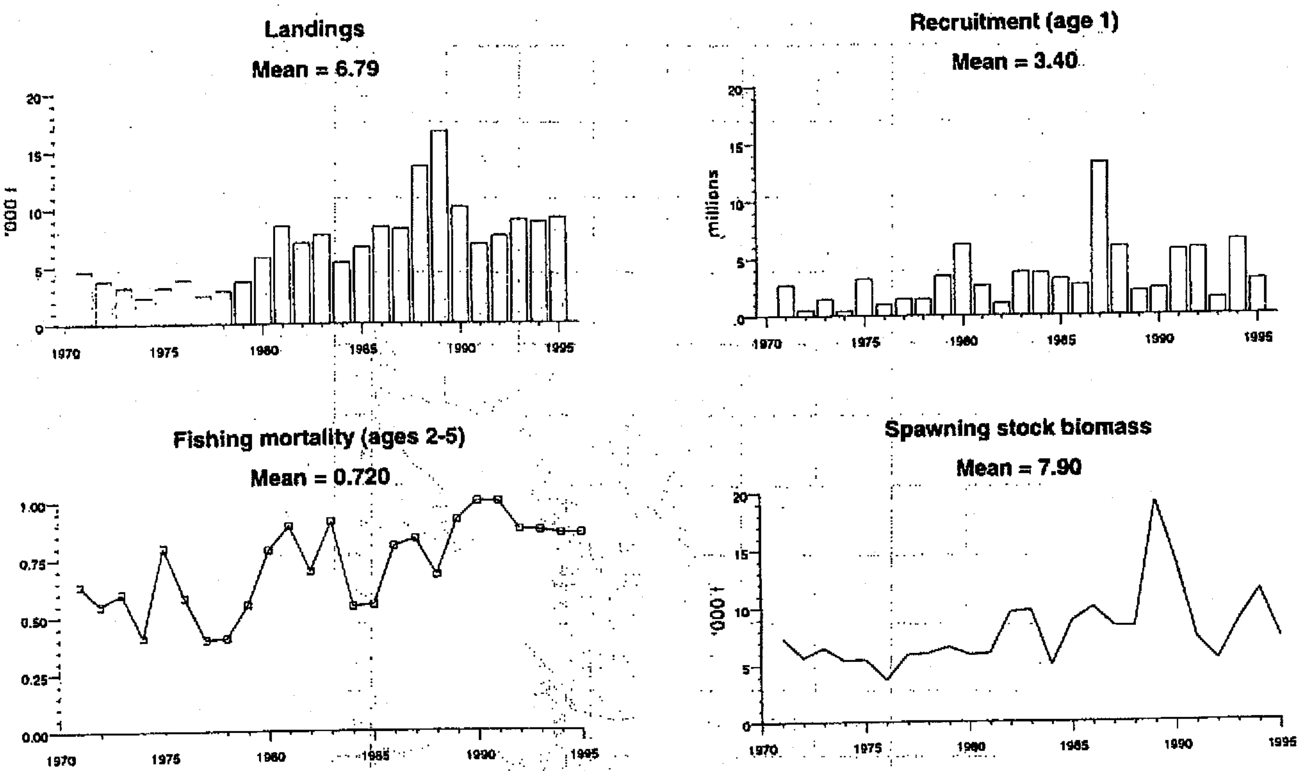


Figure 5.3.2.1.3.1. Distribution of herring in Division VIIa (North) in the fourth quarter of 1996.

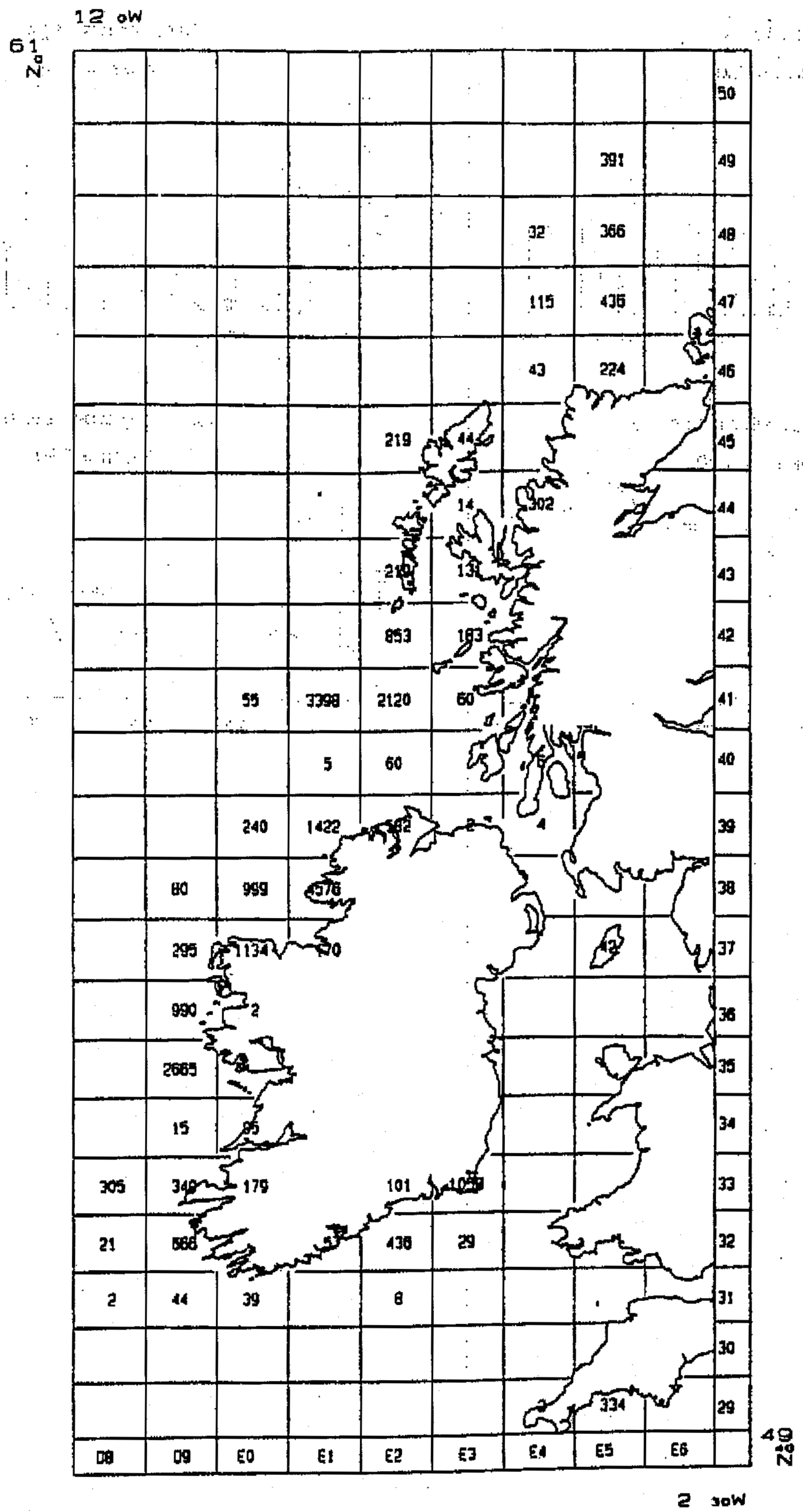


Figure 5.3.3.1.1.1. Trends in landings and recruitment (top), fishing mortality and spawning stock biomass (bottom) for sole in the Irish Sea (Divsion VIIa) from 1966–1996.

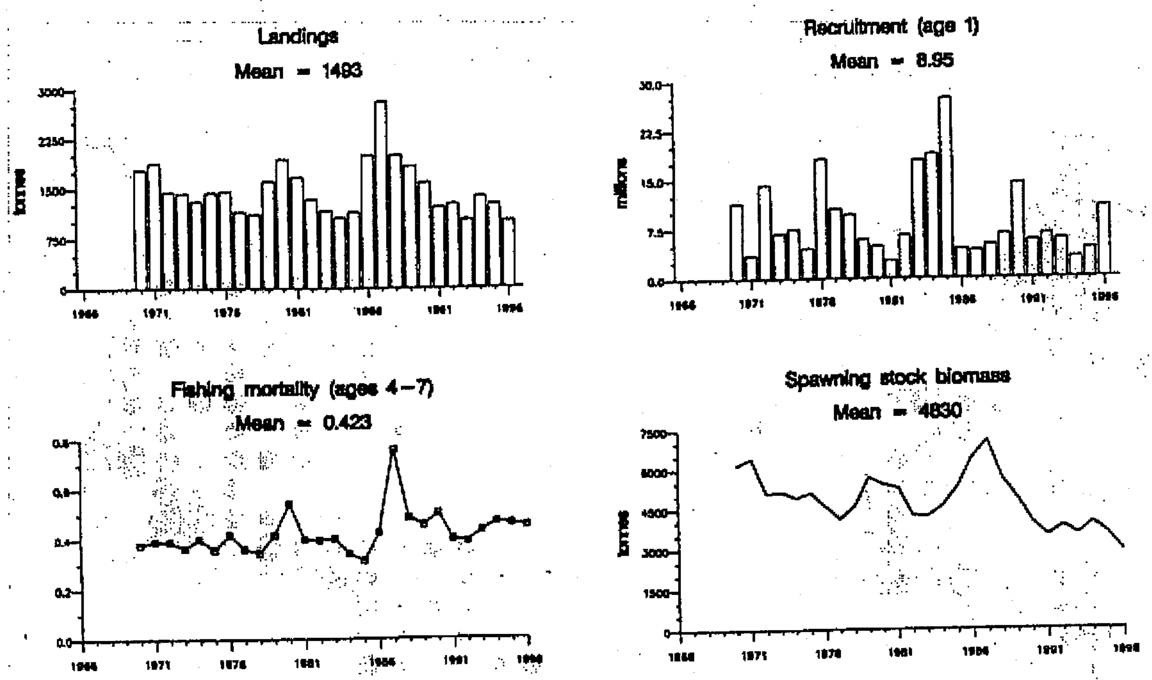


Figure 5.3.3.1.2.1. Sole in the Irish Sea (Division VIIa) - population age (years) composition from 1977–1996.

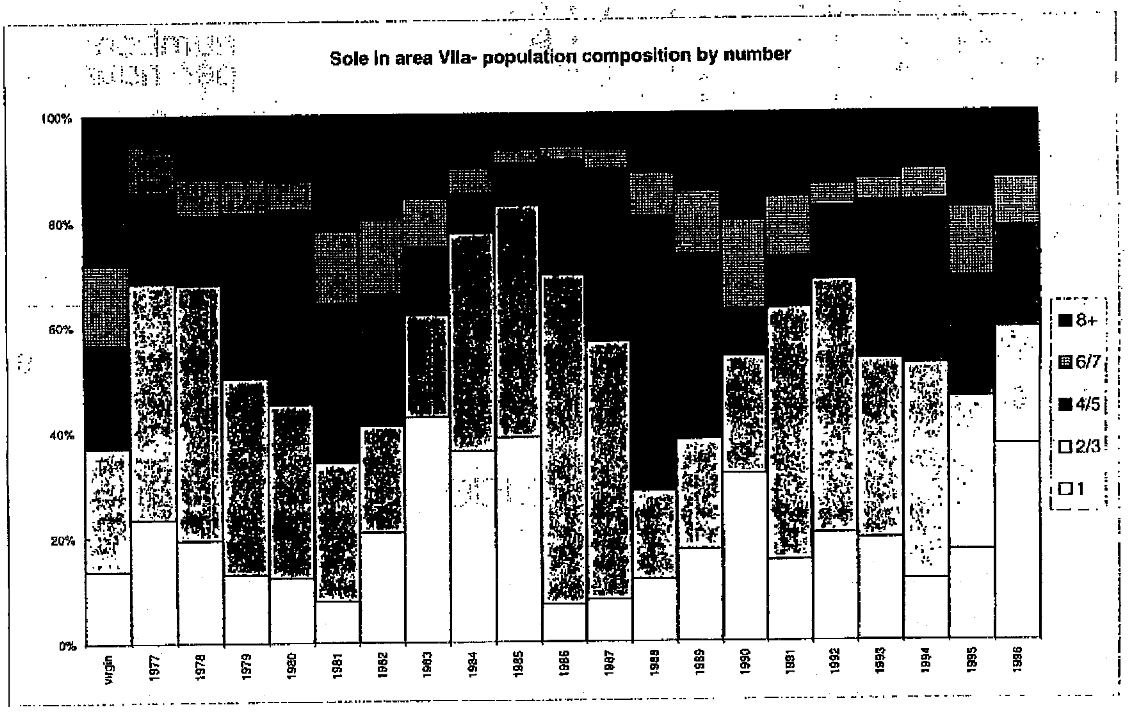


Figure 5.3.3.1.3.a. Distribution of sole, as determined by the International Beam Trawl Surveys, in the third quarter of 1990.

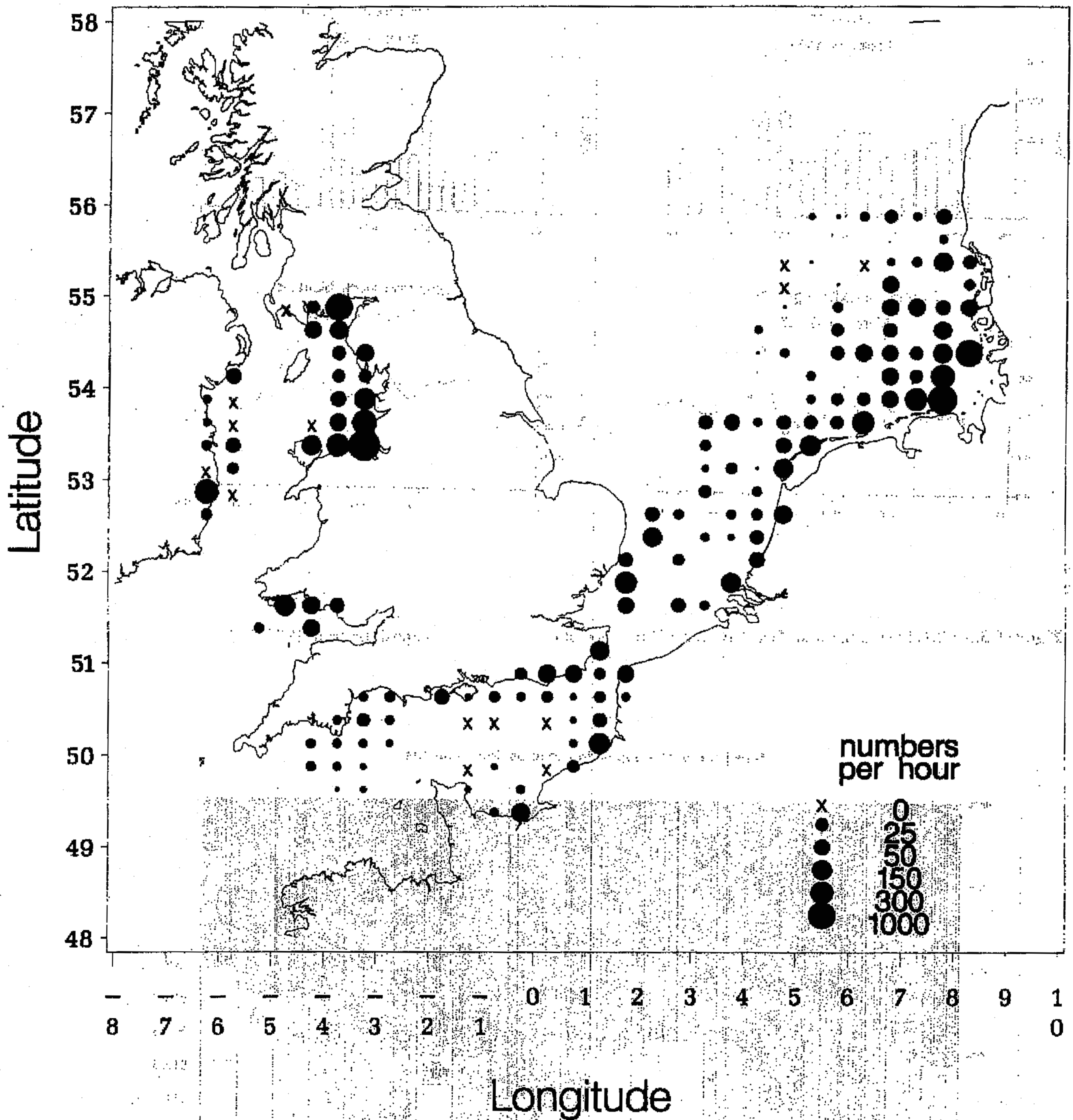


Figure 5.3.3.1.3.b. Distribution of sole, as determined by the International Beam Trawl Surveys, in the third quarter of 1996.

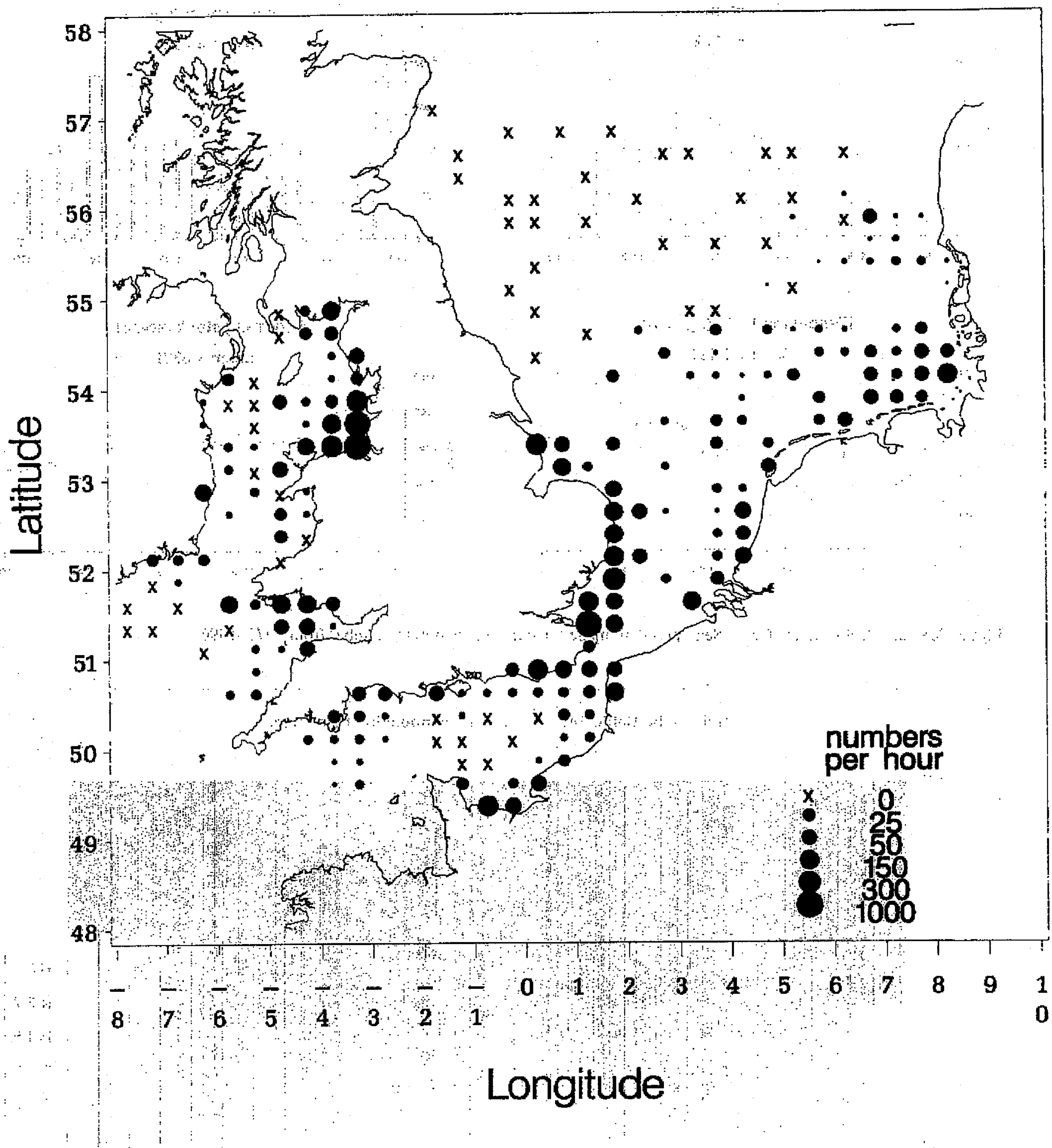


Figure 5.3.3.2.1.1. Trends in landings and recruitment (top), fishing mortality and spawning stock biomass (bottom) for sole in the Celtic Sea from 1971–1995.

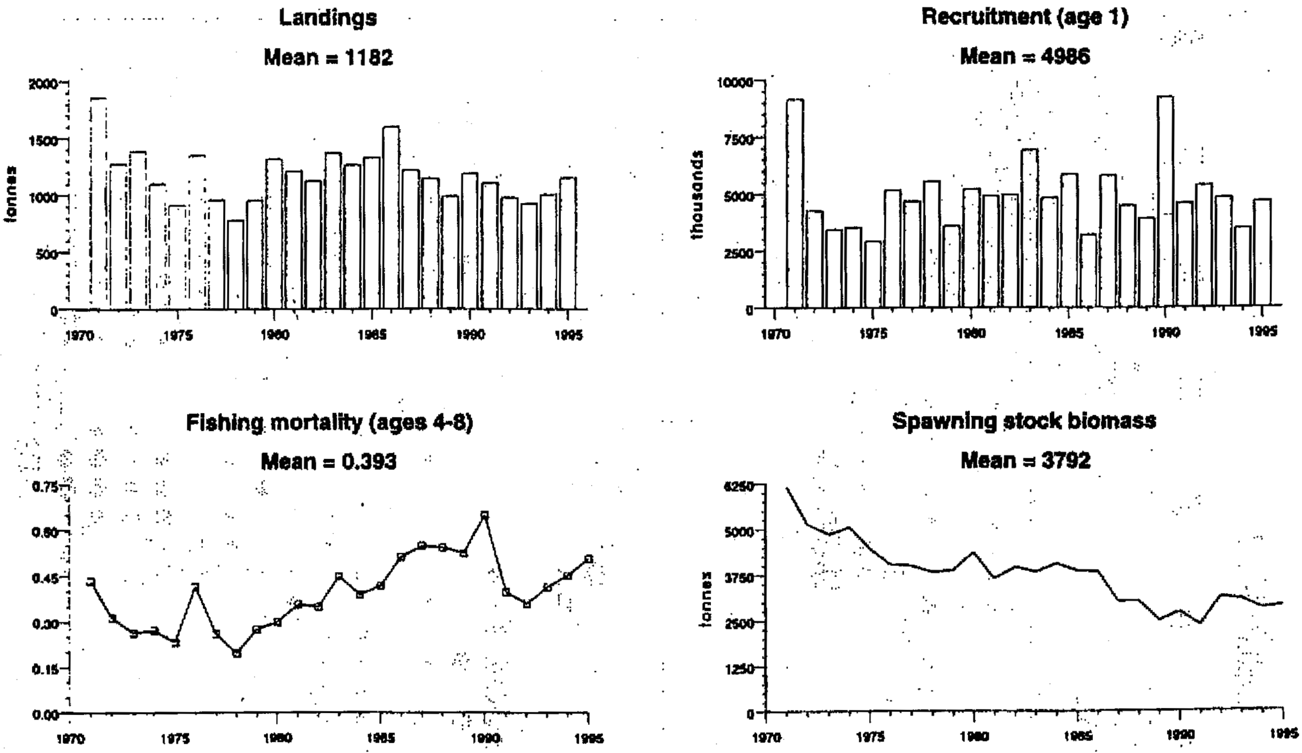
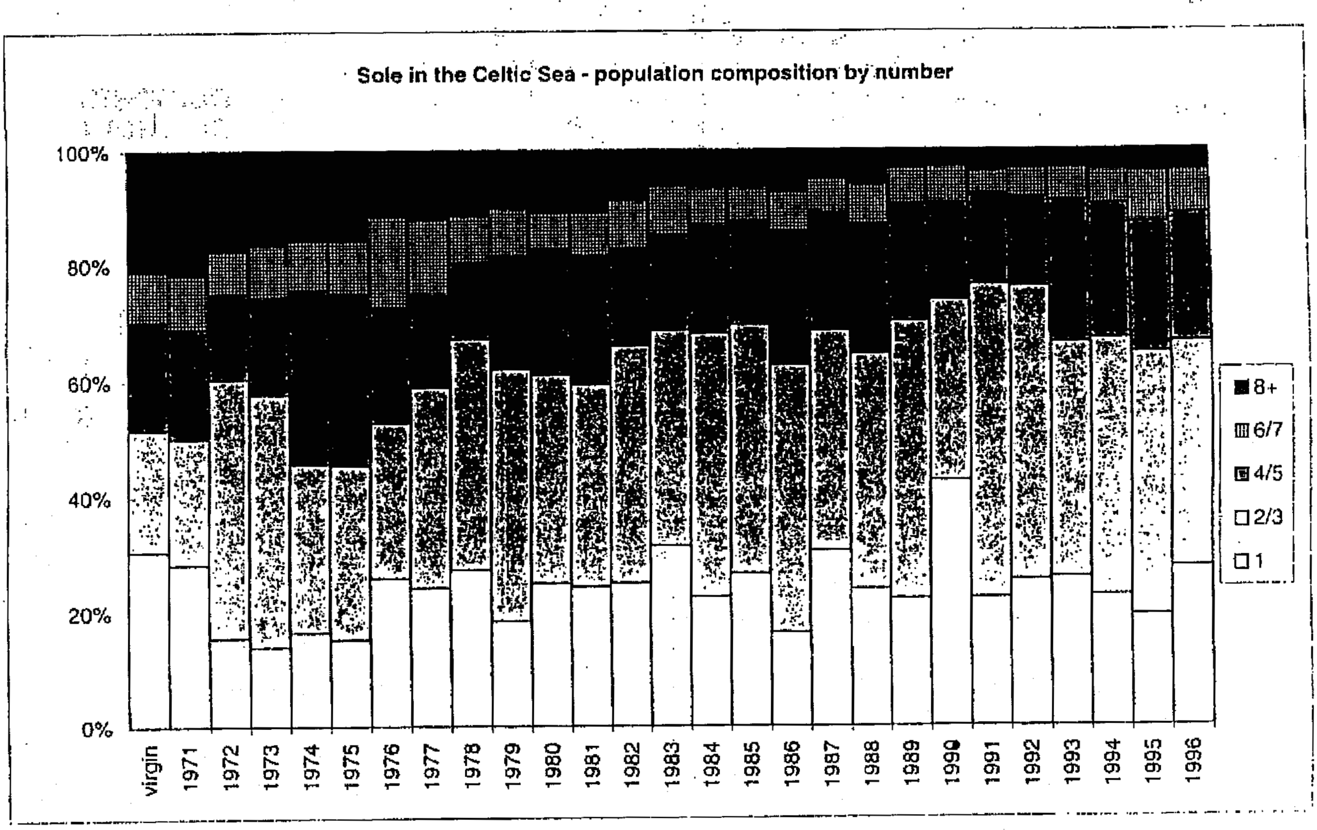


Figure 5.3.3.2.2.1. Sole in the Celtic Sea - population age (years) composition by number from 1971–1996.



5.4 OSPAR Region IV

An overview of the fisheries in this region is shown in Figure 5.4.1.

5.4.1 Fisheries in the northern parts of the Bay of Biscay (VIII a, b, d, e)

Most of the demersal fisheries in this area have mixed species catch. Although it is possible to associate specific target species with particular fleets, various quantities of cod, whiting, hake, anglerfish, megrim, sole, plaice, and *Nephrops* are taken together, depending on gear type.

In the Bay of Biscay, Celtic Sea, and western Channel, fisheries for demersal species, mainly cod, whiting, sole, and plaice, are conducted by Spain, Belgium, France, Ireland, and the UK. The principal gears used are otter trawls and beam trawls. The targeting of sole and plaice using beam trawls became prevalent during the mid-1970s, leading to an increase in the landings of these two species. The gradual replacement of otter trawls by beam trawls has occurred in the Belgian and UK fleets. In the Bay of Biscay there has been a substantial increase in the coastal gillnet fishery targeting sole.

A trawl fishery for anglerfish by Spanish and French vessels developed in the Celtic Sea and Bay of Biscay in the 1970s and expanded until 1990. The fishery has become dependent on small juvenile fish for which there is no minimum landing size. In addition, a gillnet fishery has developed in the Celtic Sea in the last decade.

Nephrops are an important component of the fisheries in this area. These fisheries developed in the 1970s and 1980s and effort increased continuously until recent years. Landings increased initially as effort increased, but these have tended to stabilize or decline at continuing high effort levels. The mesh size when fishing for *Nephrops* can lead to a significant by-catch of juvenile fish, notably hake.

The assessment units used for demersal stocks in this area are small and catches deriving from them are generally in the region of 10,000 t or less. However, the TACs set for the stocks often cover many assessment units. In addition, for a number of units, there are insufficient data for adequate assessments. This means that TACs which cover a number of heavily exploited stocks comprise a summation across units of analytical forecasts and average catches which offer no effective management control of the exploitation rate. Since a number of stocks affected by this problem are regarded as being close to or outside safe biological limits, there is a need to reconsider the areas on which TACs are set if management is to improve. In 1997, the assessment areas for cod and whiting have been expanded to include Division VIIj, k.

A notable feature of the demersal fisheries in this area is their mixed nature. Use of measures to reduce fishing mortality directly, such as effort reductions in fleets, is likely to avoid a number of the disadvantages of catch controls in regulating the exploitation rate.

The fisheries in the Celtic Sea are very similar to the fisheries in the Bay of Biscay and some of the same fleets operate in both areas. However, the technical measures in the two areas differ. The minimum mesh sizes in the Celtic Sea are often different from those in the Bay of Biscay. This difference makes enforcement more difficult since vessels can carry multiple mesh sizes and may fish in the Celtic Sea using the lower mesh sizes without being detected. It is noted, however, that the recent European Commission proposal to revise the existing conditions on technical measures attempts to eliminate this problem.

Two stocks of anchovy are considered in the Iberian Region, one in Sub-area VIII and one in Division IXa. The Spanish and French fleets fishing for anchovy in Sub-area VIII are well separated geographically and in time (the Spanish fleet operates in Division VIIIc in spring and the French fleets in Division VIIIa in summer and autumn and in Division VIIIb in winter and summer). Changes in the catch-at-age composition between the 1984–1996 period and the earlier years could be related to a higher dependence of catches on recruitment in recent years and a change in the seasonality in this fishery. The number of Spanish purse seiners for anchovy has remained stable since 1990 and a slight increase in the number of French purse seiners has been observed in the last five years. A sharp increase in fishing effort for anchovy in the Bay of Biscay has occurred since 1987, mainly due to the increased effort in the French pelagic trawl fleet.

5.4.2 Fisheries in the Iberian region (Divisions VIIIc and IXa)

The Iberian region along the eastern Atlantic shelf is an upwelling area with high productivity; this phenomenon takes place during late spring and summer due to the northerly winds and typical oceanography system in the area.

The fisheries in the region are also of a mixed nature. Different kinds of Spanish and Portuguese fleets operate in the Iberian region: one is the trawl fleet (single, pair, and crustacean trawlers) fishing for species such as hake, blue whiting, horse mackerel, megrim, anglerfish, mackerel, *Nephrops*, and cephalopods as the main species. Other fleets fishing for different target species are longliners fishing for hake and mackerel, gillnets targeted for hake, anglerfish, and mackerel, and purse seiners which target sardine and anchovy and, secondarily, horse mackerel and mackerel.

Many bottom trawlers are fishing in the southern part of Division IXa (Gulf of Cadiz); these trawlers are smaller than those operating in the northern parts of the Iberian region. The composition of their catches is also different. They are fishing for hake as well as crustaceans and cephalopods.

The number of trawlers in the Iberian region has been decreasing since the early 1980s, resulting in a decreasing trend in the overall effort in the Portuguese and Spanish fleets. The fleets operating with gillnets and longlines have also declined in terms of the number of boats in recent years. Spanish and Portuguese boats using trawl, longline or gillnets are currently subjected to a controlled and restricted system of reduced days of entrance into the harbour to land the catch for the market with the objective of decreasing the total fishing effort.

Traditionally the anchovy fishery in Division IXa was located in the Gulf of Cadiz (Sub-division IXa south), except in 1995 when the bulk of the fishery was located to the north of Portugal and to the west of Galicia (Sub-Division IXa north) and very reduced in the Gulf of Cadiz, owing to the exceptional availability of anchovy in the northern part of the Division IXa, due to environmental factors.

The catches of horse mackerel in Divisions VIIIc and IXa have been relatively stable over the last ten years. The proportion of landings by different gears has changed, i.e., trawl catches are decreasing while the purse seine catches are increasing.

The fisheries in the Iberian region are managed by a TAC system and technical measures. Common mesh sizes for trawls are 65 mm, except for trawlers directed to blue whiting or horse mackerel (40 mm). In the Gulf of Cadiz the legal trawl mesh size is 40 mm. The technical measures are minimum landing sizes and seasonal closures to protect juvenile hake.

In OSPAR Region IV, the species of interest to OSPAR are sole, hake, and mackerel, according to the terms of reference.

5.4.3 Sole

5.4.3.1 Bay of Biscay sole stock

5.4.3.1.1 Context - patterns of numbers, biomass, landing, and fishing mortality

Catches have increased continuously in the last two decades from 3000 t in the late 1970s up to 7000 t at the beginning of the 1990s. In recent years, international landings and catches have decreased gradually. The 1996 landings at 5853 t are below the level of the previous three years. Since 1984, catches of sole by French small-mesh shrimp trawlers have decreased markedly, and the gillnet and trammel net fishery has expanded (ICES, 1998b).

The mean spawning stock biomass has been 13,000 t. The SSB decreased from 15,100 t in 1987 to 13,200 t in 1990, increased again to 16,700 t in 1993, and decreased to 12,900 t in 1996. A succession of below-average recruitment in 1991–1993 has contributed to this decline in SSB since 1993. Recruitment has fluctuated around 50 million 0-groups over the period, though the 1995 year class is estimated to be more than 40 % above the mean (72 million 0-group).

From 1979–1996, fishing mortality has steadily increased, reaching its highest level in 1994 (0.54). Mean F on ages 2–6 is estimated to have been 0.51 in 1995 and 0.47 in 1996. Trends in yield, fishing mortality, spawning stock biomass, and recruitment are plotted in Figure 5.4.3.1.1.1 from ICES (1998b).

Landings of sole in the Bay of Biscay have remained at a high level since 1988, even though fishing mortality increased continuously until 1994. This is a consequence of an exploitation pattern which has been improving over the same period and rather stable recruitment. As four of the five most recent year classes are below average, SSB has declined from its peak in 1993. At the current level of F, the level of probability for SSB in 1999 to fall below the lowest level (B_{loss}) in the time series for which the assessment is reliable (1984–1996) is less than 5 % (ICES, 1998b).

5.4.3.1.2 Change in size and/or age composition

The proportion of age groups 0, 1, and 2 (juvenile fish) and age group 3+ (mature fish) in the French catches has changed over the period 1979–1996. After 1984, with the evolution of the different métiers in the French fishery, the proportion of young soles (age 3 and under) in the catches has declined.

Figure 5.4.3.1.2.1 indicates the evolution of the proportion of the number at age for the period 1979–1996.

5.4.3.1.3 Changes in spatial distribution

Maps of sole distribution from French Groundfish surveys are available at the Institute in Nantes.

5.4.4 Mackerel

5.4.4.1 Northeast Atlantic mackerel

The mackerel caught in the Northeast Atlantic waters is treated as a combined stock constituted by three components: the North Sea component (Sub-area IV, Division IIIa), the western component (Divisions VIIIa,b,e, Sub-areas VI, VII, and Division IIa) and the southern component (Divisions VIIIc, IXa), attributed to three major spawning areas (ICES, 1996). The western mackerel component is known to undertake large-scale migrations between summer feeding grounds in the North and Norwegian Seas and its spawning areas south and west of Ireland (ICES, 1998a).

The schools of this highly migratory species are harvested by quarter and area in different fisheries using different gears, across all the OSPAR regions. The bulk of the catches are taken in OSPAR Region III and in OSPAR Region II. The gears used in each fishery are briefly described in the next section.

5.4.4.1.1 Context - patterns of numbers, biomass, landings, and fishing mortality

During 1975 to 1996 the total estimated catch of the combined stock has been rather constant around a mean value of 700,000 t. In 1996 the total catch showed a reduction of about 200,000 t compared with 1995. In 1996 fishing mortality decreased (Figure 5.4.4.1.1.1), which seems to be due to TAC constraints applied to the combined stock. The North-East Atlantic mackerel is considered at present by ICES (1988 ACFM October) to be within safe biological limits.

The mackerel north component is treated in Section 5.2.4 under OSPAR Region II.

In the western component, the catches developed from low levels in the 1960s to more than 800,000 t in 1993. The main catches are taken in directed fisheries by purse seiners and mid-water trawlers. Large catches have been taken in the northern North Sea and in the Norwegian Sea from 1989–1995. The SSB declined in the 1970s from 3.5 million t and hasremained stable since then above 2.3 million t. Recruitment (at age 0) was highly variable during the 1970s, and has remained at high levels since that time. There is no separate ICES advice for the western component of the stock, which dominates with 85 % of the catches of total combined stock (ICES, 1998a).

Mackerel in the southern component is a target species for the Spanish hand line fleet during the spawning season in the Bay of Biscay (east of Division VIIIc) during which about one-third of the total catches of this component are taken. In Division IXa the adult mackerel is a by-catch of hake gillnets exploited on the slope of the continental shelf. On the continental shelf of Division IXa, juvenile mackerel is a by-catch of the bottom trawl fishery directed to hake and horse mackerel. The SSB of this component was estimated by the 1995 egg survey as 300,000 thousand t (ICES, 1997). The landings have remained relatively stable during the last twenty years (1976–1996) around 22,000 t. Figure 5.4.4.1.1.2 indicates the trends in biomass landings (ICES, 1998).

The assessment and management advice of the North-East Atlantic mackerel fisheries over the period 1999–2002 may be greatly influenced by the results of the international mackerel egg surveys which will take place in the southern and western areas in 1998 and in the North Sea in 1999.

5.4.4.1.2 Changes in size distribution and/or age distribution

In the western component, the catches have been mainly composed of 2–7 year-old fish which constitute 72 % of the total catches.

Figure 5.4.4.1.2.1 shows the proportion of biomass at age from 1980–1994.

5.4.4.1.3 Changes in spatial distribution

Unlike the North Sea mackerel and the southern mackerel, the western mackerel is known to undertake large scale migrations between summer feeding grounds in the North Sea and Norwegian Sea and its spawning areas south and west of Ireland.

Previous studies have shown that the timing and pattern of the post-spawning northerly migration has been relatively stable. The return southerly migration, however, has changed dramatically in both timing and route over the last 20 years (Walsh and Martin, 1986; ICES, 1981, 1986, 1988a, 1988b).

During the 1970s and early 1980s, this migration occurred in late summer and early autumn, with the fish moving through relatively shallow waters and giving rise to a very substantial fishery in the Minch (west of Scotland 57–58°N 6°W).

Since then the migration has occurred progressively later in the year, but has stabilized since 1992. The fish now do not cross the 4°W line until mid-January, with the fish being found west of Scotland and Ireland in February. The timing of migration across the 4°W line is of considerable importance to commercial fishermen since this latitude separates two management areas and fishing to the east of it is subject to severe quota restrictions. The later the fish arrive, therefore, the shorter the fishing season for many fishermen. Walsh and Martin (1986) suggested, based on commercial catch data, that this change may have been related to changes in the hydrography of the area following the 1970s salinity anomaly.

Recent work on the migration of mackerel (*Scomber scombrus*) has suggested that water temperature is the major environmental parameter controlling the direction and speed of migration (Walsh and Martin, 1986; Castonguay *et al.*, 1992; Walsh *et al.*, 1995; Reid *et al.*, 1997). The winter migration of the western mackerel from feeding grounds in the North Sea and the Norwegian Sea to spawning areas south and west of Ireland occurs in the months of December to March. The migration path follows the shelf edge for most of its route, with the fish being found generally between the 100 and 250 m contours (Walsh *et al.*, 1995). Walsh *et al.* (1995) showed that the migration around the north of Scotland appears to follow a track which coincides with a tongue of warmer water transported northwards up the shelf edge by the shelf edge current (SEC). Observations made during an acoustic survey in January 1995 (Reid *et al.*, 1997) indicated that when the migrating mackerel encountered an intrusion of unusually warm water onto the shelf, the fish stopped their active migration and adopted different schooling behaviour. This led to the hypothesis that the spawning migration of this species may be influenced by 'enviroregulation'. This is a process by which the fish select their immediate environments by behavioural means (Neill, 1984). If the fish find themselves in some 'non-preferred' temperature, they may swim faster or deeper in an attempt to gain more preferred temperatures. For example, Olla *et al.* (1975) showed that mackerel swam faster at water temperatures below 7 °C. Migration may be triggered when water temperature drops below a threshold, and the subsequent migration route constrained by the narrow tongue of warm water derived from the northward flowing current.

If enviroregulation is constraining the distribution and migration of the mackerel, then it would be expected that immediately prior to the start of migration, the fish would tend to concentrate in the warmest areas of the North Sea, leaving those areas which cool earlier or faster. As these areas of aggregation also cooled, migration would commence. The shelf edge area adjacent to Viking Bank is likely to be one area where the water will stay warmer longer due to Atlantic inflow along the shelf margin.

An exceptionally large number of juvenile mackerel (1996 year class) was observed in the southern North Sea and adjacent areas during 1997, and its spawning component attribution remains unknown (ICES, 1998b). Figures 5.4.4.1.3.1.a and 5.4.4.1.3.1.b indicate the distribution of the juvenile mackerel as indicated by the International Groundfish Surveys covering the entire Northeast Atlantic, carried out during the forth quarter by each nation. These databases were created, assembled and processed under the EU project Shelf Edge Fisheries and Oceanography Studies (SEFOS) from 1993–1996 (Reid, *et al.*, in press).

5.4.5 Hake

Hake is distributed over all the western Atlantic coasts from the south of the Iberian Peninsula up to northern Scotland. Two components are distinguished for assessment purposes: the northern and southern stocks. The northern stock (ICES Division IIIa, Sub-areas IV,V,VI and VII, and Divisions VIIIa, b) encompasses the northern part of OSPAR Region IV and the southern part of OSPAR Region III, its distribution does not allow to subdivide its biological features between OSPAR Regions III and IV.

The Sub-areas IV and IIIa are OSPAR Region II, but there the catches of hake are the smallest proportion compared with the other OSPAR regions where hake are caught.

Consequently the northern stock analysis is given in OSPAR IV region according to the main importance of the Bay of Biscay nursery areas regarding to the southern Ireland ones.

5.4.5.1 Hake - northern stock (Division IIIa, Sub-areas IV, VI, and VII, and Divisions VIIIa,b)

Since the 1930s, hake has been the main species supporting trawl fleets on the Atlantic coasts of France and Spain and is present in the catches of nearly all fisheries in Sub-areas VII and VIII. Spain and France take 60 % and 25 % of the landings, respectively, and the UK reports about 10 %. After a decline in landings from the mid-1980s, an increase was observed for the first time in 1995, though they decreased again in 1996 to the lowest observed. Hake is caught throughout the year, the peak landings being made in the spring–summer months. The three main gear types used by vessels fishing for hake as a target species are lines (Enland and Wales, Spain), fixed-nets (England and Wales, Spain and France) and otter trawls (all countries). By-catches of mainly juvenile hake are taken in the *Nephrops* fisheries in the northern Bay of Biscay.

5.4.5.1.1 Context - patterns of numbers, biomass, landings, and fishing mortality

The stock is considered to be close to safe biological limits. SSB increased steadily between 1978 and 1987, coincident with a period of relatively constant exploitation, but a subsequent increase in fishing mortality was associated with a substantial decline in spawning stock biomass until 1994 (120,000 t, the lowest value in the series). Spawning stock biomass increased slightly in 1995 and 1996. Recruitment has been relatively stable oscillating around the mean value of 313 million 0-group fish since the highest recorded in 1995 (500 million). Figure 5.4.5.1.1.1 indicates the trends in biomass, landings, and fishing mortality (ICES, 1998b).

5.4.5.1.2 Change in size and/or age composition

Hake catches have been mainly composed of 0–4 age groups (see Figure 5.4.5.1.2.1).

5.4.5.1.3 Changes in spatial distribution

Hake movements are indicated by the seasonal distribution of catches in the fishery. From the beginning of the year until March/April, hake are present in the northern part of the Bay of Biscay. They appear on the shelf edge in the Celtic Sea in June and July. Between August and December, the hake fishery is centered to the west and southwest of Ireland, with a decline in catch rates in shallower waters.

These patterns are well explained by ontogenetic migrations described by national bottom trawl surveys (ICES, 1994). Hake spawn from February through July along the shelf edge, the main areas extending from north of the Bay of Biscay to the south and west of Ireland. 0-groups descend to the seabed (at depths in excess of 200 m), moving to shallower water with a muddy seabed (75–120 m) by September. There are two major nursery areas: in the Bay of Biscay and off southern Ireland. When three years old, hake begin to move into the shallower regions of the Bay of Biscay and Celtic Sea, but as they approach maturity they disperse to offshore regions.

Analysis of the hake spatial distribution in the Bay of Biscay (Petigas *et al.*, 1991) shows that the locality and surface occupied do not change over the years studied while the densities do change.

Figures 5.4.5.1.3.1.a and 5.4.5.1.3.1.b indicate the spatial distribution of northern hake by age group from 1990–1992.

5.4.5.2 Hake - southern stock in the Iberian region (Division VIIIc and Sub-areas IX and X)

5.4.5.2.1 Context, pattern of numbers, biomass, landings, and fishing mortality

The situation for hake is alarming. The stock is considered to be outside safe biological limits. Landings have declined almost continuously since 1983 (25,000 t), reaching their lowest level on record in 1996 (10,000 t). Spawning stock biomass decreased very sharply between 1984 and 1986, from 59,000 t to 26,000 t, and the 1996 spawning stock biomass was near to its lowest recorded in 1995 (15,000 t). Recruitment has declined steadily since 1984 (around 120 million 0-group) and, with the exception of one year (1992), has been poor since 1989 (80 million 0-group). The assessment indicates that there has been a decreasing trend in fishing mortality since 1986. Figure 5.4.5.2.1.1 indicates the trend in biomass, landings, and fishing mortality (ICES, 1998b).

Fishing mortality in 1996 was at F_{med} (0.23). There is evidence of reduced recruitment below a spawning stock biomass of 23,000 t. At the current level of fishing mortality, the probability of SSB in 1999 remaining below MBAL is more than 50 % (ICES, 1998b).

5.4.5.2.2 Changes in size distribution and/or age composition

Within the period 1978–1996, a change in the length composition of the landings appears during 1982–1996. There has been a decrease in the estimated number of fish smaller than 30 cm (ICES, 1998b). The change is attributed by ICES (1998b) to the difficulties in obtaining good samples of small fish (age 0 and 1) since the enforcement in 1989 of a minimum legal fish size (27 cm). This is supported by an increase in mean weight in landings during the period.

Figure 5.4.5.2.2.1 indicates the evolution of the proportion of the biomass at age from 1982 to 1996.

Although for this depleted stock, changes in the size abundance of the adult hake are expected, there is no evidence that this occurs from the analysis. One explanation is that the adult hake are mainly on the slope, living on non-trawlable areas where neither the groundfish survey nor the commercial trawlers can fish.

5.4.5.2.3 Changes in spatial distribution

The Iberian region is an important nursery ground for hake. A closed area has therefore been enforced off the southwest coast of Portugal to prevent trawlers from catching juveniles during the autumn–winter when recruitment peaks.

Figure 5.4.5.2.3.1 indicates the abundance distribution of hake in OSPAR Region IV, during the fourth quarter of 1990.

5.4.6 References

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Figure 5.4.1. ICES Sub-areas and Divisions comprised in OSPAR Regions III and IV.

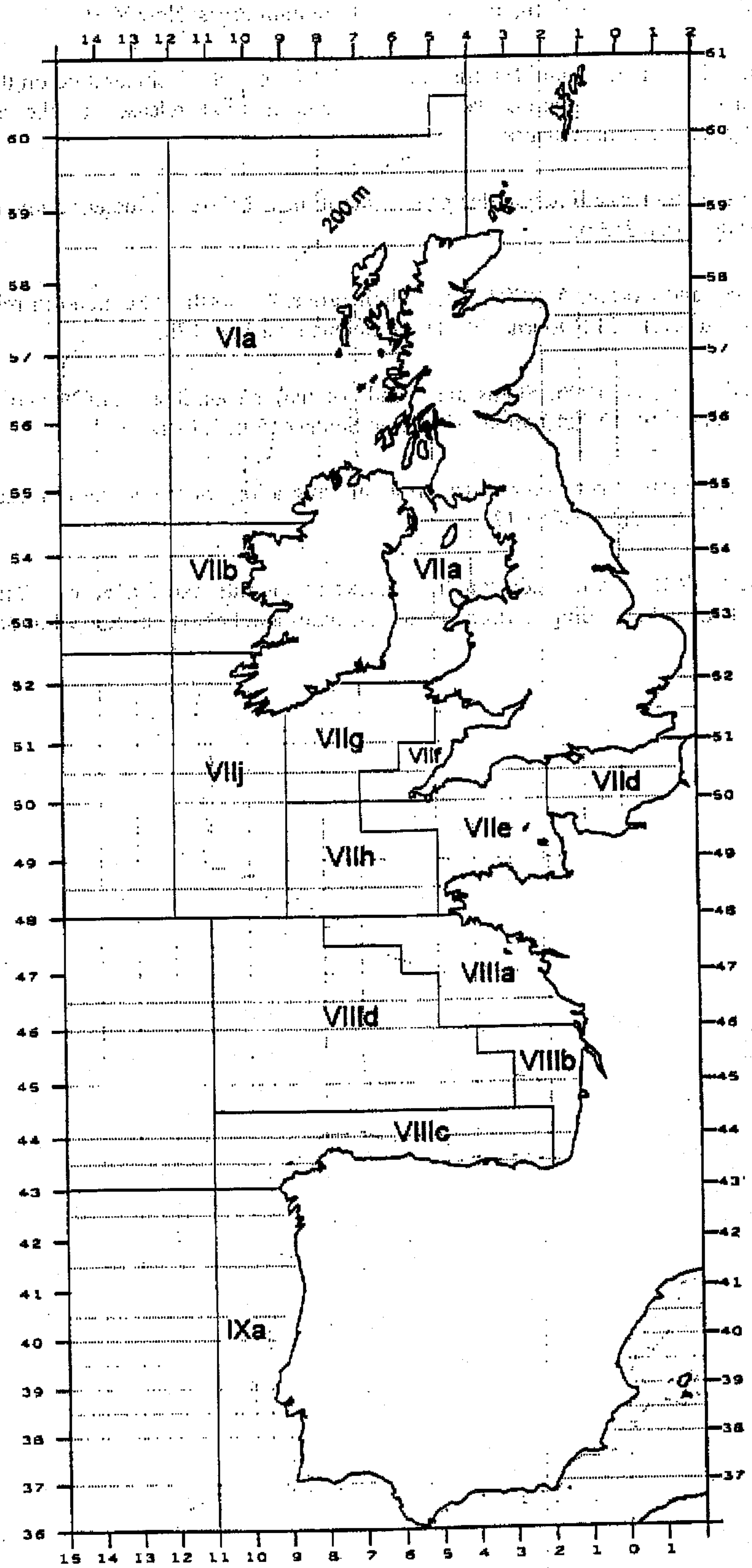


Figure 5.4.3.1.1.1. Trends in landings and recruitment (top); fishing mortality and spawning stock biomass (bottom) for sole in the Bay of Biscay from 1979–1996.

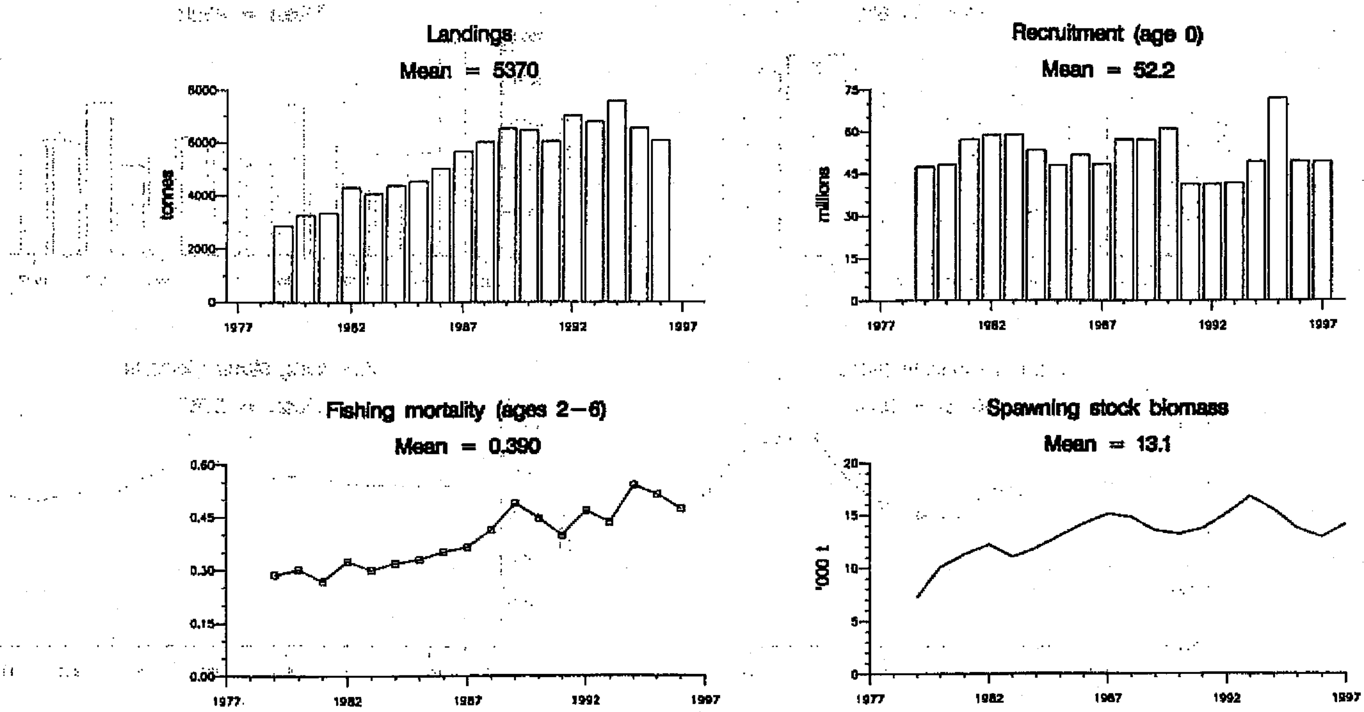


Figure 5.4.3.1.2.1. Sole in the Bay of Biscay (Sub-area VIII) - age population (years) composition by number from 1979–1996.

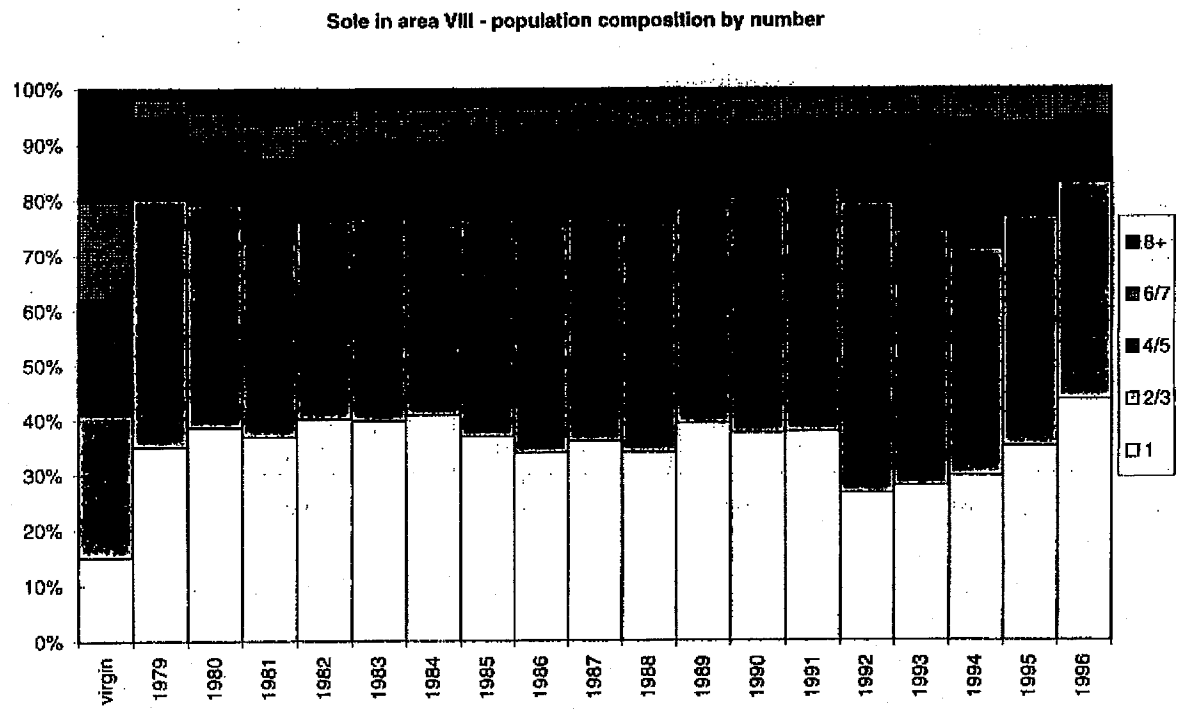


Figure 5.4.4.1.1.1. Trends in landings and recruitment (top), fishing mortality and spawning stock biomass (bottom) for North-east Atlantic mackerel from 1984–1996.

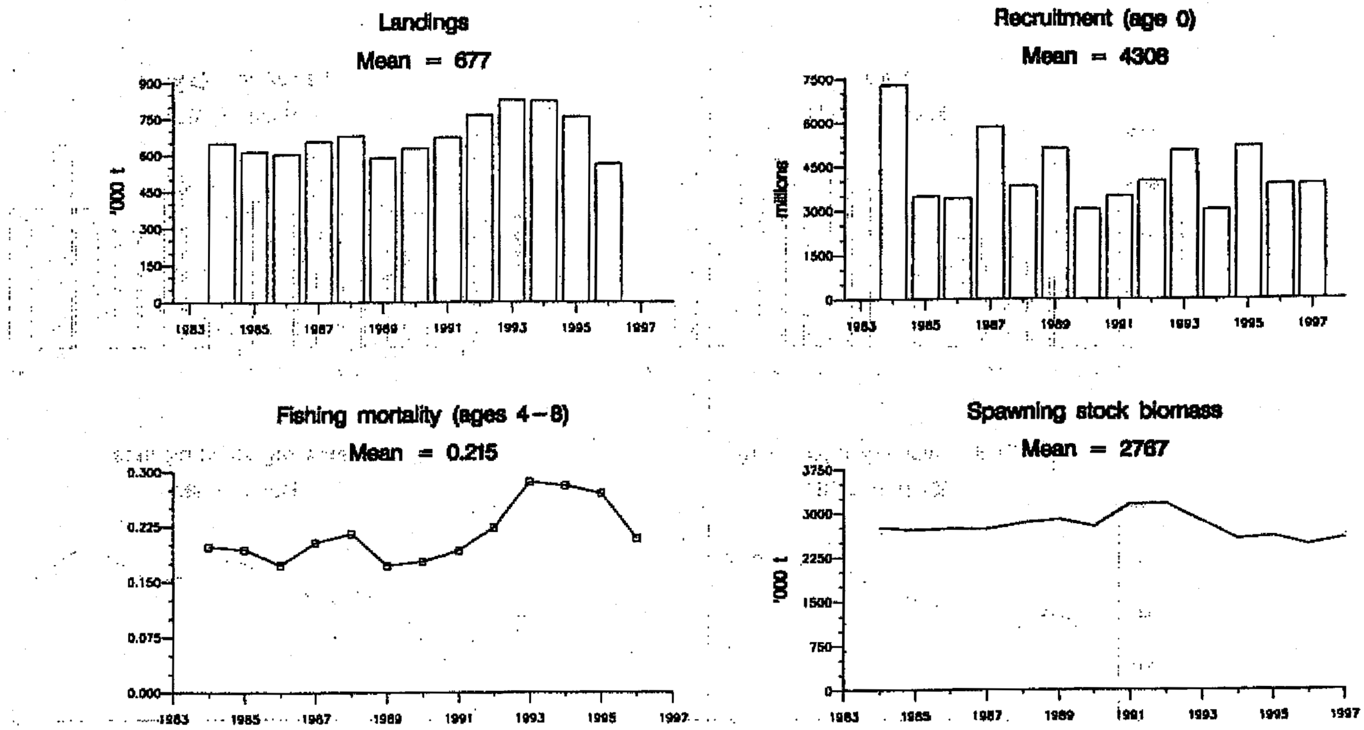


Figure 5.4.4.1.1.2. Trends in landings of Southern mackerel from 1977–1996.

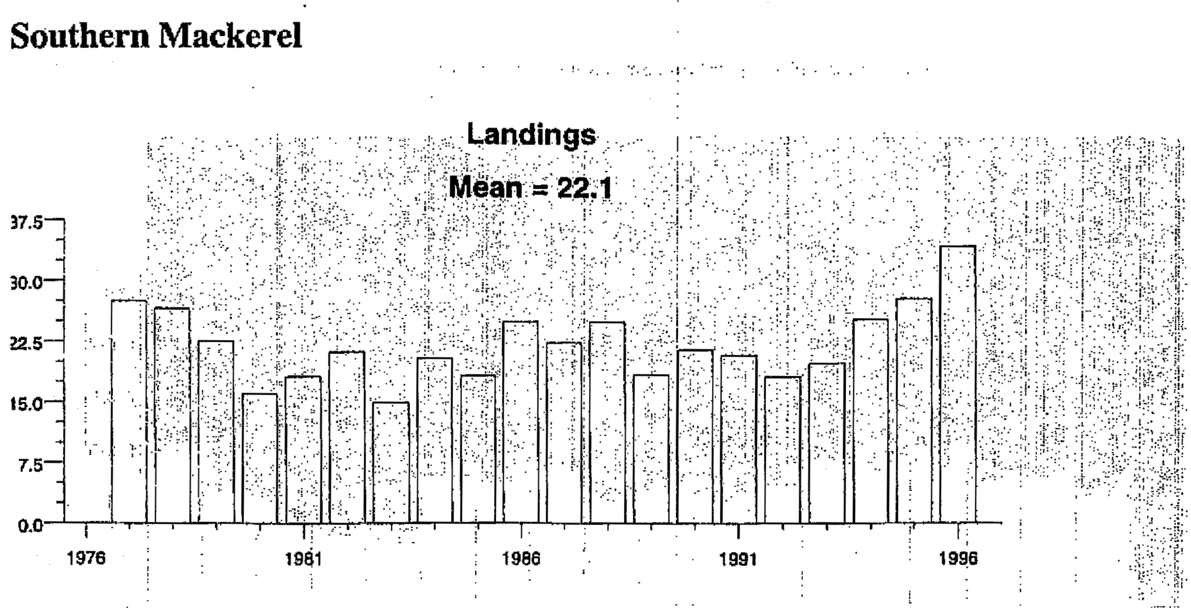


Figure 5.4.4.1.2.1. Western mackerel - age population (years) composition by number from 1980–1994.

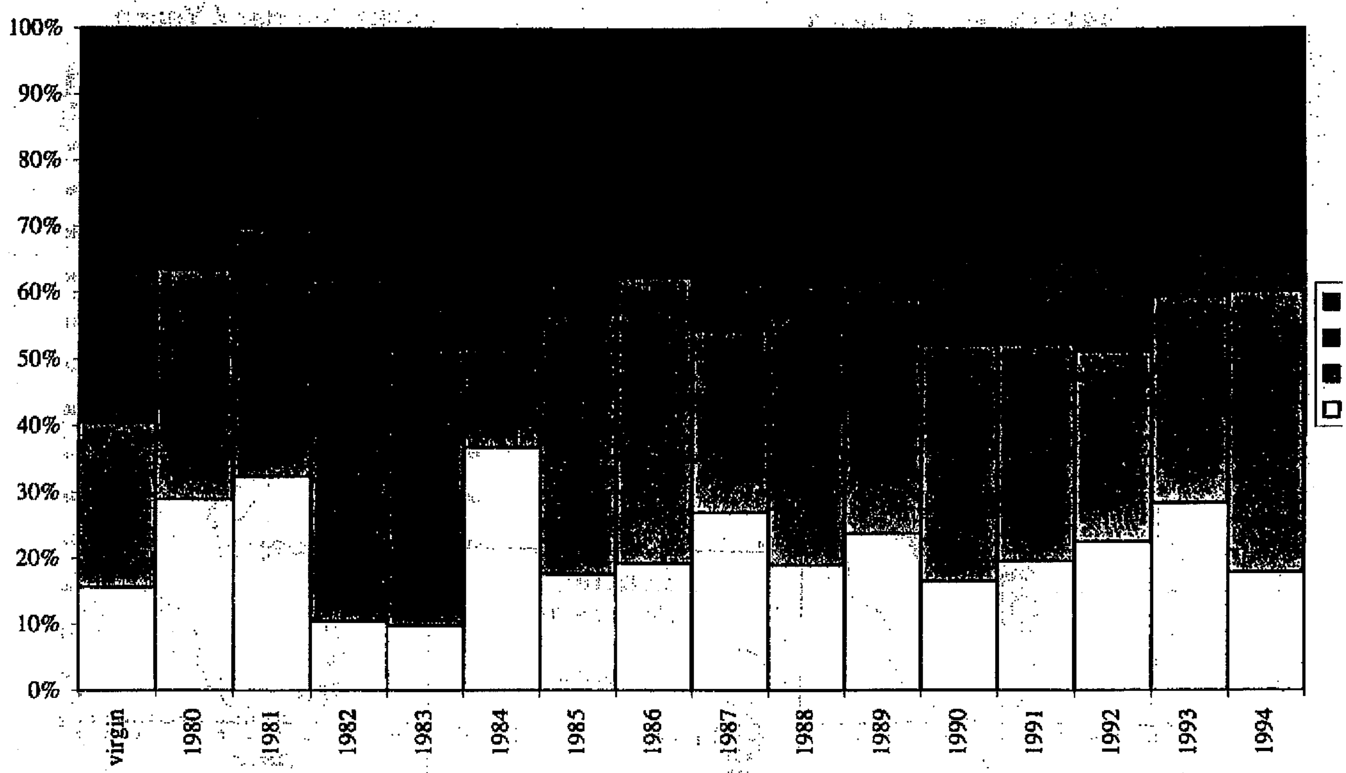


Figure 5.4.4.1.3.1.a. Distributions of juvenile mackerel (age 0) in the fourth quarter from 1989–1992.

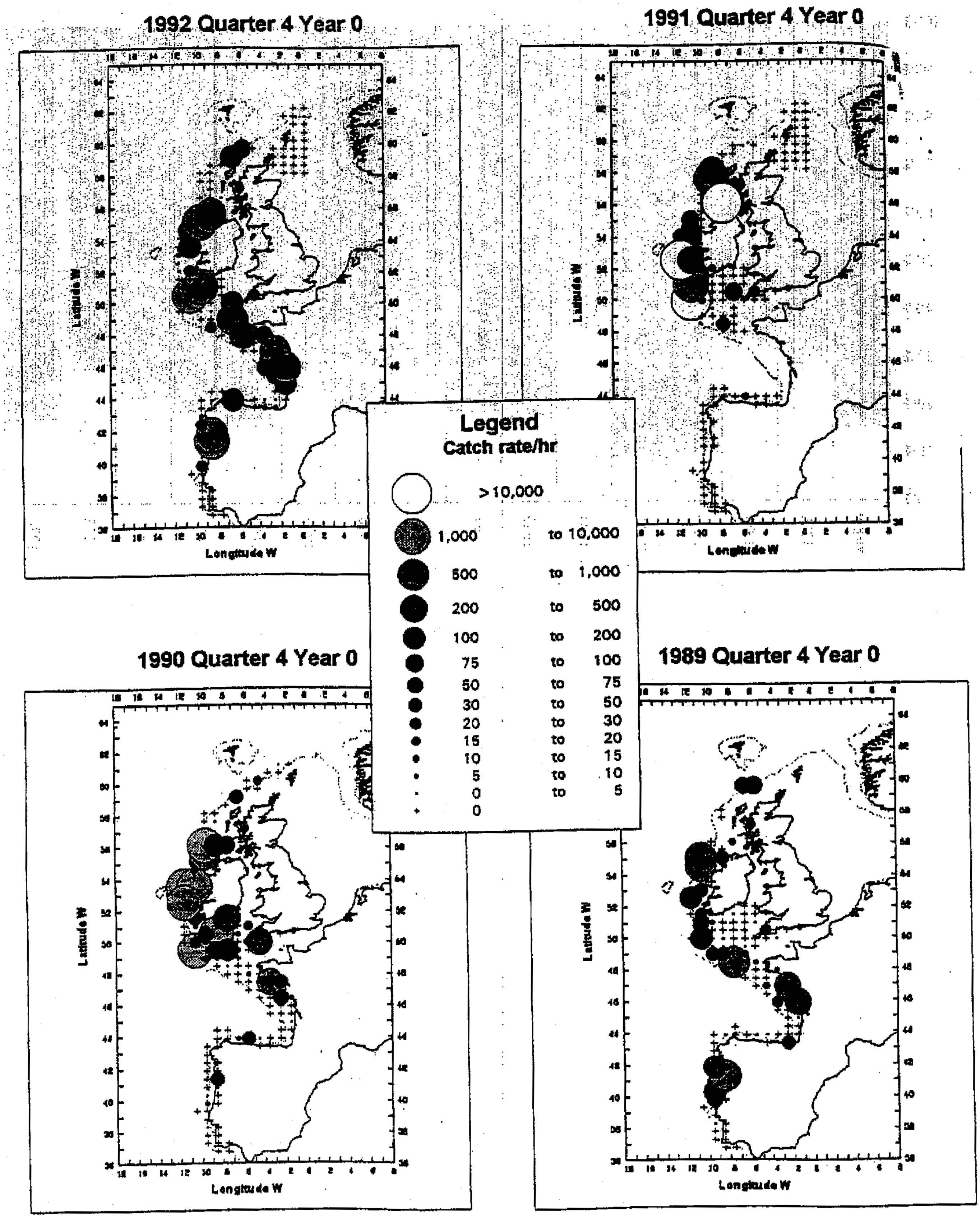


Figure 5.4.4.1.3.1.b. Distributions of juvenile mackerel (age 0) in the fourth quarter from 1993–1996.

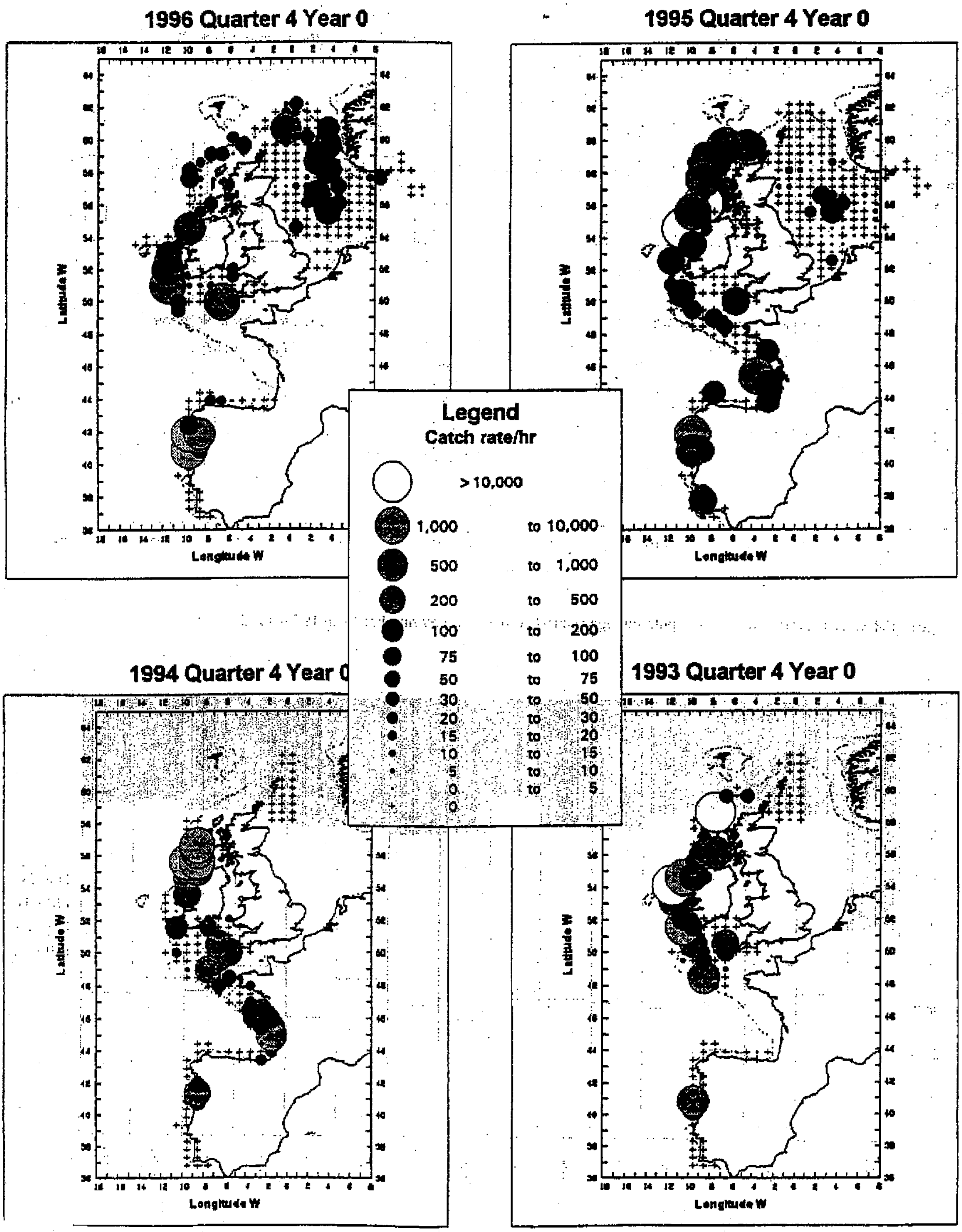


Figure 5.4.5.1.1.1. Trends in landings and recruitment (top), fishing mortality and spawning stock biomass (bottom) for the Northern stock of hake from 1978–1996.

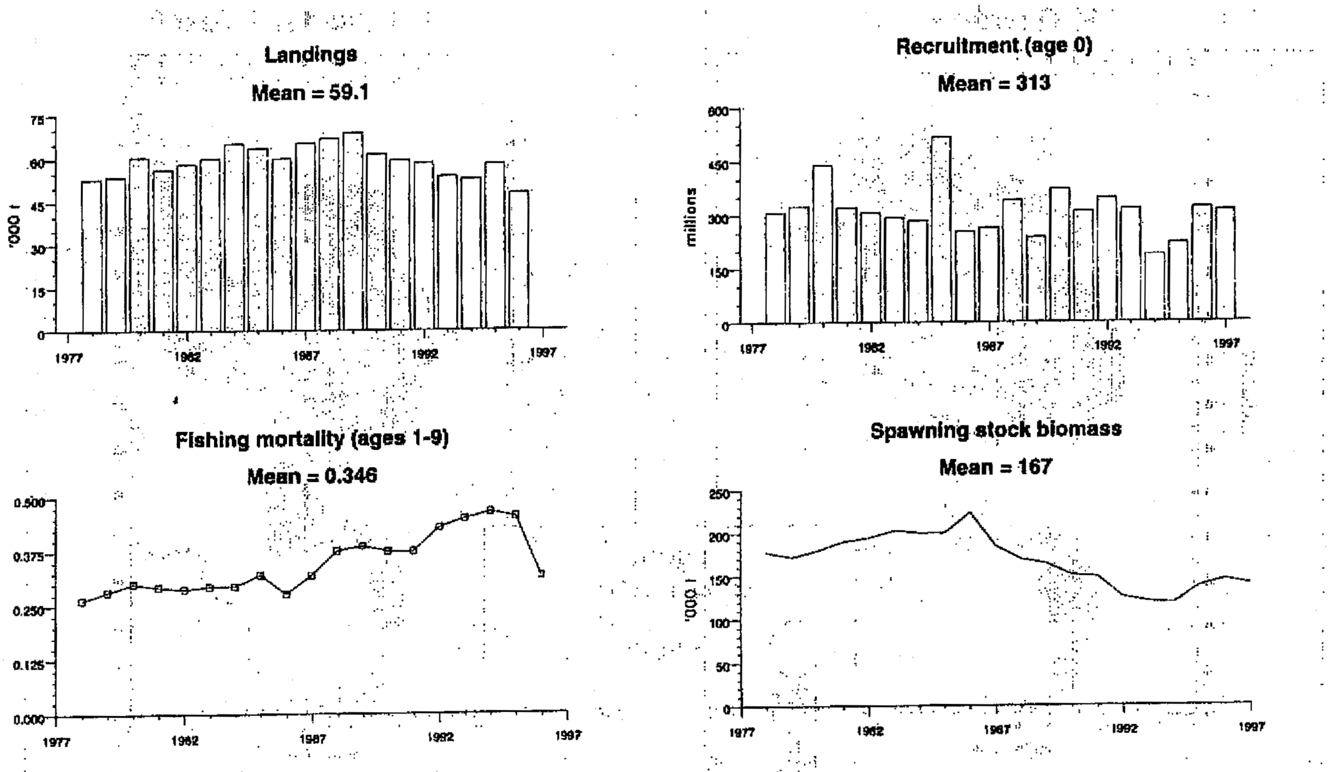


Figure 5.4.5.1.2.1. Northern hake - population age (years) composition by number from 1979–1993.

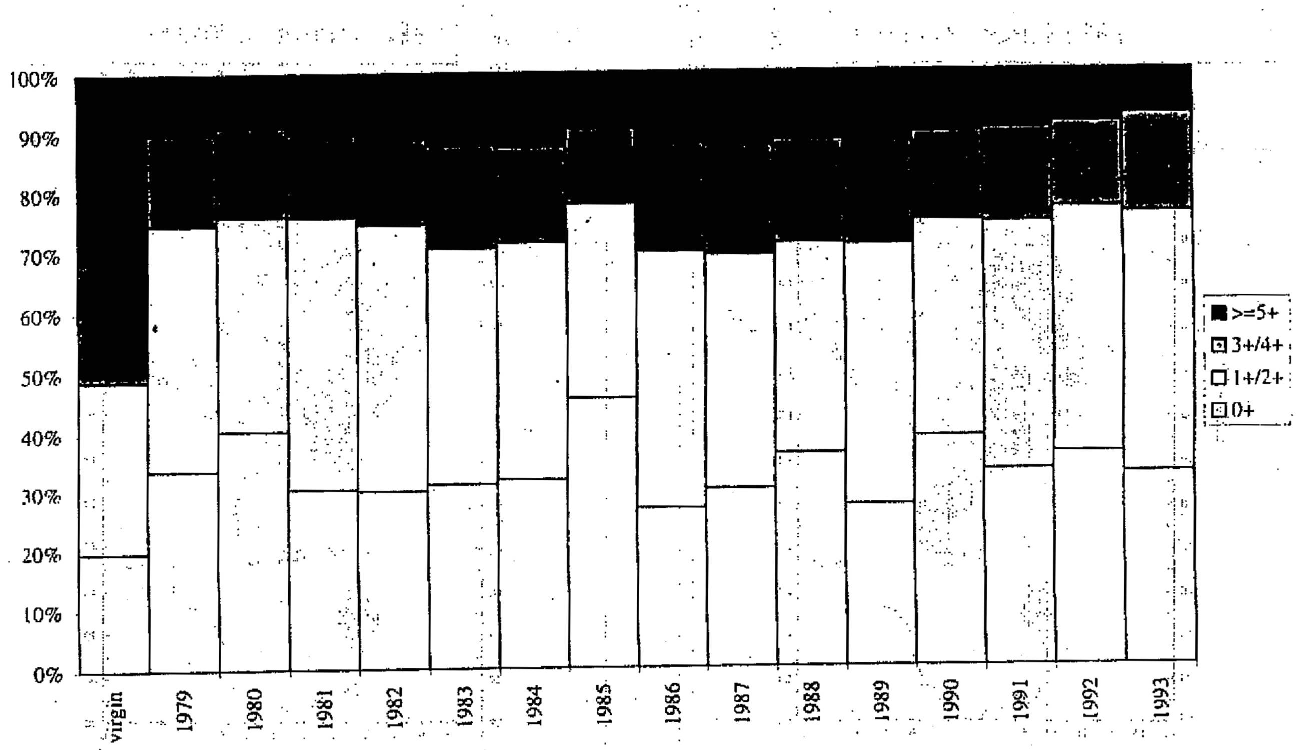


Figure 5.4.5.1.3.1.a. Distribution of hake age groups (0–5+) observed in autumn 1990 during the EVOME survey of the Bay of Biscay.

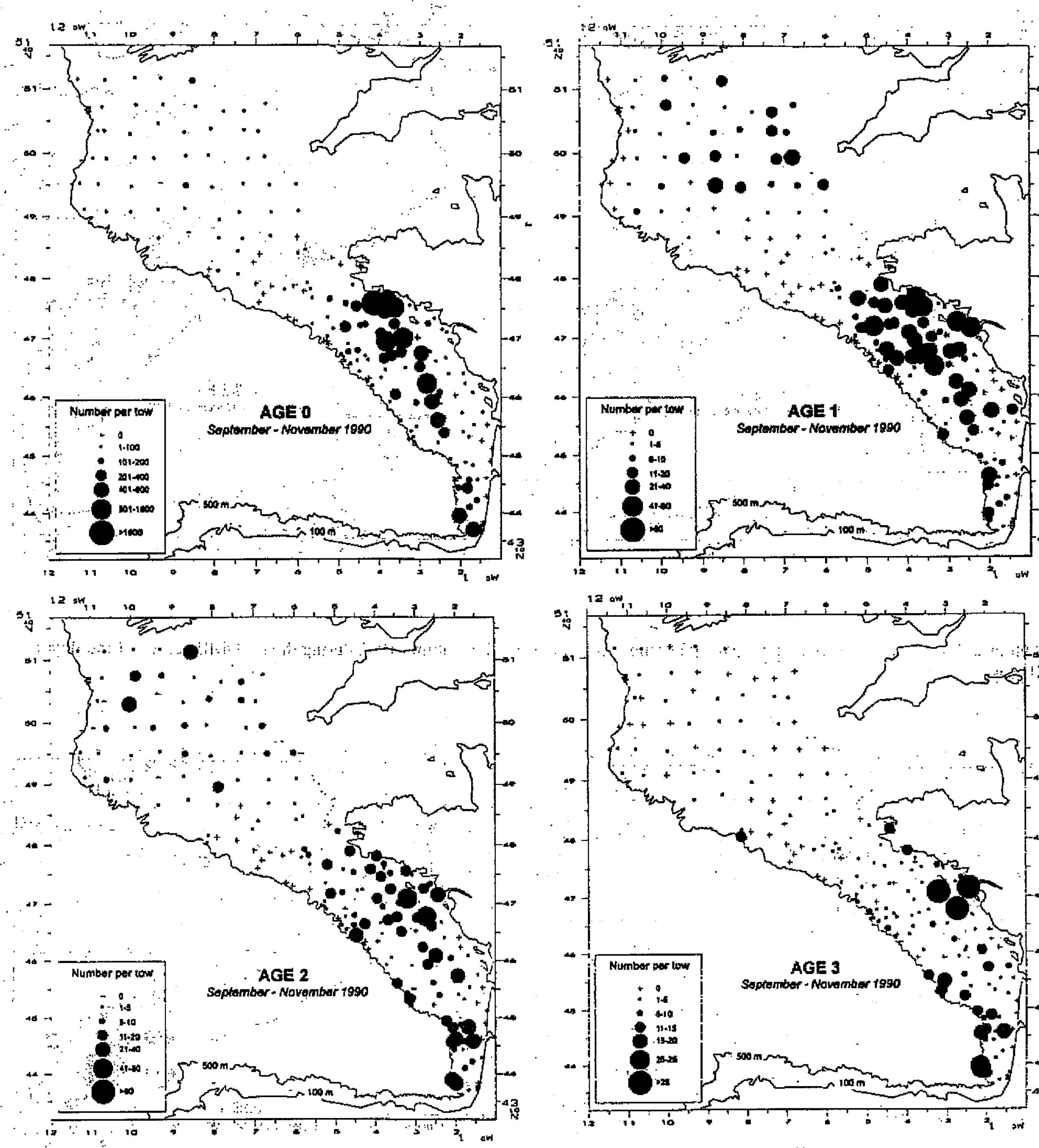
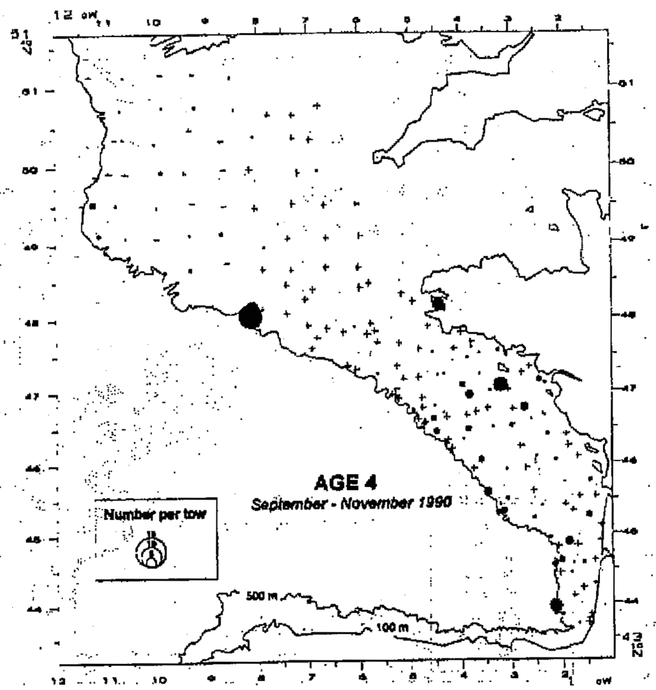


Figure 5.4.5.1.3.1.a. Continued.



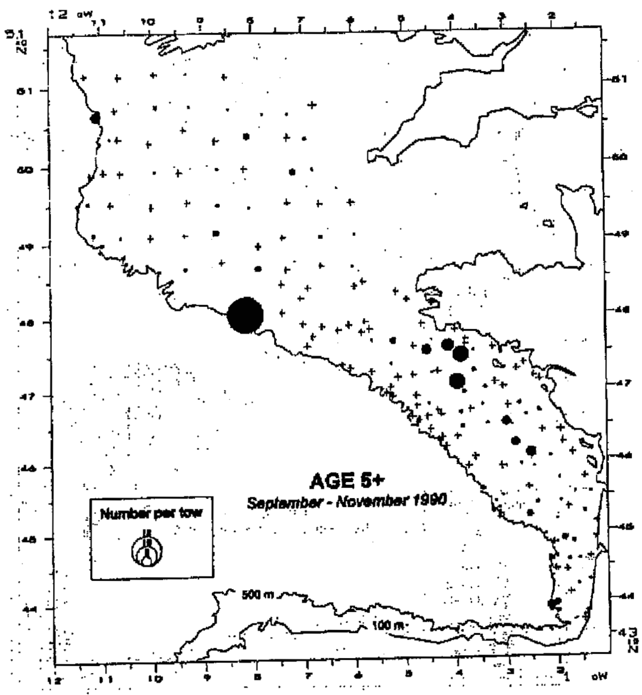
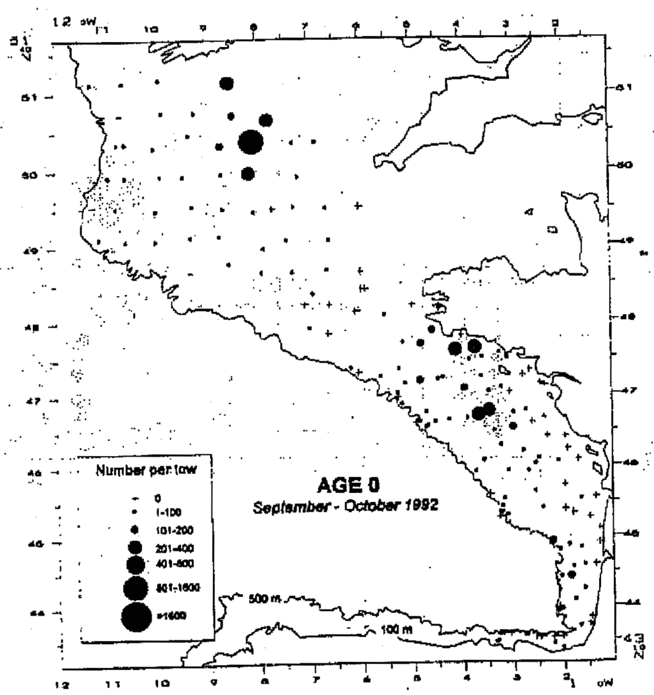


Figure 5.4.5.1.3.1.b. Distribution of hake age groups (0–5+) observed in autumn 1992 during the EVOME survey of the Bay of Biscay.



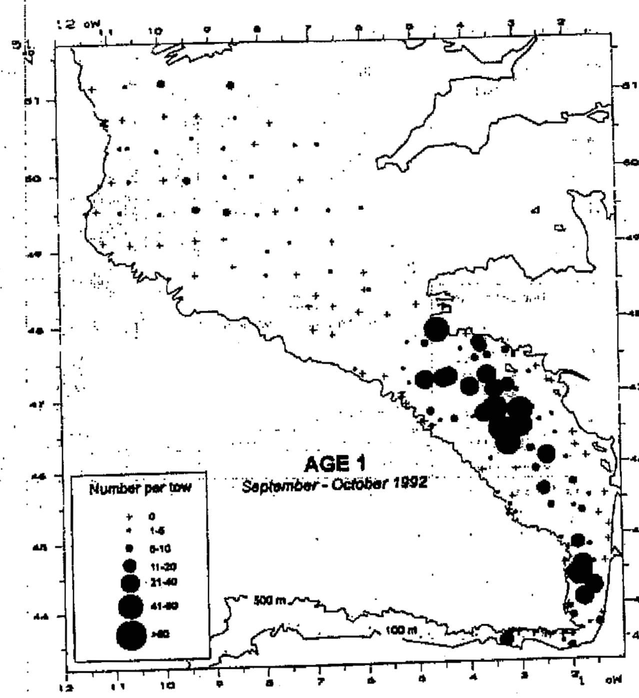


Figure 5.4.5.1.3.1.b. Continued.

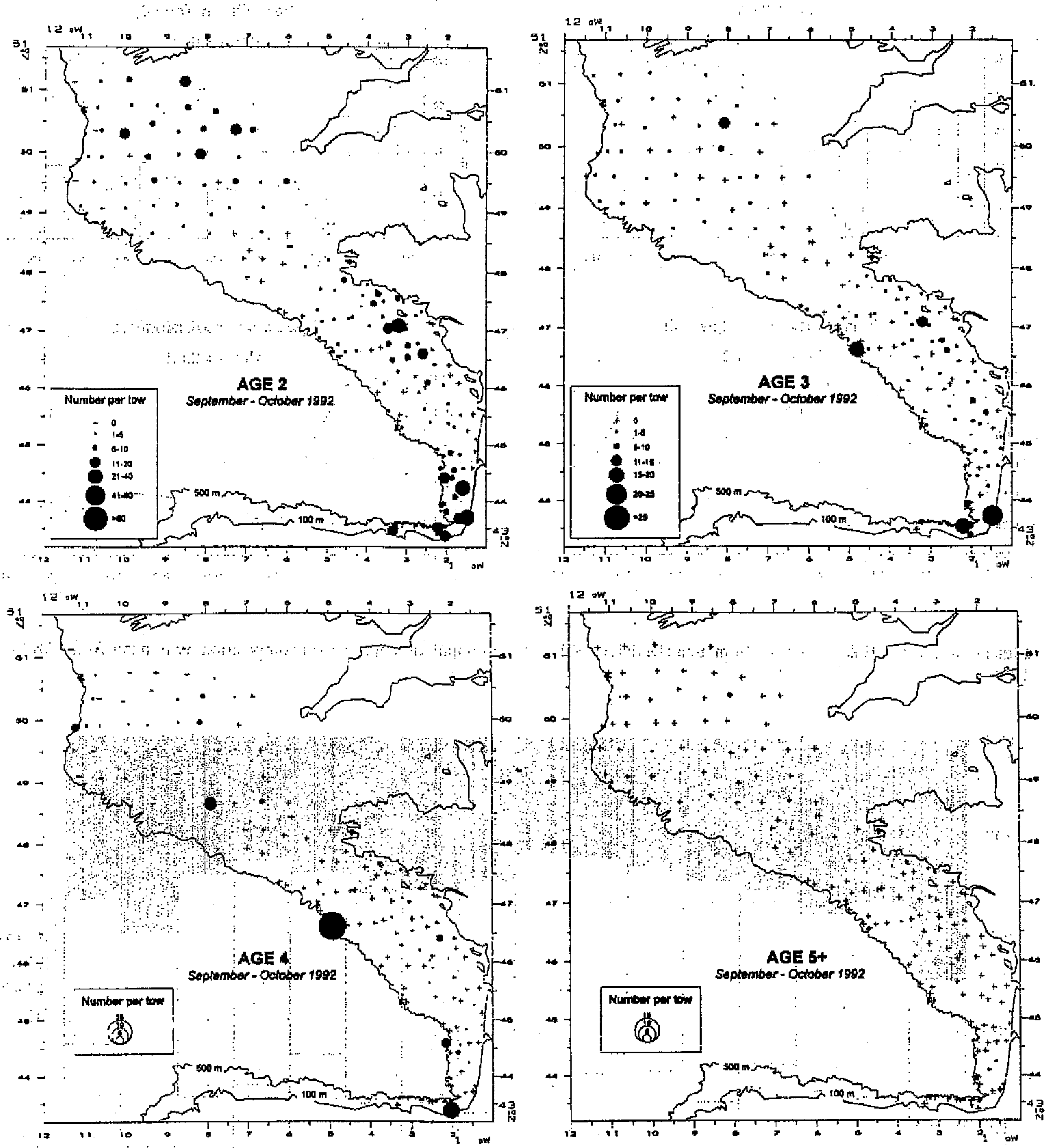


Figure 5.4.5.2.1.1. Trends in landings and recruitment (top), fishing mortality and spawning stock biomass (bottom) for the Southern stock of hake from 1982–1996.

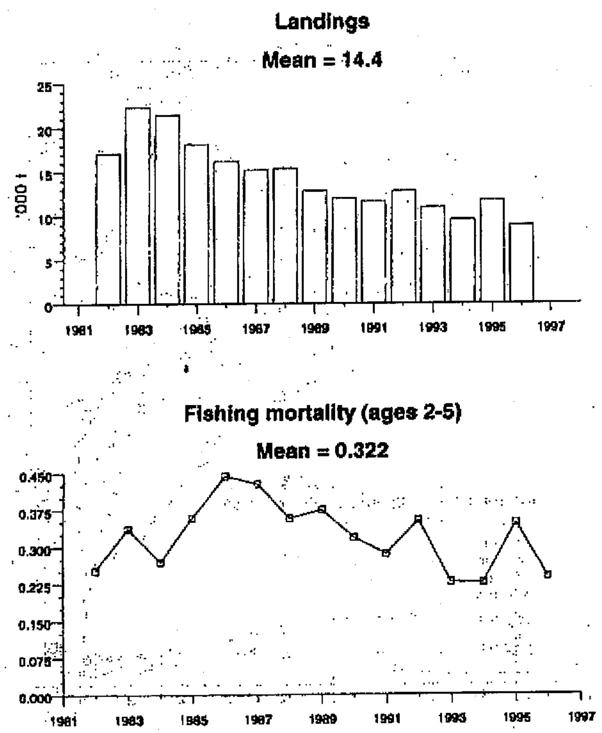


Figure 5.4.5.2.2.1. Hake in the southern area (Divisions VIIc + IXa) - population age (years) composition by number from 1982–1996.

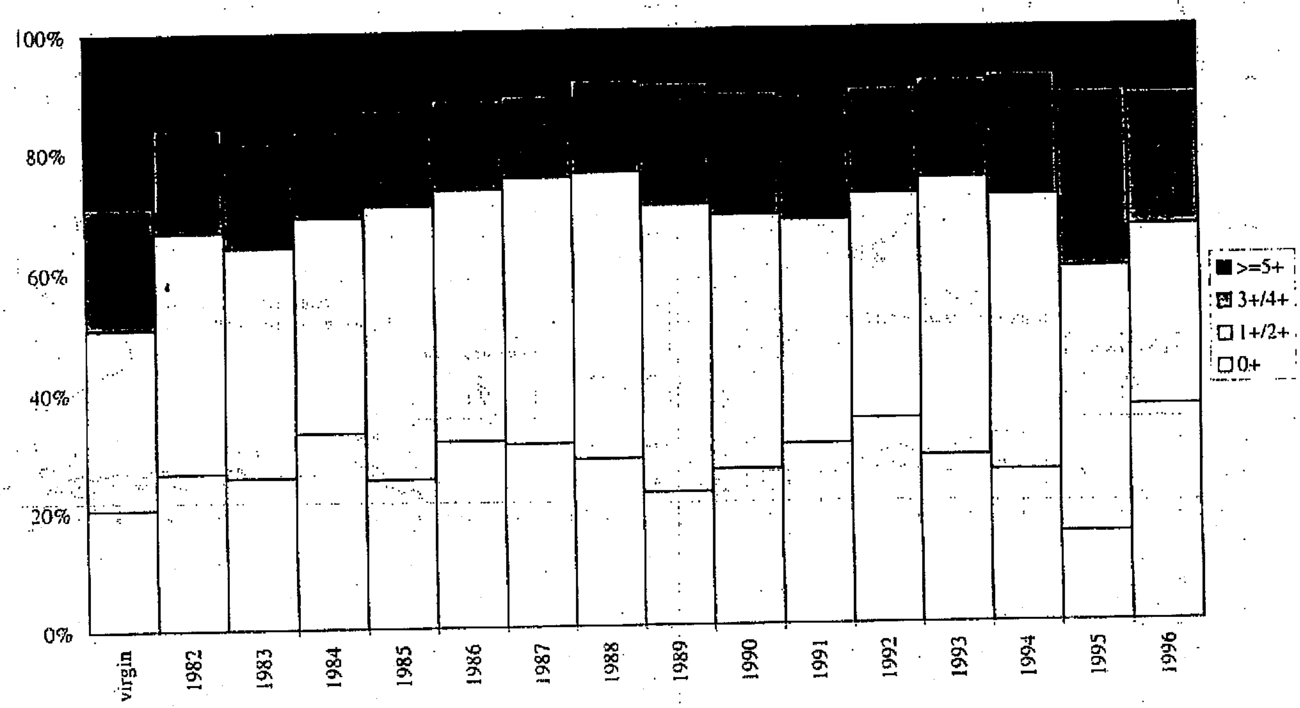
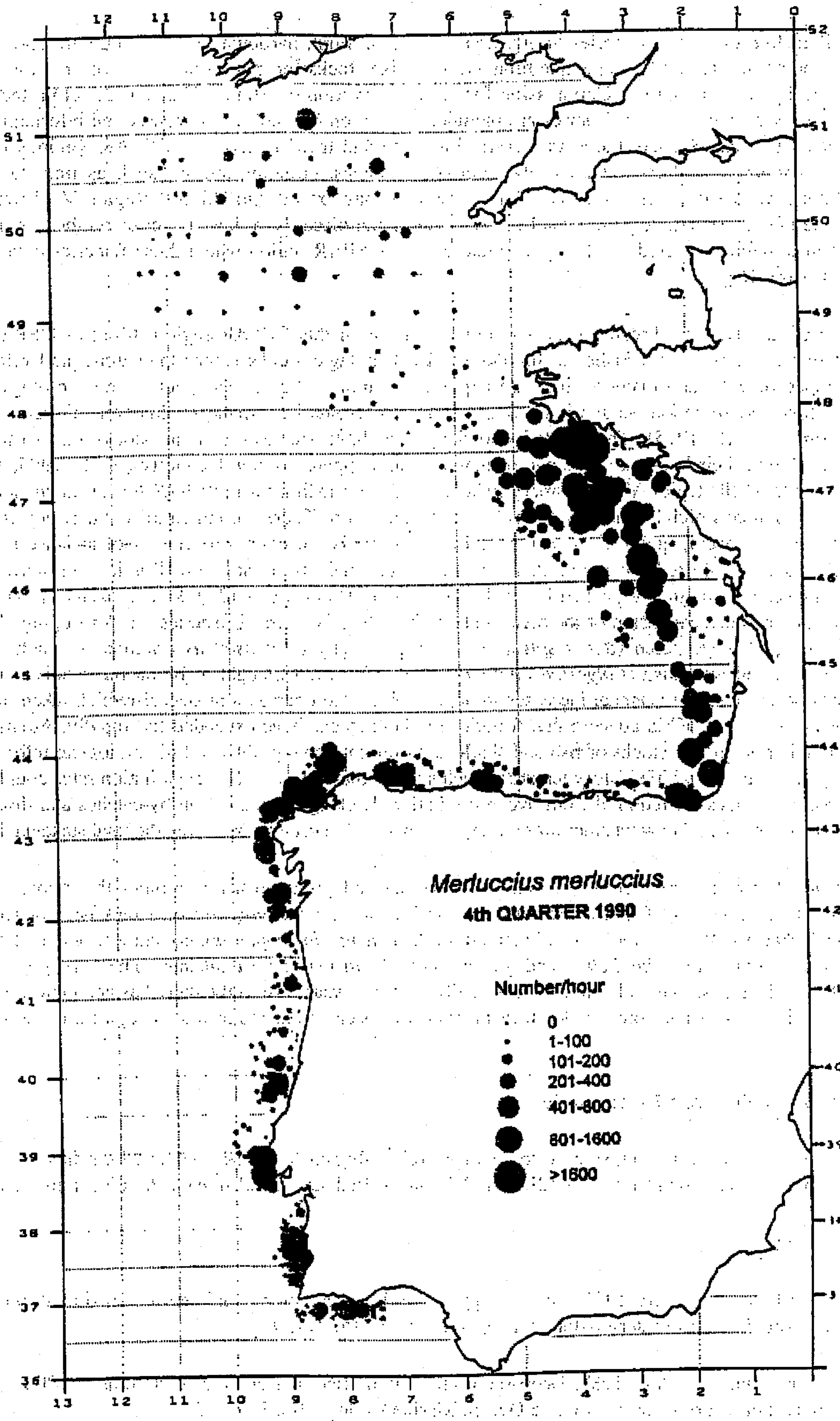


Figure 5.4.5.2.3.1. Hake distribution observed in the fourth quarter of 1990.



5.5 OSPAR REGION V

5.5.1 Description of the Fisheries

WGECO identified four classes of fisheries in this OSPAR region. First, particularly in the southern and central areas, fisheries have been prosecuted on large pelagic tuna and tuna-like stocks for many years. Advice on these fisheries and stocks is provided by the International Commission for the Conservation of Atlantic Tunas (ICCAT) rather than ICES. ICCAT reports on by-catches in these fisheries are included in Section 6.5, below. Otherwise the information here does not consider these fisheries. Second, there are several long-line and trawl fisheries in OSPAR Regions I, III, and IV which are active primarily in deep waters on the continental slopes, targeting species such as ling, tusk, argentine, grenadier, and deep-water sharks. In some years, these fisheries may extend into OSPAR Region V. However, to keep the features of the major fisheries together, catches, by-catches, and discards in these fisheries on the continental slopes are included in information reported for the appropriate coastal OSPAR region where these fisheries conduct the large majority of their harvesting.

The third group of fisheries are directed fisheries prosecuted within this OSPAR region. Most of OSPAR Region V comprises distant and deep waters. Fisheries for demersal and pelagic stocks (other than tunas and related species) generally developed recently and many of them are expanding rapidly. They target species whose biology is generally poorly known. However, some knowledge is becoming available as studies expand in this area (Gordon and Duncan, 1985, 1987; Gordon *et al.*, 1995). The limited knowledge available suggests that the stocks can sustain only low exploitation rates due to their longevity, late age of maturity, and apparently low fecundity (ICES, 1995, 1996). In the northern portion of OSPAR Region V, the primary fishery has been trawling for redfish (*Sebastes marinus* and *Sebastes mentella*); a large resource which has attracted rapidly increasing effort. There are indications that *S. marinus* includes a genetically distinct component, 'giant' *S. marinus*, and *S. mentella* is considered to consist of at least a deep-sea and oceanic stock. At least 13 fleets have joined this fishery, but the main fleets are from Russia, Germany, Iceland, and Norway. Redfish catches in Region V peaked in 1994 and 1995, at 94,000 t and 127,000 t, respectively. These increases have come through continued expansion of the areas and depths fished. New trawl fisheries are developing along the mid-Atlantic Ridge for golden-eye perch (*Beryx splendens*), orange roughy (*Hoplostethus atlanticus*), black scabbard fish (*Aphanopuus carbo*), and wreckfish (*Polyprion americanus*). CPUE is not thought to be an informative index of stock status for these fisheries. However, annual increases in depths fished and decreases in size of redfish taken in areas fished over several years support the ICES concern that fisheries on these stocks have expanded too rapidly. Attempts to extend these fisheries to other deepwater stocks of fish and sharks confront the same problems: little biological information on the stocks being targeted, but general life history features which suggest that only very low exploitation rates could be sustained and very long recovery times are required from over-exploitation. Lacking infomation on by-catches and discards in these deep-water fisheries, we do not know whether present levels of by-catch mortality are below the total sustainable level.

The fourth group of fisheries are traditional longline, handline and gillnet fisheries around the Azores and adjacent seamounts. These target a variety of species, featuring red seabream (*Pagellus bogaraveo*), wreckfish, conger eel (*Conger conger*), greater forkbeard (*Phycis blennoides*), bluemouth (*Helicolenus dactyopterus*), golden-eye perch, alfonsine (*Beryx decadactylus*), kitefin shark (*Dalatias licha*), and gulper shark (*Centropgorus granulosus*). The fishery is prosecuted by vessels up to 30 m in length, with total landings under 5,000 t in recent years. Recently, there has been interest in expanding these fisheries to other species and some exploratory surveys have been conducted in waters adjacent to those traditionally fished.

5.5.2 Trends in the five selected species

There are no fisheries for cod, herring, sole, hake, or mackerel in Region V. Occasional small catches of cod, hake, and mackerel taken on the eastern boundaries of ICES Areas VIb and VIIIc,k are included in OSPAR Regions III and IV.

5.5.3 References

Gordon, J.D.M., and Duncan, J.A.R. 1985. The ecology of the deep-sea benthic and benthopelagic fish on the slopes of the Rockall Trough, North-eastern Atlantic. Prog. Oceanogr., 15: 37–39.

Gordon, J.D.M., and Duncan, J.A.R. 1987. Deep-sea bottom-living fishes at two repeat stations at 2200 and 2900 m in the Rockall Trough, North-eastern Atlantic Ocean. Marine Biology, 96: 309–325.

Gordon *et al.* 1995.

6 DISCARDS

6.1 Discards in OSPAR Region I

6.1.1 Information on discards by gear type

The assessment Working Groups have provided very little information on discards in OSPAR Region I, either because the problem is supposed to be small in some fisheries or because very little data were available. Table 6.1.1.1 summarizes some of the main fisheries by ICES Sub-area/Division and the discard information provided.

Table 6.1.1.1. Information on discards by fishery/species, gear, and ICES Sub-area/Division in OSPAR Region I provided by ICES Assessment Working Groups.

Fishery/ species	Gear	Species discarded	Sub-area/Division			
			I+IIa+IIb	Va	Vb	XIVb
Atlantic salmon	Longline	Atlantic salmon	No fishery	No fishery	1.8–15.6 % of catch in 1982–1991[1]	
Capelin	Purse seine trawl		Not supposed to be any problem	Not supposed to be any problem	No fishery	No fishery
Herring	Purse seine trawl		Not supposed to be any problem	Not supposed to be any problem	No fishery	No fishery
Cod, redfish, haddock	Trawl, longline, gillnet		No data available	No data available	No data available	No data available
Shrimp	Trawl	Cod	No data available	No data available	No data available	32 t or 100.000 ind. in 1984[2]
		Others	No data available	No data available	No data available	No data available
Nephrops	Trawl	*Nephrops*		31.5 % of catch in 1996[3]		
		Others		No data available		

1) ICES, 1993.
2) ICES, 1997.
3) Unpublishd Icelandic data.

In addition to the fisheries included in Table 6.1.1, there is also known to be some discards of by-catch species in the deep-water fisheries in the region.

6.1.2 Commentary on quality of data and collection programmes

WGECO is not aware of any programmes for collecting information on discards in this region. Some Working Groups in the region, however, believe that there is some amount of discard in these areas based on observations, and from the historical data provided for tuning it may be concluded that unaccounted mortality probably has been large in periods (ICES, 1998). The Institute of Marine Research in Bergen, Norway, wanted to hire a commercial trawler in the 1980s for fishing together with the fleet, but the request was turned down. The suggestion may again be proposed due to problems with the assessment of Northeast Arctic cod.

There also exist earlier studies on by-catches and discards in a few of the fisheries in the region. Hylen and Jacobsen (1987) estimated the by-catch of cod in the Norwegian shrimp fishery north of 69° N (Sub-areas I and II). The estimates were based on commercial landing statistics, data from the surveillance of the shrimp fishery, and cod and shrimp surveys. They found that the number of 1–3-year old cod taken as by-catch in the shrimp fishery was relatively high compared to the number taken by the human consumption fishery (on average about 800 and 10 times higher for the 1- and 2-group, respectively). Since only fish of commercial size are landed from the shrimp fishery, most of the by-catch of these age groups was discarded. The long-term loss in yield was estimated at 20,000 t and 30,000 t for the 1982 (average strength) and 1983 (strong) year classes, respectively. In order to limit the by-catch of cod and haddock, areas have been closed in periods when the by-catch of undersized fish exceeded three per 10 kg of shrimp. The introduction of a sorting grid in 1993 has further reduced the by-catch problem in the Norwegian shrimp fishery.

McBride and Fotland (1996) made estimates of unreported catch of cod in the Norwegian commercial trawl fishery in the Barents Sea (Sub-area I) in 1989. They used information from the catch statistics, length samples from the landings, length samples from bottom-trawl surveys and data on cod-end selectivity for Norwegian bottom trawlers. Of the estimated total numbers caught in the area, 6.9 % was found to be discarded or not reported because they were below the minimum market size. They concluded that their estimates were conservative relative to the peak discard rates estimated for 1953–1954 (40 % by number and 20 % by weight) by Garrod (1967).

6.1.3 References

Garrod, D.J. 1967. Population dynamics of the Arcto-Norwegian cod. J. Fish. Res. Board Canada, 24(1):45–190.

Hylen, A., and Jacobsen, J.A. 1987. Estimation of cod taken as by-catch in the Norwegian fishery for shrimp north of 69° N. ICES CM 1987/G: 34. 21 pp.

ICES. 1993. Report of the Study Group on North-East Atlantic Fisheries. ICES CM 1993/Assess: 13.

ICES. 1997. Report of the North-Western Working Group. ICES CM 1997/Assess: 13. 356 pp.

ICES. 1998. Report of the Arctic Fisheries Working Group. ICES CM 1998/Assess: 2. 366 pp.

McBride, M.M., and Fotland, Å. 1996. Estimation of unreported catch in a commercial trawl fishery. Journal of Northwest Atlantic Fisheries Science, 18:31–41.

6.2 Discards in OSPAR Region II: North Sea including the Skagerrak, Kattegat and Channel

6.2.1 Herring

6.2.1.1 North Sea and eastern Channel

As outlined in Section 5.2.2.1.1, above, herring are caught by purse seine and mid-water trawls in the North Sea. Only the Netherlands has provided information on discard amounts in their herring fishery, which only forms a small part of the overall North Sea fishery (Table 6.2.1.1.1). Most of the Netherlands fishery uses pelagic trawls, so there is at present no information on discard rates from purse seines (the other main gear used in this fishery). In addition, no information is available on the practice of 'slippage'. This occurs when the purse seine is closed and pulled tight, but the fish are released from the net and not brought on board, usually because the fish are not of the desired size. Fish released in this way are usually dead or moribund.

A partially EU-funded project started in 1995 to estimate discards in all Danish fisheries including those in the North Sea; it will continue for three years. The EU have also part-funded a two-year project starting in June 1997 to place observers on board Scottish and Norwegian purse seiners.

6.2.1.2 Skagerrak/Kattegat

No estimates of discards were available for Division IIIa. Preliminary data from the Danish study described above indicate that there is very little discarding in the Kattegat. However, discarding may be at a high level in the Skagerrak, especially during summer when there is a demand for high quality herring for the Dutch market.

Table 6.2.1.1.1. Herring discards (tonnes) from the Netherlands fishery, total catches in subsections of the North Sea (tonnes), and proportion of this total landed by the Netherlands. Discard amounts were provided only by the Netherlands (from ICES, 1997a).

Area		1990	1991	1992	1993	1994	1995	1996*
IVa West	NL Discards	750	883	850	825	550	0	356
	Total catch	141,780	152,767	157,265	128,662	177,877	196,365	99,866
	NL %	21 %	19 %	19 %	22 %	9 %	13 %	3 %
IVa East	NL Discards	0	0	0	0	0	0	0
	Total catch	174,747	126,627	115,775	100,154	85,469	109,562	38,115
	NL %	0	0	0	0	0	0	0
IVb	NL Discards	2,560	1,072	1,900	245	-460	0	592
	Total catch	175,474	225,448	202,229	210,473	131,008	165,455	77,916
	NL %	16 %	12 %	13 %	15 %	30 %	18 %	24 %
IVc & VIId	NL Discards	5,350	2,662	2,200	2,400	2,400	0	521
	Total catch	61,082	60,685	73,981	84,878	74,078	62,905	49,565
	NL %	19 %	32 %	25 %	23 %	27 %	38 %	28 %
Total	NL Discards	8,660	4,617	4,950	3,470	2,510	0	1,469
	Total catch	553,082	565,527	549,249	524,020	467,534	534,281	264,868
	NL %	13 %	13 %	14 %	15 %	16 %	15 %	13 %

6.2.2 Mackerel

Mackerel are caught by purse seine and pelagic trawl in the North Sea. Only the Netherlands has provided information on discard amounts from their fishery, which is mostly by pelagic trawl. There is therefore no information on discard rates from purse seines or on the practice of 'slipping' (see Section 6.2.1, above). Discarding to land the highest grade of fish (high-grading) and slipping has been a problem in the past when large year classes arrive in the fishery. This may become a particular problem again when the comparatively strong 1996 year class arrives in the fishery.

An EU part-funded project started in 1995 to estimate discards in all Danish fisheries including those in the North Sea and will continue for three years. The EU have also part-funded a two-year project starting in June 1997 to place observers on board Scottish and Norwegian purse seiners.

Table 6.2.2.1. Mackerel discards (tonnes) from the Netherlands fishery, total catches (tonnes) in subsections of the North Sea including the Skagerrak/Kattegat, and proportion of this total landed by the Netherlands. Discard amounts were provided only by the Netherlands (from ICES, 1998a).

	1990	1991	1992	1993	1994	1995	1996
NL Discards	4,300	7,200	2,980	2,720	1,150	730	1,387
Total catch	305,100	365,900	367,164	390,558	475,980	323,400	212,838
NL %	4.5 %	1.3 %	1.8 %	2.0 %	0.8 %	0.4 %	0.9 %

6.2.3 Horse mackerel

Horse mackerel are caught by purse seine and pelagic trawl in the North Sea. Only the Netherlands has provided information on discard amounts from their fishery, which is mostly by pelagic trawl. There is therefore no information on discard rates from purse seines or on the practice of 'slipping' (see Section 6.2.1, above). The part of the western horse mackerel stock that is fished in the western Channel (VIIe) could not be disaggregated from the fisheries in waters further west than this.

An EU part-funded project started in 1995 to estimate discards in all Danish fisheries including those in the North Sea and will continue for three years. The EU is also partially-funding a two-year project starting in June 1997 to place observers on board Scottish and Norwegian purse seiners.

Table 6.2.3.1. North Sea horse mackerel discards from the Netherlands fishery, total catches in subsections of the North Sea including the Skagerrak/Kattegat, and proportion of this total landed from the area fished (IVb,c) fished by the Netherlands. No breakdown by nationality was available for IVb and IVc. Discard amounts were provided only by the Netherlands (from ICES, 1998a). Landings from the northern North Sea (IVa and some Norwegian IVb) of the western horse mackerel management stock included for information (no discard data available).

	1990	1991	1992	1993	1994	1995	1996
NL Discards			400	930	630	30	212
Total catch	146,387	78,594	118,769	148,445	114,402	107,381	86,987
% taken in NL fisheries area	11.9 %	14.5 %	11.7 %	2.6 %	2.2 %	7.4 %	20.4 %
Western stock landings from the North Sea	112,753	63,869	101,752	134,908	106,911	90,527	18,356

6.2.4 Demersal stocks

Information on demersal stocks and discard information for some fisheries targeting these stocks has been assembled by ICES working groups on the assessment of southern shelf demersal stocks (ICES, 1998b) and on the assessment of demersal stocks in the North Sea and Skagerrak (ICES, 1998c). Within the demersal fisheries, there are a number of gears catching a variety of species, and conversely, species are caught by a variety of gears. In an ideal situation, it would be most useful to report discards either by gear type for all species or by species for all gear types. Unfortunately this is not possible, and available information is available only for some national fisheries and very often only for the main target species of that fishery.

6.2.4.1 ICES Division VIIe

Table 6.2.4.1.1 compiles information from ICES (1998b) for ICES Division VIIe (western Channel). The discards in this area have been calculated by raising the sampled catches in these fisheries by the proportion discarded within nearby Irish fisheries (Connolly and Wheatley, 1997).

Table 6.2.4.1.1. Summary of information on by-catch (tonnes) in UK fleets in ICES Division VIIe (ICES, 1998b).

Country/gear type	Main target species	Landings (of main target species*)	Discard of main target species	Total discard (all species)
UK otter trawl (1995)	plaice, lemon sole, whiting	1271	179	1314
UK pair trawl (second and third quarters, 1995)	whiting	117	31.3	169
UK beam trawl (1995)	plaice	588	9	1402
UK fixed nets (1992/1993)	hake	30	0.4	16
UK fixed nets (1992/1993)	cod	93	0.2	0.2
UK fixed nets (1992/1993)	turbot/angler	100	6	29

*Note: It was unclear from the information given as to whether the landing figures related only to the main target species or to total landings.

In the UK otter trawl fishery, the main species discarded are dab, gurnard, (more than half total is these two species), lesser spotted dogfish and whiting. The discards are very variable seasonally, with more than half of the dab discard occurring in the first quarter, but when normalised for landing, by-catch ratio is particularly bad for dab in last quarter.

There was no sampling in the first and fourth quarter of 1995 in the UK demersal pair trawl fleet fishing for whiting. In other quarters of that year about 117 t of fish were landed and 169 t discarded. This was equivalent to approximately a half the year's landings. There was also very large differences in amounts discarded between two quarters recorded (68 % of total catch in the second quarter, 29 % in the third) The main species discarded were dab, lesser spotted dogfish, whiting, horse mackerel and gurnard.

The UK beam trawl fleet fishing for plaice in 1995 had a very high level of discard in the first quarter of the year. About a quarter of this was of cuttlefish; the other main species discarded were pout whiting (Q1, Q2, Q3), dab (Q1) and gurnard (Q1). Around 50 t of spider crab were discarded in the first half of the year.

Three gillnet fleets were monitored in 1992 and 1993; these gillnet fleets differ in their location and mesh-size of nets used. While the fleet targeting cod appears to have a low by-catch, this fishery has a high by-catch of porpoise (Berrow *et al.*, 1996).

6.2.4.2 Demersal stocks in North Sea and Skagerrak

Information for discards from the demersal fisheries in the North Sea is at present very poor. This is despite the fact that these fisheries form a very large part of the North Sea fishing fleet, and probably generate a very large quantity of discards (Garthe *et al.*, 1996). Several schemes are not yet published. The section introduction noted that ICES is not receiving up-to-date information from studies currently under way. It is of particular concern that data from some studies now complete are not being made available to ICES nor to the wider scientific community. We note a number of on-going studies (e.g. Project EC 95/094 of DG XIV) and look forward to being able to use their results in due course.

6.2.4.3 Discards from three German fleets

Three German fishing fleets were sampled between 1995 and 1997: the fishery predominantly for saithe in ICES Division IVa; the mixed gadoid fishery in ICES Divisions IVb and IVc; and the fishery by small beam trawlers for flatfish in Divisions IVb and IVc. Only information on saithe, cod, whiting, haddock, plaice, and sole was presented by ICES (1998c). Information on discards of other commercially important species was presented, but was not forwarded to WGECO. Discard sampling also did not continue throughout all of the years in question, thus making estimations of annual amounts less accurate than desirable.

A total of 795 t of the six commercially important species recorded were discarded by the German saithe fishery in 1996, from a total catch of 12,925 t (Table 6.2.4.3.1). The proportion discarded was comparatively high during the fourth quarter of the year and amounts of whiting discarded often exceeded amounts landed.

An estimated total of 164 t of these species were discarded from the mixed gadoid fishery in ICES Divisions IVb and IVc, from a total catch of around 7000 t (Table 6.2.4.3.2). Cod predominated in both catches and discards. Lack of discard recording scheme meant that discard amounts (and therefore total catch) had to be extrapolated from landing amounts, using the average of the discard proportions in relation to landings from quarters 3 and 4.

The German beam trawl fleet for flatfish was sampled in 1995. There were no records from the first quarter as catches are apparently landed by Dutch fishermen under a German flag at this time of year. A large amount of discards are released (7513 t) compared with amounts caught (13854 t) (Table 6.2.4.3.3). Expressed in catch per hour, the German beam trawl fishery in 1995 caught 68.7 kg/h marketable fish, and 148 kg/h were discarded. Estimated numbers of fish discarded by the German fleet alone exceeded 60 million in 1995.

Table 6.2.4.3.1. Discards of selected species from German saithe fisheries in Division IVa (from ICES, 1998c).

Quarter	Species	Discard	Catch
Q1 (1996)	Saithe	34	3205
	Cod	8	293
	Whiting	29	35
	Haddock	50	212
	Plaice	0	0
	Sole	0	0
Q2 (1996)	Saithe	70	2477
	Cod	7	275
	Whiting	9	21
	Haddock	71	407
	Plaice	0	0
	Sole	0	0
Q3 (1996)	Saithe	41	2636
	Cod	5	420
	Whiting	8	14
	Haddock	74	154
	Plaice	0	0
	Sole	0	0
Q4 (1996, scaled from 1995 results)	Saithe	259	3622
	Cod	30	695
	Whiting	67	96
	Haddock	33	363
	Plaice	0	0
	Sole	0	0
Total		795	12925

Table 6.2.4.3.2. Discards of selected species from German mixed gadoid fisheries in IVb and IVc (from ICES, 1998c).

Quarter	Species	Discard	Catch
Q1 (1997 scaled from Q3/Q4 results)	Cod	(13)	(546)
	Whiting	(2)	(19)
	Haddock	(1)	(98)
	Plaice	(0)	(0)
	Sole	(0)	(0)
Q2 (1996 scaled from Q3/Q4 results)	Cod	(60)	(2539)
	Whiting	(3)	(29)
	Haddock	(1)	(142)
	Plaice	(0)	(0)
	Sole	(0)	(0)
Q3 (1995)	Cod	48	2467
	Whiting	6	26
	Haddock	2	145
	Plaice	0	0
	Sole	0	0
Q4 (1996)	Cod	27	709
	Whiting	1	35
	Haddock	0	249
	Plaice	0	0
	Sole	0	0
Total		(164)	(7004)

Table 6.2.4.3.3. Discards of selected species from German flatfish fisheries in IVb and IVc (from ICES, 1998c).

Quarter	Species	Discard	Catch
Q2 (1995)	Cod	229	
	Whiting	36	
	Haddock	0	
	Plaice	2693	4365
	Sole	168	911
Q3 (1995)	Cod	105	
	Whiting	29	
	Haddock	0	
	Plaice	2156	4592
	Sole	51	498
Q4 (1995)	Cod	53	
	Whiting	20	
	Haddock	0	
	Plaice	1931	3213
	Sole	42	275
Total		7513	13854

6.2.4.4 Dutch pair, otter and beam trawl

Little information was available to WGECO on by-catch levels in these fleets (Tables 6.2.4.4.1 and 6.2.4.4.2). A discard sampling scheme has been undertaken, partially funded by the EU; but this was not available for inspection (van Beek, 1990). The pattern of discards is apparently very variable between seasons, making it difficult to make quantitative annual estimates of discards. However, in view of the importance of the beam trawl fleet and the known concern about discard levels, WGECO made a very approximate estimate of tonnage of discards for the three species.

Information on discards from the beam trawl fishery for flatfish in the North Sea is available in terms of percentage by numbers caught for the periods 1978–1982 and 1989–1990 by the Netherlands fleet (ICES, 1998c). Table 6.2.4.4.2 shows the percentage of catch discarded and numbers discarded per 100 fishing hours for the main species. Discards are dominated by dab, plaice, and sole. In order to convert the discard in numbers to discards by weight, we assumed an average weight of discards of dab to be 0.092 kg (the weight of a dab of length 21 cm); plaice to be 0.122 kg (l = 23 cm), and sole to be 0.106 kg (l = 22 cm). These weights are crude but reasonable approximations based on minimum landing size and size at 50 % retention. We then related the numbers discarded to the numbers landed. For plaice and sole the percentage discarded can be related to the numbers landed as tabulated in the Assessment Working Group Report (ICES, 1998c). The total weight of dab discards was estimated by multiplying the weight of plaice discards by the ratio of the number of dab discards over the plaice discards and taking account of the differences in mean discard weight. The results of the calculation (Table 6.2.4.4.3) should only be taken as a very crude indication of the discard level of the total international beam trawl fishery. Overall, the discarded weight of these three species is roughly the same as the landed weight of those species and these species represent the great majority of those being caught.

Table 6.2.4.2.1. Discards of selected species in the Dutch pair trawl and otter trawl fisheries (percentages of discards and numbers per 100 fishing hours).

Discards		No of samples	Area	Cod		Whiting		Haddock		Bib		Plaice		Dab	
				%	No	%	No	%	No	%	No	%	No	%	No
Average	1976-1990	45	North Sea	20	6179	53	27944	29	303	88	949	84	3647	84	20499
	1989-1990	10	North Sea	44	7827	80	44671	29	1101	na	459	na	4878	na	85780

Average percentages are weighted over total catch numbers

Table 6.2.4.2.2. Discards of selected species in the Dutch beam trawl fishery (percentages of discards and numbers per 100 fishing hours).

Discards		No of samples	Area	Plaice		Sole		Dab		Flounder		Whiting		Cod	
				%	No	%	No	%	No	%	No	%	No	%	No
Average	1976-1990	49	North Sea	49	29064	16	239	98	100953	81	1411	85	8508	66	2762
	1978	8	North Sea	41	16910	9	912	98	75310	100	197	93	12926	41	1591
	1979	9	North Sea	55	27101	5	908	99	103525	100	64	71	3508	49	1735
	1980	9	North Sea	58	50052	5	246	96	80832	80	1547	70	5860	71	8217
	1981	8	North Sea	59	42494	21	2125	96	92940	84	6152	81	6454	87	5520
	1982	5	North Sea	20	7503	29	5214	98	54234	0	0	86	15663	49	630
	1989-1990	6	North Sea	46	32972	22	8263	99	202118	54	640	96	12469	61	32
			Plaicebox	83	83433	8	935	99	160347	100	201	88	3852	80	10215
			Nonplaicebox	36	17489	18	2658	97	75979	81	1647	85	8953	51	1308

Average percentages are weighted over total catch numbers

Table 6.2.4.4.3. Estimate of discards of plaice, sole, and dab from the beam trawl fleet in the North Sea (see text for sources and methods of calculation) for years between 1978 and 1990.

Species	Discards (tonnes)	Catch (tonnes)
Plaice	42,000	181,000
Sole	1,600	22,600
Dab	110,000	>110,000

6.2.4.5 Danish fleets

Little information is yet available from the early days of the 3-year programme which commenced in 1995 (Table 6.2.4.5.1).

Table 6.2.4.5.1. Discards in percent of catch for selected Danish fishing fleets.

	Danish vessel groups								
	Trawlers			Seiners	Gill-netters				
	Demersal North	Demersal South	Industrial		Cod	Plaice	Turbot	Hake	Common sole
Number of hauls	7	64	75	50	110	15	25	16	3
% discard	25.5	8.0	0.0	18.7	4.8	8.2	20.1	0.0	38.2
Sprat									
Monk	<0.1			<0.1			0.2		
Nephrops									
Whiting									2.1
Dab	0.4	0.2		2.6	<0.1	2.7			34.4
Haddock		<0.1		4.3	<0.1				
Hake					<0.1				
Mackerel		<0.1					1.3		
Turbot		<0.1				0.2	4.6		
Plaice	<0.1	2.6		<0.1		0.8	0.4		
Saithe	24.7	<0.1		<0.1	0.1		<0.1		
Herring									
Cod	0.2	4.9		1.1	2.2	2.5	5.5		
Common sole							<0.1		

6.2.4.6 English fisheries

Proportions of three species (whiting, haddock, cod) discarded in English fly seine, *Nephrops*, otter and pair trawl fisheries from 1994–1997 are presented in Tables 6.2.4.6.1, 6.2.4.6.2, and 6.2.4.6.3. Unfortunately, it is impossible with the information presented to WGECO to scale these figures up to provide tonnage estimates.

Table 6.2.4.6.1. Discards rates by weight and number for whiting in selected English fisheries.

Whiting		Q1			Q2			Q3			Q4		
Gear type	Year	% by no.	% by WT	No. hauls	% by no.	% by WT	No. hauls	% by no.	% by WT	No. hauls	% by no.	% by WT	No. hauls
Fly seine	1994	No data	No data	0	95	98	15	98	97	12	No data	No data	0
Nephrops	1994	94	84	10	100	97	3	97	94	7	99	98	11
Nephrops	1995	96	90	11	No data	No data	0	No data	No data	0	94	88	16
Nephrops	1996	85	75	12	98	No data	12	No data	No data	0	93	86	17
Nephrops	1997	78	77	5	No data		0						
Otter	1994	47	29	27	64	91	28	91	83	34	5348	37	18
Otter	1995	66	58	12	26	48	33	48	35	17	48	34	15
Otter	1996	33	23	29	62	100	23	100	100	17	24	17	22
Otter	1997	19	18	22	53		64						
Pair	1994	76	59	8	39	59	12	59	46	6	92	87	6
Pair	1995	No data	No data	0	80	92	8	92	87	13	71	55	16
Pair	1996	No data	No data	0	42	60	11	60	47	8	No data	No data	0
Pair	1997	36	24	5	100		6						

Table 6.2.4.6.2. Discard rates by weight and number for haddock in selected English fisheries.

Haddock		**Q1**			**Q2**			**Q3**			**Q4**		
Gear type		% by no.	% by WT	No. hauls	% by no.	% by WT	No. hauls	% by no.	% by WT	No. hauls	% by no.	% by WT	No. hauls
Fly seine	1994	No data	No data	0	93	88	15	99	98	12	No data	No data	0
Nephrops	1994	93	81	10	21	13	3	100	100	7	97	82	11
Nephrops	1995	82	46	11	No data	No data	0	No data	No data	0	81	67	16
Nephrops	1996	78	72	12	89	79	12	No data	No data	0	65	49	17
Nephrops	1997	79	78	5	No data	No data	0						
Otter	1994	69	59	27	41	26	28	66	50	34	28	18	18
Otter	1995	No data	No data	12	38	19	33	93	84	17	85	72	15
Otter	1996	70	48	29	58	46	23	67	57	17	30	19	22
Otter	1997	74	74	22	39	36	64						
Pair	1994	100	100	8	61	50	12	74	60	6	0	0	6
Pair	1995	No data	No data	0	84	77	8	73	53	13	53	31	16
Pair	1996	No data	No data	0	39	26	11	32	21	8	No data	No data	0
Pair	1997	0	0	5	50	48	6						

Table 6.2.4.6.3. Discard rates by weight and number for cod in selected English fisheries.

Cod		Q1			Q2			Q3			Q4		
Gear type		% by no.	% by WT	No. hauls	% by no.	% by WT	No. hauls	% by no.	% by WT	No. hauls	% by no.	% by WT	No. hauls
Fly seine	1994	No data	No data	0	98	93	15	96	88	12	No data	No data	0
Nephrops	1994	30	9	10	43	8	3	18	8	7	69	41	11
Nephrops	1995	76	25	11	No data	No data	0	No data	No data	0	45	12	16
Nephrops	1996	40	8	12	34	8	12	No data	No data	0	80	41	17
Nephrops	1997	92	38	5	No data	No data	0						
Otter	1994	30	11	27	38	13	28	66	35	34	38	23	18
Otter	1995	12 data	2	12	12	6	33	47	22	17	33	14	15
Otter	1996	17	4	29	19	4	23	4	1	17	47	21	22
Otter	1997	39	19	22	22	8	64						
Pair	1994	10	2	8	45	21	12	57	32	6	18	4	6
Pair	1995	No data	No data	0	18	5	11	23	8	8	No data	No data	0
Pair	1996	No data	No data	0	18	5	11	23	8	8	No data	No data	0
Pair	1997	10	1	5	11	3	6						

6.2.4.7 Scottish demersal fleet

The Scottish discard sampling scheme in the North Sea has been in operation since 1975. This time series is used in assessments of the haddock and whiting stocks as being representative of the overall discarding practice. Tonnes discarded by Scottish demersal vessels in the North Sea are presented in Table 6.2.4.7.1; no data on overall catches were available.

Table 6.2.4.7.1. Annual estimates of discards (total biomass in tonnes) for species caught by Scottish demersal vessels in the North Sea.

Species	1988	1989	1990	1991	1992	1993
Anglerfish	64	182	172	431	125	362
Cod	2230	11,846	8338	3727	2586	5946
Common dab	8947	3528	2712	1351	1919	4360
Cuckoo ray	436	132	194	218	269	360
Grey gurnard	2030	1617	3018	2058	3380	4632
Haddock	42,289	29,742	24,569	33,027	43,709	60,714
Hake	7	4	208	41	40	16
Herring	1159	478	168	3393	95	1465
Horse mackerel	81	530	664	304	130	657
Lemon sole	2690	3388	1095	3064	1415	1005
Lesser argentine	37	50	75	212	58	419
Lesser spotted dogfish	462	178	114	295	261	734
Long rough dab	1122	425	511	1305	1187	1135
Mackerel	14	193	231	711	55	257
Megrim	130	47	79	22	43	70
Norway pout	90	110	74	9489	1778	863
Plaice	2344	1915	946	520	580	1696
Poor cod	34	29	27	70	56	54
Red gurnard	25	18	13	423	95	376
Saithe	137	863	332	1698	1533	9510
Spotted ray	21	0	0	10	0	6
Starry ray	4850	637	1821	1951	2657	2854
Whiting	26,672	27,576	25,520	32,251	24,332	32,165
Witch	668	577	192	486	246	262
Other	643	929	830	779	1021	1066

The tabulated estimates are obtained using the weighted ratio estimator under the fill-in from a study of fish discarded by Scottish demersal fishing vessels.

6.2.5 *Nephrops*

Sampling of discards from the *Nephrops* fishery in the North Sea has been comparatively good although sampling levels and strategies vary between the various grounds (Table 6.2.5.1). There was some indication available as to which fish species were discarded along with undersized *Nephrops*. There is a wide variation in amounts discarded between fisheries which is probably related to variations in gears used (UK vessels use a net with a panel that allows the escape of small fish). As with other fisheries, market demand also influences the type of fish discarded. Undersized *Nephrops* formed the bulk of most discards, but there was a wide mixture of fish discarded as well. In the Farn Deeps, whiting was the major species discarded; while dab and long-rough dab were the main species discarded in the Kattegat.

Notable *Nephrops* grounds not sampled included the Noup and Fladden Ground.

Table 6.2.5.1. *Nephrops* discards and total catches from some North sea fishing grounds (from ICES, 1997b).

Area	Tonnes	1990	1991	1992	1993	1994	1995	1996*
Skagerrak	Discards					2642	3,171	1373
	Total catch					3811	4,611	3511
Kattegat	Discards					2648	926	
	Total catch					2736	1,549	
Firth of Forth	Discards	383	245	303	553	1498	596	886
	Total catch	2294	1634	2016	2797	2938	2194	2317
Moray Firth	Discards	191	289	308	214	152	464	463
	Total catch	2287	1808	1880	2037	1756	1601	1727
Farn Deeps	Discards	1040	820	756	383	1166	530	990
	Total catch	3278	2883	2219	3413	4863	3098	3469
Botney Gut - Silver Pit	Discards			203	268	331	281	187
	Total catch			1101	1296	1326	1469	1084
Total of fisheries	Discards	(1614)	(1354)	(1570)	(1418)	8437	5968	(3899)
	Total catch	(7859)	(6325)	(7216)	(9543)	17430	14513	(12108)

6.2.6 *Pandalus* in IIIa and IVa East

ICES (1998d) records discards of *Pandalus* from *Pandalus* fisheries in the main North Sea fishery in ICES Divisions IIIa and IVa (East). These estimates were based on proportions of *Pandalus* in the Norwegian catch with a carapace length of less that 15 mm. There is no record of non-*Pandalus* discards. High grading occurs with the discard of medium-sized fresh shrimps and retention of large boiled shrimps.

Table 6.2.6.1. *Pandalus* discards and total catches in ICES Divisions IIIa and IVa (east) in the north-eastern North Sea (from ICES, 1998d).

	1990	1991	1992	1993	1994	1995	1996
Discards	1723	765	713	1340	426	642	1282
Total catch	11881	12362	13728	14059	12076	13938	15515

6.2.7 German brown shimp (*Crangon*) fishery

Walter (1997) described the amounts of discard in the brown shimp fishery off Lower Saxony. Commercial shrimp represented 11 % of the mass of the catch. The majority of the catch by weight was undersized shrimp (64 %), other invertebrates (8 %) and fish (11 %) (Table 6.2.7.1). The highest discard ratio was in August, with much lower ratios in spring and autumn. Plaice were present in all samples of discards, with herring (73 % of samples) being the next most common commercial species in the discards. These formed the majority of the fish discard by mass also.

Table 6.2.7.1. Total catch and discards (tonnes) from the German brown shrimp fishery off Lower Saxony in 1993 (Walter, 1997).

Month	Total catch	Discard shrimp	Discard fish	Other discarded invertebrates
April	2240	1155	496	67
May	3520	1969	476	550
June	3794	2741	303	264
July	4701	3614	483	147
August	7127	5289	743	572
September	8015	5888	962	387
October	6308	5166	356	111
November	2305	1754	221	17
Total	38009	27576	4040	2114

6.2.8 Other fisheries

The previous section includes a summary of all information made available to WGECO by the Assessment Working Groups. There appears to be large areas within the fisheries considered by these groups where an attempt to estimate discards has not been carried out. In addition, there are fisheries where there appears to be no assessment. Some examples are the fishery for Norway pout, gillnets in the North Sea, and seine nets.

6.2.9 References

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6.3 Discards in the OSPAR III Region

6.3.1 Herring

6.3.1.1 Herring discards in the Celtic Sea and Division VIIJ herring

The estimated herring catches from 1987–1996 for the combined areas by year and by season (1 April–31 March) are given in Tables 6.3.1.1.1 and 6.3.1.1.2, respectively. The reported catch taken during the 1996/1997 season, including the estimates of herring discards and unallocated landings, was over 17,000 t compared with 23,300 t during the previous season. The decrease was mainly due to marketing difficulties during early 1997 and a reduced level of discarding.

The level of discarding in this fishery is believed to have decreased in recent years as fishermen have become more expert in identifying suitable shoals for the Japanese roe market and in controlling the amounts of fish in their nets. Nevertheless, discards may on occasion reach a high level particularly if the fishery is allowed to remain open despite marketing difficulties, and we are not aware of specific studies on the propertion of catch lost by slippage. During the first quarter of 1997, the landings from Division VIIa (South) and Division VIIg were raised by 10 % to include discards as in previous years. The level of discards for the remainder of the season is not believed to have been significant.

Table 6.3.1.1.1. Celtic Sea and Division VIIj herring landings by calendar year (t), 1987–1996. (data from ICES, 1997/Assess:8.) Note: These figures may not in all cases correspond to the official statistics and cannot be used for management purposes.

Year	Discards (t)	Total (t)
1987	4,200	27,300
1988	2,400	19,200
1989	3,500	22,700
1990	2,500	20,200
1991	1,900	23,600
1992	2,100	23,000
1993	1,900	21,100
1994	1,700	19,100
1995	700	19,000
1996[1]	3,000	21,800

[1] Preliminary

Table 6.3.1.1.2. Celtic Sea and Division VIIj herring landings (t) by season (1 April–31 March) 1987/1988–1996/1997. (Data from ICES, 1997) Note: These figures may not, in all cases, correspond to offical statistics and cannot be used for management purposes.

Year	Discards (t)	Total (t)
1987/1988	4,000	26,200
1988/1989	3,400	20,400
1989/1990	3,600	23,100
1990/1991	1,700	18,600
1991/1992	2,100	25,600
1992/1993	2,000	21,200
1993/1994	1,800	18,600
1994/1995	1,900	19,300
1995/1996	3,000	23,300
1996/1997	600	17,400

6.3.1.2 Herring discards in Divisions VIa (North), VIa (South), and VIIb,c

The main catches in 1996 from this fishery in Division VIIa (North) were taken by the UK (Scotland), and in Division VIIa (South) were taken by Ireland who took over 95 % of the total allocated catches. The total amount of unallocated catches in 1996 was over 8,600 t which was considerably higher than that recorded for 1995.

The catches taken in this area from 1982–1996 are shown in Table 6.3.1.2.1. There were no estimates of discards reported in 1995–1996, and there are no indications that discarding is a major problem in this fishery even though substantial catches from this fishery in recent years have been taken in a 'roe' fishery. Reports, however, have been received of quantities of discarded herring taken by bottom trawlers fishing in the areas adjacent to known spawning grounds, but it has not been possible to q uantify the amounts.

Table 6.3.1.2.1. Discards and total catches (tonnes) of herring in Division VIa (North) in 1982–1996 and in Divisions VIa (South) and VIIb,c, in 1989–1996 (ICES, 1997a).

	Division VIa (North)			Division VIa (South) and Divisions VIIb,c	
Year	Discards	Misreported	Total	Discards	Total
1982			92,630		
1983			63,523		
1984		19,142	63,864		
1985		4672	38,994		
1986		10,935	71,078		28,785
1987		18,647	44,105		48,600
1988		11,763	35,516		29,100
1989	1550	19,013	33,945	1000	29,200
1990	1300	25,266	44,774	2530	43,969
1991	1180	22,079	32,388	3400	37,700
1992	200	22,593	28,888	100	31,850
1993	820	24,397	32,020	250	36,800
1994	700	30,234	24,619	700	33,900
1995		36,687	33,794		27,792
1996		56,007	26,105		32,500

6.3.1.3 Quality of data and collection programmes

Reasonably reliable data appear to be available for the quantities of herring discarded in these targetted herring fisheries. Official dadta collection programmes do not occur in all fisheries, but the results of a recent EU-funded project (EU Project BIOECO/93/17) indicated that the overall discard rate of 10 %–20 % used by previous Working Groups for the Celtic Sea and Division VIIj fishery was realistic.

6.3.2 Mackerel

In some fisheries, e.g., those in Subareas VI and VII, mackerel is taken as a by catch in the horse mackerel fisheries. Reports from these fisheries have suggested that discarding may be significant because of the low mackerel quota relative to the high horse mackerel quota - particularly in those fisheries carried out by freezer trawlers. In autumn 1997 an EU-funded programme involving Norway and Scotland commenced with the intention of studying the performance of the purse seine fisheries for herring and mackerel in OSPAR Region II. This programme will provide data on discards for these fleets. At present only one country, the Netherlands, provides information on mackerel discards but this information is not applied to any other fleets outside the region. The discarding of small mackerel may again become a problem in all areas if the 1996 year class is very strong as seems possible at present.

An EU programme carried out by Spain studied the rate of discards of all species taken by the Spanish fleets, fishing in Sub-areas VI, VII, and VIIIc. The results of this study (Perez *et al.*, 1994) showed that the discard rates varied by species, area, and fishing fleet. The observed levels of discards were between 0.2 %–25.7 % for horse mackerel, and between 0.1 % and 8.1 % for mackerel.

Table 6.3.2.1. Catches and discards of mackerel in Sub-area VI. Discards not estimated prior to 1978. (Data from ICES, 1998.)

Year	Catch	Discards
1978	166,900	15,100
1979	223,600	20,300
1980	224,700	6,000
1981	337,600	2,500
1982	344,500	4,100
1983	337,400	22,300
1984	307,700	1,600
1985	390,875	2,735
1986	104,100	
1987	183,700	
1988	118,700	3,100
1989	123,900	2,600
1990	120,600	5,800
1991	120,200	10,700
1992	151,526	9,620
1993	136,167	2,670
1994	135,728	1,390
1995	145,700	74
1996	130,150	255

6.3.3 Horse mackerel

Spain, Portugal, Ireland, Denmark, and the Netherlands have directed trawl and/or purse seine fisheries for horse mackerel. In OSPAR Region III, the Western horse mackerel stock is caught. Only one country provides data for discards. Therefore the amount of discards given in Table 6.3.3.1 is not representative for the total fishery. In the discard study described by Perez *et al.* (1994), observed levels of discards between 0.2 %–25.7 % were found for horse mackerel.

Table 6.3.3.1. Total landings and discards (tonnes) of the complete western horse mackerel fisheries. Specific landings for Divisions VIa and VIIa-c,e-k are included.

Year	Discards	Total landings	Landings Division VIa	Landings Divisions VIIa-c,e-k
1982	-	41,587	6283	32,231
1983	-	64,862	24,881	36,926
1984	500	73,625	31,716	38,782
1985	7500	80,551	33,025	35,296
1986	8500	105,665	20,343	72,761
1987	-	157,240	35,197	99,942
1988	3740	188,100	45,842	81,978
1989	1150	268,867	34,870	131,218
1990	9930	373,463	20,794	182,580
1991	5440	333,555	34,415	196,926
1992	1820	37,050	40,881	180,937
1993	8600	433,145	53,782	204,318
1994	3935	388,875	69,546	194,188
1995	2046	510,597	83,486	320,102
1996	16,870	16,870	81,259	252,823

6.3.4 Demersal trawl fisheries of OSPAR Region III

For stocks assessed by the Working Group on the Assessment of Southern Shelf Demersal Stocks (WGSSDS), a distinction can be made between discard data which are used in the Working Group assessments of particular stocks, and those resulting from discard sampling programmes in which all (or most) species in the catch have been recorded.

Tables 6.3.4.1 to 6.3.4.3 give the quantities of discards from Irish trawl fleets operating during 1996 in ICES Divisions VIIb, VIIg, and VIIj, respectively, estimated by raising the sampled catches in relation to the quarterly landed catch of target species by the corresponding fleets. A detailed description of the Irish discard sampling scheme is given in a working document (Connolly and Wheatley, 1997; WGSSDS Report 1997). A range of species are discarded depending on the area and target species. In Division VIIb, dogfish, grey gurnard, and haddock comprise 60 % of the total discards by the Irish trawl fishery (Table 6.3.4.1); in Division VIIj, dogfish, haddock, megrim, and whiting comprise 60 % of the total discards (Table 6.3.4.2); and in Division VIIg, discards are dominanted by whiting and haddock (Table 6.3.4.3).

Discard sampling is not a routine part of sampling for any of the countries contributing to the WGSSDS and it is mostly dependent on external funding.

Table 6.3.4.1. Discarding (tonnes) in the Irish otter trawl fleet operating in ICES Division VIIb during 1996 (data from ICES, 1998b). Mesh size > 80 mm.

Quarter	Q1	Q2	Q3	Q4	Total
Target species	Megrim	Megrim	Megrim	*Nephrops*	1996
Landings of target species (tonnes)	237	166	163	100	
Total discards (tonnes)	264	760	1677	270	2972

Table 6.3.4.2. Discarding (tonnes) in the Irish otter trawl fleet operating in ICES Division VIIj during 1996 (data from ICES, 1998b). Mesh size > 80 mm.

Quarter	Q1	Q2	Q3	Q4	Total
Target species	Megrim	Whiting	Whiting	Whiting	1996
Landings of target species (tonnes)	364	383	260.5	121	
Total discards (tonnes)	121	326	62	107	618

Table 6.3.4.3. Discarding (tonnes) in the Irish otter trawl fleet (OTB) and beam trawl (TBB) fleet operating in ICES Division VIIg during 1996 (data from ICES, 1998b). Mesh size: OTB > 80 mm: TBB >.

Gear	OTB	OTB	TBB	TBB
Quarter	Q2	Q3	Q2	Q4
Target Species	*Nephrops*	Whiting	Angler fish	Angler fish
Landings of target species (tonnes)	277	600	45	9
Total discards (tonnes)	71	649	7	2

Table 6.3.4.4. Estimates of quantities of fish discarded were available for the following stocks which are assessed by the Working Group on Northern Shelf Demersal Stocks (WGNSDS).

Species	Area	Fleets	Years
Whiting	VIa	All Scottish trawlers	1976–1996
Whiting	VIIa	*Nephrops* trawlers	1981–1996
Haddock	VIa	All Scottish trawlers	1976–1996

Discarding by Scottish trawlers in Area VIa is estimated by the Marine Laboratory in Aberdeen, based on an observer programme. Estimates of discarding from Area VIIa *Nephrops* fleets are obtained by analysing samples of discard material provided by skippers (Fisheries Research Centre, the Marine Institute, Dublin and Department of Agriculture for Northern Ireland, Belfast). An EC-funded programme to sample Irish vessels including those in the Irish Sea started in 1995 and will finish in 1997 (FRC, Dublin). A further EC-funded programme which commenced in 1996 and is coordinated by CEFAS, Lowestoft, includes estimation of discarding by sectors of the Northern Ireland trawl fleet previously unsampled for discards (DANI, Belfast). Other estimates of discarding are available from studies carried out by the English Sea Fish Industry Authority (Hepples, 1993; Emberton *et al.* 1995; see WGNSDS Report 1997) in the Irish Sea in 1992 and again in 1993/1994.

Table 6.3.4.5 gives the quantities of VIa and VIIa whiting and VIa haddock discarded annually by sampled fleets since 1984, and the percentage of the total catch of these fleets that was discarded. The percentage discarded by age class in 1995 and 1996 (by number) is shown in Table 6.3.4.6. More detailed information for the different Scottish trawl fleets operating in VIa is available from the Marine Laboratory, Aberdeen, but was unavailable to WGECO during this meeting.

Table 6.3.4.5. Estimates of weight of discards of whiting and haddock (tonnes) by trawl fleets operating in Divisions VIa and VIIa. Percentages of the total catches of the sampled fleets are given where available (data from ICES, 1998c).

	Whiting				**Haddock**	
Year	**VIa all fleets**		**VIIa N. Ireland *Nephrops* fleets**		**VIa all fleets**	
	Discards (t)	% of total catch	Discards (t)	% of total catch	Discards (t)	% of total catch
1993	11,855	54 %	2702	-	16,904	47 %
1994	18,964	59 %	1186	-	11,192	44 %
1995	15,944	54 %	2153	57 %	8794	42 %
1996	11,776	48 %	3494	67 %	11,826	47 %

The discard sampling carried out by the SFIA (UK) in 1993 and 1994 provided estimates of discard rates of plaice, sole and whiting by English otter trawlers, *Nephrops* trawlers, beam trawlers and anchor seines. The results are summarised below.

For plaice and sole, discarding was confined mainly to fish below the minimum landing size, and was influenced by depth and geographic location. An average of 63 % of whiting above the MLS of 27 cm was discarded. Discarding of whiting was more variable than discarding in the flatfish and was controlled mainly by marketing factors as whiting is of comparatively low value (Table 6.3.4.7).

Table 6.3.4.6. Discard rates of plaice, sole, and whiting by English otter trawlers, *Nephrops* trawlers, beam trawlers and anchor seines.

	Number of hauls sampled	**discarded (%)**	**Standard error**	**Main age range discarded**
Sole	131	8	1.2	3–4
Plaice	162	55	2	2–3
Whiting	147	88	1.9	1–3

Table 6.3.4.7. Percentage discarded is given by fleet and species below.

	Whitefish otter trawl	***Nephrops* trawl**	**Beam trawl**
Sole	5 %	<1 %	3 %
Plaice	58 %	54 %	67 %
Whiting	72 %	97 %	93 %

Data on discards of non-commercial species in the Northern Ireland *Nephrops* fleet for 1996 are shown in Table 6.3.4.8. This fleet fishes otter trawls with 70 mm cod ends. Square mesh panels have been mandatory in recent years.

Table 6.3.4.8. Discards (kg) of non-commercial species by the Northern Ireland *Nephrops* fleet fishing the Western Irish Sea in 1996 (data from ICES, 1997b).

Species	Total
Apphrodite	8
Argentine	636
Alloteuthus	7292
Bib	16,629
Mud balls	5065
Cuckoo ray	4973
Spurdog	6145
Eledone	77,986
Flounder	3158
Four bearded rockling	454
Fries goby	1933
Common goby	48
Grey gurnard	182,848
Red gurnard	35,796
John Dory	592
Lemon sole	24,832
Swimming crab	5801
Lesser spotted dogfish	62,735
Norway pout	248,089
Poggie	155
Queen scallop	824
Snake blenny	4297
Homelyn ray	3900
Spotted dragonet	4784
Solenette	130
Sprat	727
Snake blenny	3187
Thickback sole	6014
Tub gurnard	13,218
Lesser weever	181
Blue whiting	2557
Buccinum	26,016

In 1994 Ireland commenced an EU-funded programme to sample discards at sea. Estimates of discarding in 1996 for the main Irish fleets operating in VIIa and VIa are shown in Table 6.3.4.9.

Table 6.3.4.9. Estimated discards (tonnes) from the main Irish fleets operating in Divisions VIIa and VIa during 1996.

	Division VIIa			Division VIa	
Gear	Otter trawl	Twin rig	Beam trawl	Seiner	Otter trawl
Total (t)	687	433	52	162	3011

6.3.5 Nephrops

Details of the *Nephrops* sampling procedures used by different countries were given in the Report of the Study Group on Life Histories of *Nephrops* (SGNEPH) (ICES, 1996). SGNEPH looked at *Nephrops* sampling because the Working Group had identified, during its presentation of sampling levels by FU, quite wide variations in sampling levels. SGNEPH attempted to identify good practice and optimum sampling strategies. Sampling mainly targets vessels directed at *Nephrops*. While *Nephrops* landings are nearly always adequately sampled, discards have to be sampled at sea and this has resourcing implications which limit the frequency of sampling. In some sampling programmes fish discards are also sampled.

A summary of the availability of discard sampling from *Nephrops* trawlers within OSPAR Region III is given in Table 6.3.5.1. This identifies by *Nephrops* Management Area and/or Functional Unit the availability of *Nephrops*, commercial fish, non-commercial fish, and benthos discard samples from *Nephrops* trawlers. In nearly all cases, the benthos is not recorded. Only in Scotland, Spain, and Portugal are the non-commercial discards sampled. In Scotland, the fish discards are collected specifically for the ICES area based fish assessment Working Groups, and are aggregated to match fish stock assessment areas. These data are available from the WGNSDS, WGSSDS, and the Working Group on the Assessment of Demersal Stocks in the North Sea and Skagerrak (WGNSSK) and from the various pelagic working groups.

Table 6.3.5.1. Details of the discard sampling available from *Nephrops* trawlers.

ICES Division	Functional Unit	*Nephrops*	Commercial fish	Non-Commercial fish	Benthos
VIa	N Minch (11)	Y	Y	Y	N
	S Minch (12)	Y	Y	Y	N
	Firth of Clyde	Y	Y	Y	N
VIIa	Irish Sea E (14)	Y	Y	N	N
	Irish Sea W (15)	Y	Y	Y	Y
VIIb,c,j,k	Purcupine Bank (16)	Y	Y	Y	N
	Aran ground (17)	N	N	N	N
Viif,g,h	Celtic Sea (20–22)	Y	Y	N	N

Table 6.3.5.2 shows the discards of fish in the Functional Units used for *Nephrops* assessment within OSPAR III. Some of these data were collected during the course of one-off sampling programmes, e.g., the EU-funded project BIO/ECO/93/003, while other discard programmes are currently funded by the EU. By-landings, i.e., the fish landed from *Nephrops* trawlers, are readily available. While this does not quantify discards, where discard data are not available, it would indicate the commercial species most likely to be discarded if data were available.

Table 6.3.5.2. Discards and landings (in tonnes) of *Nephrops* in the Functional Units (FUs) within OSPAR Region III (ICES, 1997).

***Nephrops* North Minch (FU11)**

Year	Discards	Landings
1990	252	2201
1991	388	2440
1992	312	3220
1993	27	2920
1994	1541	3410
1995	741	3166
1996	267	2341

***Nephrops* South Minch (FU12)**

Year	Discards	Landings
1990	639	4202
1991	109	3998
1992	343	3819
1993	529	3891
1994	414	4160
1995	398	3988
1996	384	3430

***Nephrops* Firth of Clyde (FU13)**

Year	Discards	Landings
1990	395	2745
1991	215	2697
1992	85	2471
1993	240	2866
1994	309	2180
1995	548	3680
1996	555	3681

***Nephrops* Irish Sea East (FU14)**

Year	Discards	Landings
1991	859	461
1992	495	151
1993	618	217
1994	514	39
1995	504	33
1996	450	36

***Nephrops* Irish Sea West (FU15) (Northern Ireland fleet)**

Year	Discards	Landings
1991	217	6024
1992	731	5112
1993	892	5355
1994	667	5841
1995	839	5401
1996	711	5601

***Nephrops* Irish Sea west (FU15) (Ireland fleet)**

Year	Discards	Landings
1991	817	3371
1992	546	2370
1993	739	2715
1994	354	1768
1995	743	3247
1996	631	2255

***Nephrops* Irish Sea west (FU15) (Total International Catch)**

Year	Discards	Landings
1991	1034	9395
1992	1277	7482
1993	1631	8070
1994	1021	7609
1995	1582	8648
1996	1342	7856

***Nephrops* Celtic Sea (FU20–22)**

Year	Discards	Landings
1991	3047	3295
1992	3874	4165
1993	3436	4586
1994	4237	5130
1995	4555	5922
1996	3921	4889

It should be remembered that the exploitation pattern generated on fish by *Nephrops* trawls is likely to be quite different from that generated by directed fin-fish vessels. The *Nephrops* permitted mesh size is considerably smaller than that permitted for fish. In Regions 1 and 2, the *Nephrops* mesh size is 70 mm, while the fish mesh size ranges from 80 to 100 mm. In Region 3, the *Nephrops* mesh size is generally 55 mm, and the fish mesh size is 65 mm. Sampling programmes looking at fish discards need to ensure that the *Nephrops* vessel stratum is sampled. These smaller mesh sizes are only permitted if certain catch composition conditions are met. EC Council Regulation 3094/86 specifies for Regions 1, 2, and 3 that a minimum of 30 % by weight in the retained catch must be *Nephrops*, and that the proportion of protected (Annex II) species must not exceed 60 %. In the UK, national technical measures specify that square mesh panels of a mesh size of 80 mm (75 mm in Division VII) must be fitted to *Nephrops* trawls. Square mesh panels allow small fish, particularly whiting and haddock, to escape before reaching the codend, and significantly reduce the quantities of small fish which have to be discarded. Quantities of the major discard species in the Northern Ireland *Nephrops* fleet are shown in Table 6.3.5.3.

Fish discarding from *Nephrops* trawlers is subject to a wide range of factors. While technical measures, such as by-catch limits and minimum landing sizes, determine discarding practice, other factors also come into play, such as the market demand for certain species and the seasonal nature of the *Nephrops* fisheries. In the eastern Irish Sea (FU 14), whiting which exceed the minimum landing size can be discarded through lack of market demand for this species, while other species like plaice and sole are carefully sorted and only undersized fish are discarded. Some *Nephrops* fisheries are carried out in the winter, others mainly in the summer. The fish species composition can differ significantly with the seasons. Estimates of the fish discards need to be weighted by the seasonally directed *Nephrops* fishing effort.

Table 6.3.5.3. Estimated quantities of fish and invertebrates discarded in the Nortern Ireland *Nephrops* fisheries in 1996 (data from ICES, 1997).

Species		Tonnes discarded
Whiting	*Merlangius merlangus*	2494
Dublin Bay Prawn	*Nephrops norvegicus*	711
Norway pout	*Trisopterus esmarkii*	248
Dab	*Limanda limanda*	211
Plaice	*Pleuronectes platessa*	206
Starfish	*Asteroidea*	194
Grey Gurnard	*Eutrigla gurnardus*	183
Haddock	*Melanogrannes aeglefinus*	139
Brown Crab	*Cancer parangus*	107
Horse Mackerel	*Trachurus trachurus*	97
Poor Cod	*Trisopterus minutus*	87
Curly Octopus	*Eledone cirrosa*	78
Common Dragonet	*Callionymus cirrosa*	65
Long Rough Dab	*Hippoglossoides platessoides*	63
Lesser Spotted Dogfish	*Scyliorhinus canicula*	63
Other species		297
Total		5742

6.3.6 Other species

6.3.6.1 Deep-water fisheries

There have been relatively few discard studies in the deep-water fisheries of the ICES area. One Irish study which compares the discards of trawl and long-line fisheries in ICES Sub-area VIa was published in 1996 by Connolly and Kelly, and new data are being collected from commercial vessels by observers from several countries. (Catch and discards from experimental trawl and long-line fishing in the deep water of the Rockall Trough. J. Fish. Biol. Supplement A, 49: 132–144.)

Other discard studies are being carried out as part of the EC FAIR Project (95/655) *Developing deep-water fisheries: data for their assessment and for understanding their interaction with and impact on a fragile environment.* The Marine Laboratory (UK) has undertaken two trips on Scottish commercial vessels to observe discards of deep-water species in ICES Sub-area VIa. These data are still in a raw format and a preliminary analysis is not yet available. French studies estimate the discards of *Coryphaenoides rupestris* from Sub-areas VI and VII as 34 % of the landings (V. Allain, unpubl.). Norway has collected some data from the Reykanes Ridge in 1996 and further sampling is planned for 1997. These data will be available in due course.

6.3.6.2 Marine mammals

A programme to assess the cetacean by-catch in the Irish and UK set gillnet fisheries in the Celtic Sea was conducted from 1992–1994 using volunteer observers. Observers were present for the hauling of over 2500 km of net which caught 43 harbour porpoises and four common dolphins. The by-catch rate was 7.7 porpoises per 10,000 km hour of net immersion. The estimated total annual by-catch of 2200 porpoises (95 % c.l. 900–3500) is 6.2 % of the estimated number of porpoises in the Celtic Sea and there is serious cause for concern about the ability of the population to which they belong to sustain this level of by-catch (Tregenza *et al.*, 1997).

Eleven fisheries were investigated in a study of the by-catch of marine mammals in pelagic trawl fisheries of the North-east Atlantic. In one fishery (Irish herring trawling), four grey seals were caught in approximately 100 hours of trawling. Eleven different pelagic trawl fisheries (Dutch horse mackerel, French tuna, French hake, French sea bass) caught a total of 18 dolphins during a total of 1300 hours of trawling. The extent of observation and the number of observed by-catches was insufficient to make a reliable estimate of overall catch rates by gear type, but the average total catch rates were between 1.1 to 1.5 cetaceans per 100 hours of trawling (Morizur *et al.*, 1997).

A number of other studies in the North-east Atlantic were identified by WGECO as containing important information on discards, but were not seen during the meeting (e.g., Goujon *et al.*, 1993a,b; Goujon *et al.*, 1996; Antoine *et al.*, 1997).

6.3.7 References

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6.4 Discards in OSPAR Region IV

There have been few published studies of discard practices in this region; the largest was an EU partially-funded project (EC DGXIV PEM/93/005) reported on by Perez *et al.* (1995). This was reviewed by WGECO at its last meeting (ICES, 1995), but unfortunately the report was not available at this meeting. A part of these data were made available for review by Sánchez *et al.*, Olaso *et al.* (1996) and the information in Perez (1996) which relates to this region.

Discards of target species in the trawl fishery can reach over 60 % of the catch (Table 6.4.1). These levels of discarding are in broad agreement with the levels reported by Perez *et al.* (1996) for Area VII, those data show that typically 50 % of the haul may be discarded (Table 9 in Perez, 1996).

Table. 6.4.1. Catches and discards (kg per hour fishing) of the principal retained species in the Spanish trawler fleet in ICES Division VIIIc West in 1997 (first quarter). (From Sánchez *et al.*, 1997.)

	Trawls			Pair Trawls		
	Catch	Discards	Discard / total catch	Catch	Discards	Discard / total catch
Four-spotted megrim	1.60	0.40	0.25			
Megrim	1.80	0.80	0.44			
Angler (*L. budegassa*)	4.10	0.20	0.05	0.30		
Angler (*L. piscatorius*)	1.50			0.20		
Hake	17.30	8.60	0.50	44.00	10.60	0.24
Blue whiting	62.10	37.20	0.62	59.10	1.20	0.02
Mackerel	56.80	0.40	0.01	0.10		
Horse mackerel	39.00	0.10	0.01	0.10		
Nephrops	0.60					

Nearly all non-target species caught in the Spanish trawl fishery in area VIIIc are discarded at a rate of 100 % (Table 6.4.2) (Olaso, 1996).

Table 6..4.2. Fauna caught and discarded (kg per 100 fishing hours) by the Spanish fleet in ICES Sub-area VIIIc in 1994 (from Olaso, 1996).

TAXA	Catch	Discards	Discards/Catch (%)
CRUSTACEA			
DECAPODA			
Anomura			
Munida undetermined	138	138	100
Paguridae undetermined	1	1	100
Pagurus alatus	8	8	100
Pagurus prideauxi	33	33	100
Brachyura			
Liocarcinus depurator	41	41	100
Marcropipus tuberculatus	4	4	100
Polybius henslowi	83	83	100
Decapoda undetermined	7	7	100
Macrura			
Polycheles typhlops	1	1	100
Natantia			
Chlorotocus crassicomis	1	1	100
Dichelopandalus bonnieri	3	3	45
Plesionika heterocarpus	1	1	100
Processa spp.	1	1	100
Solenocera membranacea	4	4	100
MOLUSCA			
BIVALVIA			
CEPHALOPODA			
Decapoda			
Alloteuthis spp.	2	2	100
Rossia macrosoma	10	10	100
Sepiidae undetermined	4	1	100
Octopoda			
Octopoda undetermined	2	1	26
CNIDARIA			
Anthozoa undetermined	71	71	100
POLYCHAETA			
Aphroditae aculeata	2	2	100

Table 6.4.2. Continued.

TAXA	Catch	Discards	Discards/Catch (%)
FISH			
ANACANTHINI			
Antonogadus macrophtalmus	1	1	100
Gadiculus argenteus	135	135	100
Micromesistius poutassou	2775	530	23
Trisopterus luscus	96	4	4
Trisopterus spp.	3	2	100
Merluccius merluccius	473	16	3
GOBIODEI			
Callionymus undetermined	10	10	100
Argentine sphyraena	26	12	45
Sardina pilchardus	1	1	100
MYCTOPHOIDEI			
Myctopoidei undetermined	2	2	100
NOTIDANOIDEI			
Scyliorhinus canicula	931	225	990
PERICOIDEI			
Trachurus trachurus	578	10	36
Cepola rubescens	16	16	100
Trachinus draco			
PLEURONECTOIDEI			
Amoglossus laterna	5	5	100
Lepidorhombus boscii	119	31	26
SCOMBROIDEI			
Scomber scombrus	301	20	26
ZEOMORPHI			
Capros aper	63	63	100

There is no information available on discards in the southern (Portuguese) part of Division IXa. The bottom trawl fisheries may discard undersized hake, horse mackerel, sardine, and *Nephrops* as these species have a minimum landing size.

6.4.1 References

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Sánchez, F., Olaso, I., and Goñi, R. 1997. Changes in bottom trawl survey catch rates in dogfish (*Scyliorhinus canicula*) and thornback ray (*Raja clavata*) in the Cantabrian Sea between 1983–1997. WGECO Working paper.

Table 6.4.2. Fauna caught and discarded (kg/100 fishing hours) by the Spanish fleet in ICES Sub-area VIIIc in 1994 (from Olasso, 1996).

6.5 OSPAR Region V

There are no discard data reported for fisheries on groundfish in OSPAR Region V. Recent studies have recorded by-catch in two trips for redfish in this Region, using midwater pelagic trawls, but these data are in preliminary form, and were not provided to WGECO. Due to the very low intensity of coverage of this study to date, these data should not be considered representative of by-catch and discarding in this fishery. Within a couple more years the existing programmes may be able to provide some picture of the incidental catches in this fishery.

Species discarded in the Azores line and gillnet fisheries include various scabbard fish (*Aphanopuus sp.*), skates, greater forkbeard, morid cod, anglerfish (*Lophius piscatorius*), and rabbit fish (*Chimaera monstrosa*). There are no quantitative estimates of amounts of discards of any of the these species.

Particularly after its 1994 annual meeting, ICCAT has sponsored programmes to monitor by-catch and discarding in fisheries for tunas and tuna-like fish. Their reporting regions do not correspond directly with the OSPAR Regions, and ICCAT (1996) contains only lists of species and qualitative descriptions of by-catch. Moreover, for the 1996 report, responses to the ICCAT inquiries about by-catch had been received by fewer than 20 of 95 possible fisheries, so coverage is incomplete everywhere, including OSPAR Region V. Results of these on-going studies will be reported in future ICCAT annual reports. Based on this preliminary information, main species taken as by-catch in fisheries for tuna and related species in the North Atlantic are sharks, particularly blue and mako shark, and there is some catch of sea turtles and seabirds in these fisheries.

6.6 Discussion of the Information on Discards

Discards and by-catch clearly constitute one of the major ecosystem effects of fishing; sometimes even potentially greater that the direct harvest of the target species. The quantity and quality of data on discards needs to improve, in order to enable evaluation of the ecosystem effects of fishing and to increase the reliability of present single and multi-species assessments. Two types of improvements are needed. More programmes must be implemented to quantify discards, and the data which are collected need to be handled better. Instituting additional discard monitoring programmes can have major cost implications, particularly because the programmes must be scientifically credible. Cost is not a major factor in improving the quality of handling and presenting data which have been collected.

The requirements for scientifically sound programmes for data collection and handling on cetacean by-catches have been reviewed in depth by Northridge (1996). This author recommends a number of specific practices which are generally applicable, and not just specific to monitoring of by-catch of cetaceans. These practices take note that:

- Self-reported magnitudes of by-catches are often unreliable, and rarely credible.
- Except for non-target species whose populations are routinely reported as numbers instead of weight (seabirds, marine mammals), by-catch and discards should be reported in absolute weight (or percent of catch, if the catch is recorded in weight on the same record), not in numbers.
- Gear and units of effort associated with by-catch and discards should always be reported.
- Position information should always be reported.
- By-catch and discards should be disaggregated to species.
- If estimates are based on a sample of the catch, the sampling fraction and strategy should be reported. The strategy for selecting the subsample should be unbiased.
- Coverage of the progamme relative to the whole fishery should be reported, as should the scheme for allocating coverage. Coverage also should be wide enough, and apportioned in an unbaised manner among vessels; such that the data are representative of the fishery. (These last two practices are statistical design issues, and should be dealt with empirically).
- Questionaires do not work.

This is not an exhaustive list of features required by a programme to collect data on by-catch and discards, but it highlights some major considerations. If WGECO is to conduct credible investigations in this important area of ecosystem effects of fishing, the requirement for these programmes must be given priority by Member Countries of ICES, and by the management agencies it advises.

In this context, WGECO notes with concern that interm data and, in a few cases even reports of completed studies, are not making their way to ICES, and are not being taken note of by assessment working groups. Now that many national laboratories and agencies are implementing programmes to quantify this important aspect of the effects of fishing, ICES should be making full use of the information being collected.

Iceland and Norway have both introduced special regulations to reduce the numbers of discards. Unfortunately, they have no discard monitoring and sampling programmes to evaluate the effects of these regulations. **WGECO recommends the collection and analysis of data which would make it possible to evaluate the effectiveness of these management measures.**

At this meeting two points became clear with regard to how ICES deals with data on discards. First of all, the information ICES has and uses on discards is so incomplete that it compromises both the quality of some analytical work, and the reputaiton of ICES as a center of knowledge on fisheries in the North-east Atlantic. Second, there are diverse perspectives on discard issues, and the diverse perspectives need to be brought more fully into the discussions ICES has on discards.

Therefore, WGECO reccommends that a Study Group be formed with Terms of Reference to review and provide guidance for how ICES and its Working Groups address the discard issue.This Study Group must have membership both from assessment working groups and from the wider environmental community.

7 CHANGES IN ABUNDANCE OF NON-TARGET SPECIES

7.1 OSPAR Region 1

Information on trends in abundance of non-target species in OSPAR Region I are only available separately for the different shelf and fishing areas within this region, especially for the Barents Sea and for the shelf areas around Iceland and Greenland.

All three studies presented represents boreal systems within OSPAR Region I.

7.1.1 East Greenland case study

Before the collapse of the cod stock, the fishery was mainly carried out by large stern trawlers and factory ships, which used otter trawls equipped with very heavy ground gears to protect net damages because of the rough fishing grounds. The by-catch was mainly discarded, except for commercially valuable species like catfish.

Since 1992, the shrimp fishery has expanded to all traditional fishing areas off West Greenland and to a lesser extent off East Greenland (Hvingel *et al.*, 1996a, 1996b). This fishery uses nets with smaller mesh openings than the traditional cod fishery. Data of by-catches and discards are not available, but it seems likely that this increasing fishery on shrimps could influence the recovery of the cod stocks negatively and could also be responsible for a high amount of discards.

7.1.1.1 Description of information source

Rätz (1997) derived abundance indices for non-target species in this area from annual groundfish surveys covering the shelf areas and the continental slope of West and East Greenland. These surveys were primarily designed for the assessment of cod commencing in 1982 and carried out by the German FRVs 'Walther Herwig II and III'. A standard fishing gear and a standard survey design were used to make the catch data comparable over the time period.

The data set was not available to WGECO and could not be further evaluated during the meeting. Therefore, this chapter only deals with the changes of non-target species already described by Rätz (1997).

7.1.1.2 Results and interpretation

<u>Long rough dab (Figure 7.1.1.2.1)</u>

In comparison with the mean indices of the 1980s, the most recent estimates of abundance increased off East Greenland by a factor of 2.7, whereas the long rough dab became less abundant in West Greenland waters. A similar reversal was observed for the geographical distribution pattern of the stock. In the early 1980s, only one-tenth of the total stock was distributed off West Greenland. The stock off East Greenland has recovered and these waters are now inhabited by the largest portion of the combined stock.

<u>Catfish (Figures 7.1.1.2.2. and 7.1.1.2.2.3)</u>

The abundance indices of both common and spotted catfish show comparable trends over the time period, but the trend is more pronounced for spotted catfish than for common catfish. Both species show an increasing trend off East Greenland and a decreasing trend off West Greenland, in combination with a geographical shift of the main part of the stock from West to East Greenland.

<u>Starry ray (Figure 7.1.1.2.4)</u>

The abundance of starry ray shows no clear trend over the observed time period like the other 3 non-target species. The abundance off East Greenland is more or less stable, but the dramatic decrease in abundance within West Greenland waters should be a reason of concern.

In summary, there is an increasing trend in abundance for these four non-target species in East Greenland waters. Climatic changes should not be excluded as possibly influencing this development, but it seems more obvious that the collapse of the cod fishery in 1992 and the low fishing effort during the following years have influenced these positive trends.

7.1.2 Barents Sea as a case study

This study presents results from the Norwegian bottom trawl survey in the Barents Sea (Jakobsen *et al.*, 1997) (see also the short description of the survey in Section 3). Most of the results shown here have not been presented before. The study is separated into two parts. One, for the commercially exploited species mainly caught as by-catch in fisheries targeted at other species and, in addition, long rough dab and polar cod. The other one adresses species of no or little commercial value.

7.1.2.1 Species with commercial interest

Greenland halibut

The fishery on Greenland halibut increased in the mid-1960s by the introduction of international trawlers. Landings reached as high as 80,000 tonnes in the early 1970s. Since 1992, the fishery has been regulated by allowing only direct fisheries from long-line and gillnetters. Trawl catches are limited to by-catch only. The maximum percentage of by-catch as a percentage of Greenland halibut onboard at any time was set to 10 % initially, but this was reduced to 5 % in 1994. But still the by-catch from trawlers constitutes the bulk of total landings. The survey results presented in Figure 7.1.2.1.1.a and 7.1.2.1.1.b covers only a fraction of the Greenland halibut stock and the results are probably dominated by a higher proportion of young fish. Within the survey area, the abundance seems to have increased until 1989–1990 and has decreased since then. Observed mean length shows the opposite trend with a minimum in 1990 and an increase after 1990.

Spotted Catfish and Catfish

The two stocks are not assessed within the ICES system. There exists a small direct fishery by long-liners but it is reasonable to believe that most of the catches are taken as by-catch by commercial trawlers. The common catfish are more abundant closer to coastal areas in the west. The species shows a remarkable similarity in trends both for abundance indices and mean length. Both species reached a minimum in abundance in 1988 and have increased since then. And both species show increased mean length in recent years.

Halibut

The stock size is at a very low level and observed variations both in abundance indices and mean length can be explained by sampling variability.

Long Rough Dab

This species has only been targeted in experimental fisheries by international trawlers. The by-catch of long rough dab is expected to be quite low. Long rough dab is quite abundant in the Barents Sea and the trends shown in Figure 7.1.2.1.1.a demonstrates some of the dynamics of the stock. A minimum abundance index of 147 million was observed in 1987 while the maximum of 944 million was observed in 1993. The trend in abundance is related to spatial distribution. Only 60 % percent of the stock was observed in the eastern subarea (D) in 1987–1988 while as much as 90 % of the stock was observed there in 1993.

Polar Cod

This species have been targeted by Russian trawlers in some periods. The survey covers only a relative small proportion of the stock and most of the observed variations can very well be related to changes in geographical distribution.

7.1.2.2 Species with no or little commercial value

Trends in abundance indices for 12 species or species groups are presented in Figure 7.1.2.2.1. All the species show large variations. Most of the species seem to have some kind of peak around 1990 which coincides with very high recruitment for cod and haddock that year. And some of the species seem to reach an overall maximum for the period in 1995 or 1996. Some of the species show a low abundance for the period 1987–1988 when the overall temperature in the water mass was at low levels. Whether the described variations are due to environmentally induced changes in distribution, catchability or availability to the sampling trawl or to changes in the effort in the trawl fleets operating in the Barents Sea is not known.

7.1.3 References

Hvingel, C., Siegstad, H., and Folmer, O. 1996a. The Greenland fishery for northern shrimp (*Pandalus borealis*) in Davis Strait in 1995 and January–October 1996. NAFO SCR Doc., 96/102, Ser. No. N2806:1–29.

Hvingel, C., Siegstad, H., and Folmer, O. 1996b. The Greenland fishery for northern shrimp (*Pandalus borealis*) in Denmark Strait in 1995 and January–October 1996. NAFO SCR Doc. 96/117, Ser. No. N2814:1–24.

Jakobsen, T., Korsbrekke, K., Mehl, S., and Nakken, O. 1997. Norwegian combined acoustic and bottom trawl surveys for demersal fish in the Barents Sea during winter. ICES CM 1997/Y:17. 26 pp.

Rätz, H.-J. 1997. Structures and changes of the demersal fish assemblage off Greenland and trends in near bottom temperature, 1982–1996. NAFO SCR Doc. 97/5, Serial No. N2830.

7.2 OSPAR Region II

Several recent studies have published trends in the abundance of fish species which are not the target of specific fisheries. We deal with each study individually and then provide an overall summary.

7.2.1 Rijnsdorp *et al.* (1996)

This study compared the standardized catch rates of a suite of species taken by five gears at two different time periods, an early period (1906–1909) and a more recent period (1990–1995). For our purposes, we compared the catch rates for the two most comparable gears used in the different periods, a 20′ otter trawl (40 mm mesh) used in the early period and the GOV trawl (20 mm mesh) used more recently (Table 7.2.1.1). Of 19 non-target species for which trend data were available, 18 species appear to have decreased in abundance. In making these comparisons, the potential effects of using different fishing gears must be kept in mind. This is particularly emphasised by the relative catch efficiencies of the two gears. The GOV trawl is approximately 40 times more efficient than the 20′ otter trawl, so all the catches in the earlier period have been multiplied by 40 to make them comparable with catches in the later period. However, to counter this the mesh size used between 1990 and 1995 was half that used in 1906–1909, thus greater numbers of the smaller fish should have been taken by the gear in the later period.

Table 7.2.1.1. Changes in catch per hour between 1906–1909 and 1990–1995 (Rijnsdorp *et al.*, 1996).

Species	1906–1909	1990–1995	Direction of trend
Spurdog	10.0	0.1	Decreased
Thornback ray	2.8	<0.05	Decreased
Poor cod	21.6	6.8	Decreased
3 bearded rockling	1.2	<0.05	Decreased
5 bearded rockling	11.2	0.2	Decreased
Tub gurnard	1.2	0.6	Decreased
Grey gurnard	90.0	13.3	Decreased
Red gurnard	0.4	<0.05	Decreased
Bull rout	0.8	0.2	Decreased
Hooknose	2.4	1.1	Decreased
Red mullet	0.0	1.0	Increased
Lesser weever	808.8	9.4	Decreased
Greater weever	178.0	0.0	Decreased
Dragonet	74.0	1.7	Decreased
Scaldfish	186.8	0.1	Decreased
Long rough dab	96	0.3	Decreased
Common dab	975.2	176.8	Decreased
Lemon sole	0.4	0.2	Decreased
Solenette	457.6	0.5	Decreased

7.2.2 Heessen and Daan (1996)

This study analysed long-term trends, over the period 1970–1993, in the International Bottom Trawl Survey in ten non-target North Sea fish species (Figure 7.2.2.1). Clear increasing trends are apparent in six of these species, starry ray, poor cod, grey gurnard, bull rout, long rough dab, and lemon sole, over the entire period, while common dab shows an increasing trend from 1982. Variation in the abundance of bib and four bearded rockling has been highly variable, but no strong temporal trend is apparent. Variations in the abundance of spurdog has been dominated by the exceptional peak in abundance in 1978. From 1970 to 1976, and from 1981 to 1993, abundance of this species appears to have fluctuated at relatively low levels.

7.2.3 Heessen (1996)

Time series data for forty fish species sampled by the International Bottom Trawl Survey in the North Sea over the period 1970–1993 are given in this study. From this, time series data for a further 26 non-target species can be extracted (Figure 7.2.3.1). Many of these species appear and disappear from the data sets periodically through the time period. For some species, however, clear trends are apparent.

7.2.4 Walker and Heessen (1996)

This study specifically examines trends in the populations of some of the skate and ray species in the North Sea. Data from the February IBTS from the period 1970–1993 for six species of ray are presented (Figure 7.2.4.1). Data is also presented indicating changes in the amount of skates and rays landed from various parts of the North Sea (Figure 7.2.4.2). Few trends are obvious over this time period, with the exception of the steady increase in catch rates of starry ray. This species is one of the few skates and rays with no commercial value. It is invariably discarded when caught. The most heavily targeted ray, the thornback ray, has all but disappeared in the southeastern North Sea (see Section 7.2.1).

7.2.5 Greenstreet and Hall (1996)

This study of long-term changes in the groundfish species assemblage structure in the northwestern North Sea presents species abundance data over the period 1929–1993 for ten species identified as either typifying the species assemblages of three sub-areas, or discriminating between them. Seven of these are non-target species (Figure 7.2.5.1). The data analysed was the Scottish August Ground Fish Survey for the period 1980–1993. The earlier data (1929–1953) were collected using the same trawl gear by Scottish fisheries research vessels over the months July to September in each year. Catch rates were adjusted to take into account the differing trawl speed of the various vessels used. The data suggest that variability in the abundance of non-target species was much greater from 1929–1953. Long-term trends in the abundance of several of the non-target species is also apparent. Spurdog were more abundant in the early period, and still seem to be declining during the later period. On average, common dab and long rough dab were no more abundant in the 1980s than in the period 1929–1953, however, common dab abundance has increased steadily throughout the later period. Lemon sole and Norway haddock are more abundant now than they were during 1929–1953, while grey gurnard are scarcer now than in 1929–1953, although their abundance has increased steadily from 1980–1993.

7.2.6 Rogers and Millner (1996)

This study presents data on changes in the abundance of eight non-target fish species over the period 1973–1995 for three separate regions off the southeast coast of England. The area divisions are shown in Figure 7.2.6.1, while the trends in abundance are given in Figure 7.2.6.2. The data indicate considerable fluctuations in abundance and few consistent trends. In recent years, eelpout appears to have declined in all three areas, while sea snail numbers have increased in area 3. Changes in abundance do not seem to correlate closely between areas.

7.2.7 Corten and van den Kamp (1996)

This study provides trends in abundance data for 12 species from 1970 to 1994 in the southern North Sea (corresponding to ICES Roundfish Areas 5 and 6). The data analysed come from the International Bottom Trawl Survey. Again, considerable fluctuations in abundance were apparent, but few obvious long-term trends were obvious (Figure 7.2.7.1). The exception to this was the clear increase in the abundance of lesser weever since 1989.

7.2.8 Conclusions

A considerable amount of effort has been spent collecting groundfish data over many years and analysing these for time series trends. Few trends are apparent and these are rarely consistent for any particular species over different areas and between different studies. It is rarely possible to relate changes in the abundance of particular species to changes in the fishing regime. However, some concensus across studies is reached with respect to trends in the abundance of the skates and rays.With the exception of the starry ray, declines in the populations of these species seem to be indicated by most studies. The greater weever is another species which seems now to be almost absent from the southeastern North Sea. When comparing studies in this way, some regard must be given to the possible confounding effects of variation in the sampling efficiency between studies. This is due to a number of factors, such as variation in the type of fishing gear, each differing in catch efficiency for different species, or differences in the way samples are sorted once on board ship. These are discussed in some detail in Section 3.1.

The effect of fishing on non-target species is clearly of great importance if the ecosystem effects of fishing are to be given serious consideration. Greenstreet and Hall (1996) demonstrated significant differences between the species composition of non-target components of the groundfish species assemblage between two time periods, 1929–1953 and 1980–1993. These were not associated with changes in species diversity, except in the area where fishing effort had been highest for the longest period of time. This suggests that sustained fishing pressure can have effects on the relative abundance of non-target species. This is perhaps an area where more integrated and collaborative research is required.

7.2.9 References

Corten, A., and van den Kamp, G. 1996. Variation in the abundance of southern fish species in the southern North Sea in relation to hydrography and wind. ICES Journal of Marine Science, 53: 1113–1119.

Greenstreet, S.P.R., and Hall, S.J. 1996. Fishing and the ground-fish assemblage structure in the north-western North Sea: and analysis of long-term and spatial trends. Journal of Animal Ecology, 65: 577–598.

Heessen, H.J.L. 1996. Time series data for a selection of forty fish species caught during the International Beam Trawl Survey. ICES Journal of Marine Science, 53: 1079–1084.

Heesson, H.J.L., and Daan, N. 1996. Long-term trends in ten non-target North Sea fish species. ICES Journal of Marine Science, 53: 1063–1078.

Rijnsdorp, A.D., Leeuwen, P.I., van, Daan, N., and Heessen, H.J.L. 1996. Changes in the abundance of demersal fish species in the North Sea between 1906–1909 and 1990–1995. ICES Journal of Marine Science, 53: 1054–1062.

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: 1094–1112.

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: 1085–1093.

7.3 OSPAR Region III

7.3.1 Sampling the non-target species of OSPAR Region III

Long time-series of otter trawl groundfish surveys in this OSPAR region are described in Table 7.3.1.1 (ICES, 1997) and range in duration from 16 years (Scottish groundfish survey and English Celtic Sea and Western Approaches survey) to surveys initiated in the current year. None of these datasets were available to this Working Group for analysis of time-series trends for specific non-target species. As all the ICES Divisions within this region have experienced fishing effort for many years before the start of these demersal surveys, it is unlikely that trends in decreasing abundance of non-target species could be directly attributed to fishing activity. In the North Sea, analysis of time-series from the beginning on the 20th Centrury have been necessary to observe changes in abundance of non-target species (see Section 7.2).

Table 7.3.1.1. List of bottom trawl surveys in 1996 in Sub-areas VI, VII, and VIII, and Division XIa.

The Scottish Groundfish Survey in Division VIa (code: SGF6a)

Start:	1981
Gear:	36/47 GOV trawl, large rubber bobbins, 20 mm liner
Timing:	quarter 1 (March since 1986)
Target:	cod, haddock, whiting, saithe and herring
Stratification:	by rectangle
Depth strata:	no
No of hauls:	40
Continuation:	continued in 1997 and 1998
Contact:	Andrew Newton, SOAEFD, Aberdeen, Scotland UK

The Scottish Groundfish Survey in Division VIb (code: SGF6b)

Start:	1985
Gear:	48ft Aberdeen trawl, large rubber bobbins, 35 mm cover
Timing:	quarter 3 (September)
Target:	haddock
Stratification:	by rectangle
Depth strata:	no
No of hauls:	45
Continuation:	continued in 1997 and 1998
Contact:	Andrew Newton, SOAEFD, Aberdeen, Scotland UK

The Scottish Mackerel Recruit Survey (code: SMR)

Start:	1985
Gear:	36/47 GOV trawl, large rubber bobbins, 20 mm liner
Timing:	quarter 4 (November / December)
Target:	mackerel only until 1995 (cod, haddock, whiting, herring added in 1996)
Stratification:	by rectangle
Depth strata:	no
No of hauls:	50
Continuation:	Longterm, area redefined 1997 (see text)
Contact:	Andrew Newton, SOAEFD, Aberdeen, Scotland UK

West Coast Groundfish Survey (Code: WCGS)

Start:	1990
Gear	commercial trawl, rockhoppers, 20 mm liner
Timing	quarter 4 (October/November)
Stratification	by rectangle
Depth strata:	no
Target	commercial species
No Hauls	71
Continuation	1997 and 1998
Contact	Paul Connolly, FRC, Dublin, Ireland

The Irish Sea Recruit Survey (code: ISRS)

Start:	1983
Gear:	3-bridle otter trawl, close contact groundgear, 20 mm codend
Timing:	quarter 2 (June) and quarter 3 (September)
Stratification:	fixed stations
Depth strata:	no
Target:	cod, whiting, haddock and plaice
No of hauls:	28 each survey
Continuation:	continued in 1997, will be discontinued in 1998
Contact:	Paul Connolly FRC, Dublin, Ireland

Table 7.3.1.1. Continued

The Irish Sea and Celtic Sea Groundfish Survey (code: ISCSGS)

Start:	1997
Gear:	20/25 GOV trawl, standard groundgear, 20 mm liner
Timing	quarter 4 (October)
Stratification	by rectangle
Depth strata	<50, 50-100, 100-150, 150-200, 200-250, >250.
Targets:	commercially important species
No hauls:	50
Continuation:	will commence in 1997
Contact:	Paul Connolly, FRC, Dublin, Ireland.

The West and South Coast of Ireland Recruit Survey (code: WSCRS)

Start:	1992
Gear:	dual purpose otter trawl, medium bobbins, 20 mm codend
Timing:	quarter 3 (July)
Stratification:	by depth, fixed stations
Depth strata:	no
Target:	inshore juvenile fish
No of hauls:	74
Continuation:	continued in 1997
Contact:	Paul Connolly, FRC, Dublin, Ireland

The Celtic Sea and Western Approaches Groundfish Survey (codes: CSGF)

Start:	1981
Gear:	Portuguese high-headline trawl, medium rubber bobbins, 20 mm liner, tickler
Timing:	quarter 1 (March)
Stratification:	by depth and latitude
Depth strata:	0-89, 90-114, 115-139, 140-179, >180 m
Target:	mackerel and commercially important species
No of hauls:	75
Continuation:	continuing in 1997 and 1998
Contact:	John Nichols, MAFF, Lowestoft, England UK

The Northern Ireland Groundfish Survey in Division VIIa (code: NIGFS)

Start:	1991
Gear:	Otter trawl, rockhoppers, 20 mm liner
Timing:	quarter 1 (March), quarter 3/4 (September/October) (also June 1991-94)
Stratification:	by depth, area and bottom type (7), fixed stations
Depth strata:	<50 m, 50 m+
Target:	commercially important species
No of hauls:	45 per survey
Continuation:	March and September surveys to be continued in 1997 and 1998
Contact:	Mike Armstrong, DANI, Belfast, Northern Ireland UK

The German Survey in the western waters (code: GSWW)

Start:	1991
Gear:	36/47 GOV trawl, standard groundgear, 20 mm liner
Timing:	quarter 2 (April)
Stratification:	by rectangle
Depth strata:	no
Target:	commercially important species
No of hauls:	40
Continuation:	continued in 1996, not in 1997; again in 1998 as part of the mackerel / horse mackerel egg surveys and subsequently triennial
Contact:	Nils Hammer, BFA-ISH, Hamburg, Germany

7.3.2 Abundance of non-target species using data from the International Beam Trawl Surveys

The catch rates of all fish sampled by the English beam trawl surveys in the western waters of the British Isles have been described in Reports of the Beam Trawl Study Group (ICES, 1994; ICES, 1996), and the results have been analysed and discussed by Rogers *et al.* (1998). As explained in Section 7.3.2, it is unlikely that recent changes will be observed in abundance of non-target species because of the short duration of this dataset. Nevertheless, plots of mean abundance of dab, dragonet species, dogfish, poor cod, and solenette are shown in Figure 7.3.2.1. Fluctuations in annual abundance are likely to be attributable to changes in year class strength, even in those species, such as the dogfish, which have been considered to be particularly vulnerable to fishing activity.

7.3.3 Disappearance of the common skate from the Irish Sea

The best documented example of the effects of fishing activity on species abundance in this OSPAR region is the decline in abundance of the common skate *Raia batis* (Brander, 1981). In 1902, the skate was reported to be common and was frequently landed by trawl and line fisheries, and was also present as a by-catch in shrimp nets. Abundance began to decline in the 1950s and since the mid-1970s the species has been considered to be rare. During the ten year period from 1969–1979, the landings of skate at Concarneau declined by 82 %. Several aspects of the biology of the skate make it vulnerable to fishing, particularly the slow growth rate and high age at maturity. The rate of egg laying of skate is not known but can be estimated as 40 per year, the age at first maturity is thought to be 11 years, and the stock is likely to collapse under conditions of total mortality which exceed approximately 0.37. The only effective protection for the skate is probably a complete halt to all kinds of demersal fishery in which it is caught. As this is unrealistic, it must be accepted that this species and others like it will be fished out as a consequence of the exploitation of other demersal fish (Brander, 1981). In areas where relic populations exist, the closure of these areas, which is an approach advised by ACFM for the North Sea, should be considered.

7.3.4 References

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

ICES. 1994. Report of the Study Group on Beam Trawl Surveys in the North Sea and Eastern Channel. ICES CM 1990/G:59.

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ICES. 1997. Report of the International Bottom Trawl Survey Working Group. ICES CM 1997/H:6.

Rogers, S. I., Rijnsdorp, A. D., Damm, U., and Vanhee, W. 1998. Demersal fish populations in the coastal waters of the UK and continental NW Europe from beam trawl survey data collected from 1990 to 1995. Journal of Sea Research, 37.

7.4 OSPAR Region IV - Trends in Non-Target Species and their Relationship to Fishing

A stratified random bottom-trawl survey has been carried out by Spain since 1983 (except 1987) along the Cantabrian Sea continental shelf and Galician waters. Each year, from 50 to 70 tows of 30 minutes were taken in depths from 30 to 500 metres. Abundances and biomasses of all species have been recorded. Details of the survey procedures are summarized in WP-17 and Olaso *et al.* (in press).

The difficulties in applying the data from this survey to the term of reference are the same as for other studies and other regions. The surveys quantify the trends in non-target species under the standard assumptions of constant catchability, etc. However, to determine the part of the trend caused by fishing, rather than by random variation or environmental forcing, requires both good data on intensity and distribution of fishing over the period of the survey, and the ability to partition variance among possible causal factors. Neither the data on fishing nor the ability to determine causation of historic influences are available. Also, as with other OSPAR regions, the surveyed areas have been fished for many years prior to the survey. Therefore, the biggest effects of fishing on the non-target species may have happened before the survey began in recent decades.

From the Spanish survey, data on thornback ray and lesser spotted dogfish (*Scyliorhynus canicula*) were converted into indices of abundance over time. These species were selected because they are thought to be likely to show direct impacts of fishing on abundance of non-target species (ICES, 1996). Other potentially vulnerable species highlighted in that reference occurred in numbers too low for estimation of trends in biomass over time.

7.4.1 Results

Lesser spotted dogfish. Abundance was variable through the 1980s, and then showed a consistent decline from 1990 to 1995. This trend seems to be reversed in the two most recent years, with the biomass in 1997 very similar to the biomass in 1990.

Thornback rays. Abundance appears stable in the early 1980s, although between 1986 and 1990, there is substantial variation in abundance. These fluctuations are so great that it is unlikely that they are caused by changes in the total size of the population, but more probably reflect changes in distribution or in local availability to the fishery. As with lesser spotted dogfish, there appears to be a declining trend from 1990 to 1995, with a substantial increase in the two most recent years.

7.4.2 Discussion

Both of these species are most abundant in the shallowest depth stratum (30-100 m). Trawling is prohibited by regulation inshore of the 100 m contour, although some illegal fishing is known to occur in these areas. Even if there are taken in fisheries, their survival rate when discarded has been measured at over 90 % (Kaiser and Spencer 1995). Because of the prohibition on fishing and the high survivorship if discarded, neither species would be expected to show noteworthy change in abundance due to trawling. In that context, it is interesting that the declining trend in abundance through the 1990s was reversed the second year after artificial barriers were placed inside the 100 m contour, to make illegal trawling much more difficult to conduct. This could have two interpretations. Possibly illegal fishing was so intense that the populations of these two elasmobranchs were unable to maintain themselves with the level of by-catch mortality they were suffering. Alternatively, it is reported that the survey gear has had to align trawl tracks in proximity to the artificial reefs which were constructed, in order to sample in the stratum. If the artificial reefs are attractive to these species, then catch rates might be elevated in years since 1994. In that case changes in availability to the fishing gear continue to influence greatly the trend over time. Only more detailed work at fine spatial scales can begin to disentangle these possible causes of the observed pattern.

There is also a trawl survey conducted in waters from 20-500 m off the coast of Portugal. This survey began using a stratified random design in the mid 1980s, although the design changed to fixed stations in 1989. Working Paper (WP-10) reported on the multispecies assemblages identified by various multivariate analysis of the catch data. Because these assemblages were defined in part (usually primarily) by species for which there are directed fisheries, they are not appropiate for looking at changes in abundance of nontarget species. However, the data series may contain such information for some non-targetted species. The data should be analysed futher, first for non-target species which are sampled reliably by the survey, and then for trends in the abundance of those species.

7.4.3 References

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7.5 OSPAR Region V

In the areas where the groundfish and midwater pelagic fisheries operate in OSPAR Region V, there are only a few opportunistic and partial surveys. No time series exist on abundance of target species, let alone non-target species. However, records of exploratory cruises by several countries in the 1970s provide potentially important information on species composition, size composition, and ages of many species, particularly in the areas around Porcupine and Rockall Banks (Ehrich, 1983; Gordon, 1986; Gordon and Duncan, 1987b; Merritt *et al.*, 1991a, 1991b). Repeat surveys of these areas, or full monitoring of catch composition from fisheries, might give data useful for comparative analyses in light of recent expansion of fisheries. A series of surveys on Rockall Bank has been conducted since the mid-1980s. Unfortunately, neither data nor scientific reports from that series have been published yet; nor have they been made available to ICES. Concerns about the possible over-exploitation of some target species in this region (See Section 5.5) suggest that any species taken regularly as by-catch in these fisheries may also be impacted. However, there are no data to shed light on this matter.

For the traditional fisheries around the Azores, Portugal initiated a series of surveys starting in 1993. These surveys focused on collecting abundance and biological information on the species targeted by the fisheries, for comparison with data from surveys in the early 1980s (ICES, 1996). Information was recorded on a few non-target species in recent surveys, but it has not been established if comparable data can be recovered from the earlier surveys.

A Term of Reference for WGDEEP might be to examine both the Rockall and the Portugese survey data sets, reporting something on trends in non-target species.

There are no surveys which would give information about changes in abundance of species taken as by-catch in the fisheries for tunas and tuna-like fishes, although the by-catch monitoring programme which has been implemented may give useful information if continued for an adequate period. There are particular concerns about impacts on some species of shark and there are discussions among FAO, ICES, and ICCAT with regard to expanding programmes to monitor the status of sharks in several areas, including the mid-Atlantic.

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Figure 7.1.1.2.1. Abundance indices off West and East Greenland, and total for long rough dab.

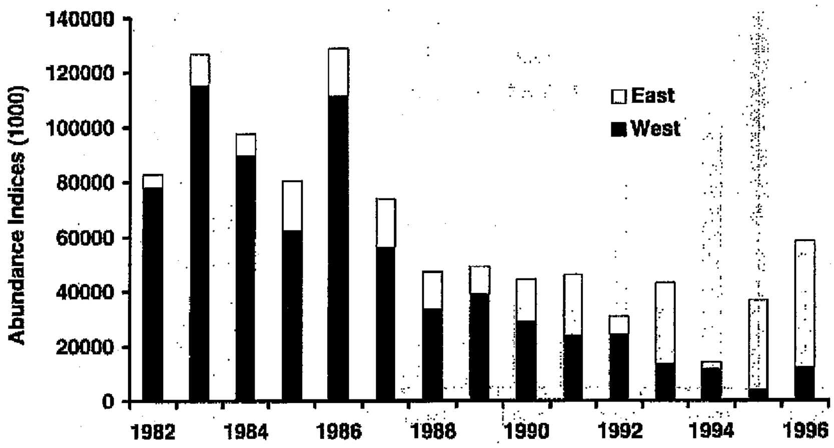


Figure 7.1.1.2.2. Abundance indices off West and East Greenland, and total for common catfish.

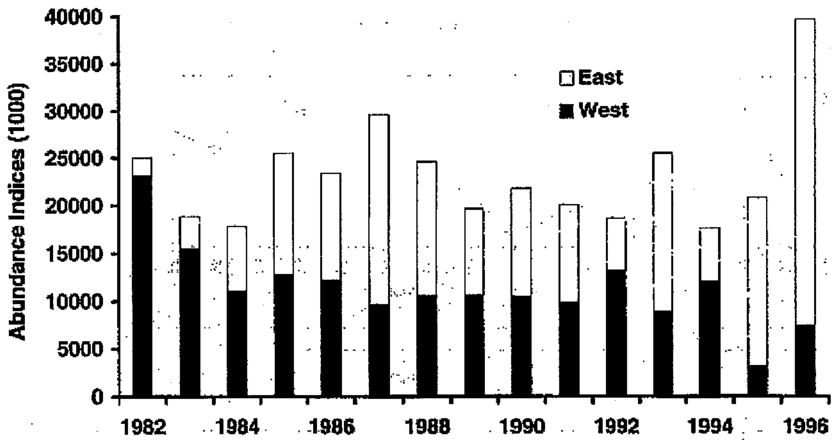


Figure 7.1.1.2.3. Abundance indices off West and East Greenland, and total for spotted catfish.

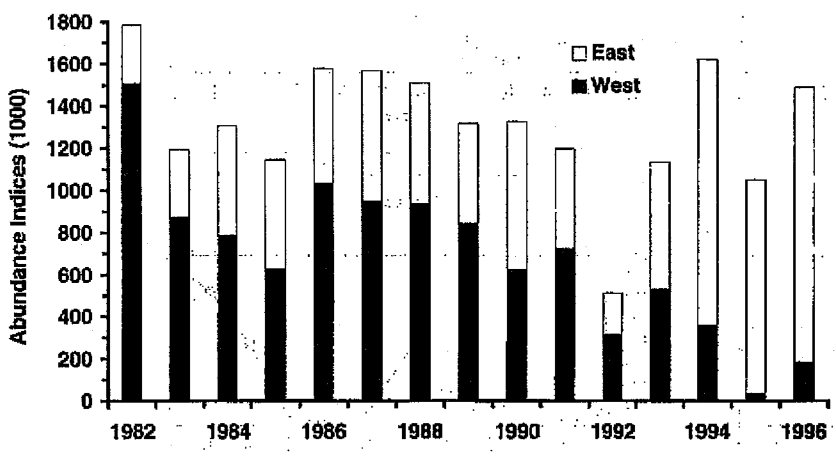


Figure 7.1.1.2.4. Abundance indices off West and East Greenland, and total for starry ray.

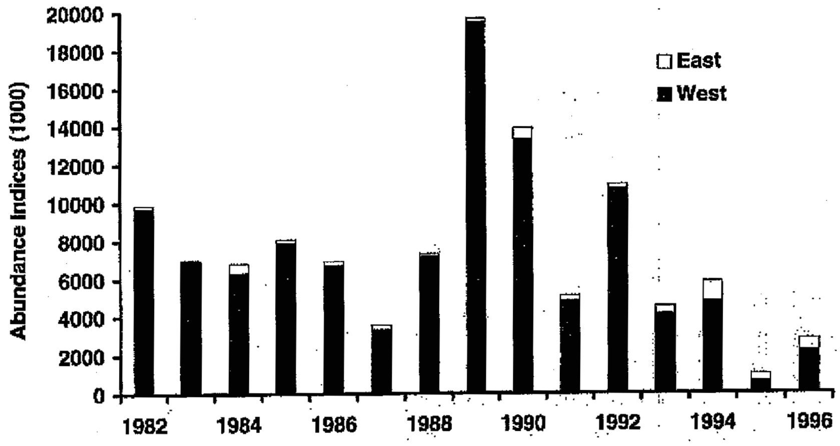


Figure 7.1.2.1.a. Trends in abundance indices for some selected species in the Barents Sea bottom trawl survey (subareas A, B, C, and D).

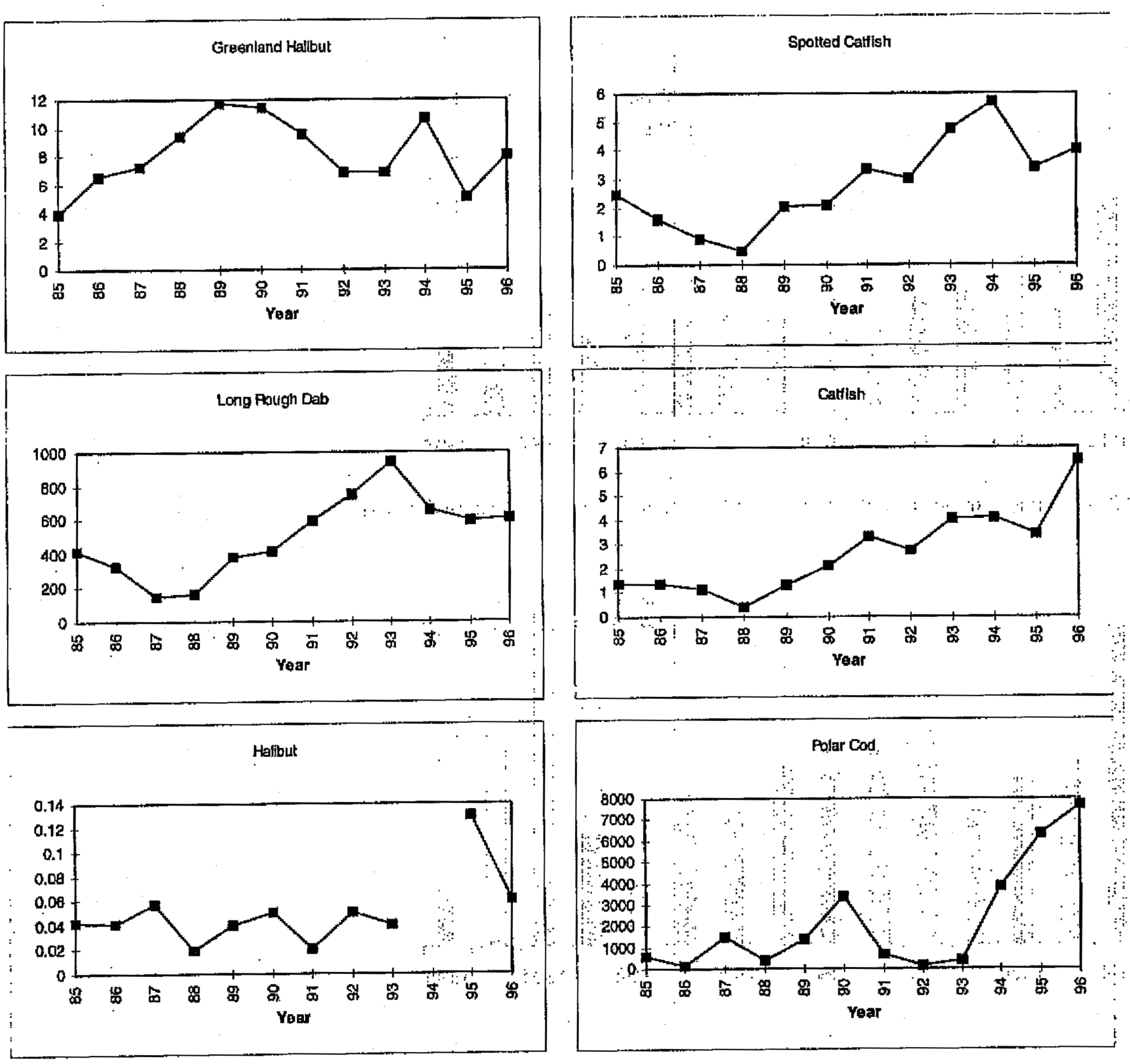


Figure 7.1.2.1.b. Trends in estimated mean length for some selected species in the Barents Sea bottom trawl survey (subareas A, B, C, and D).

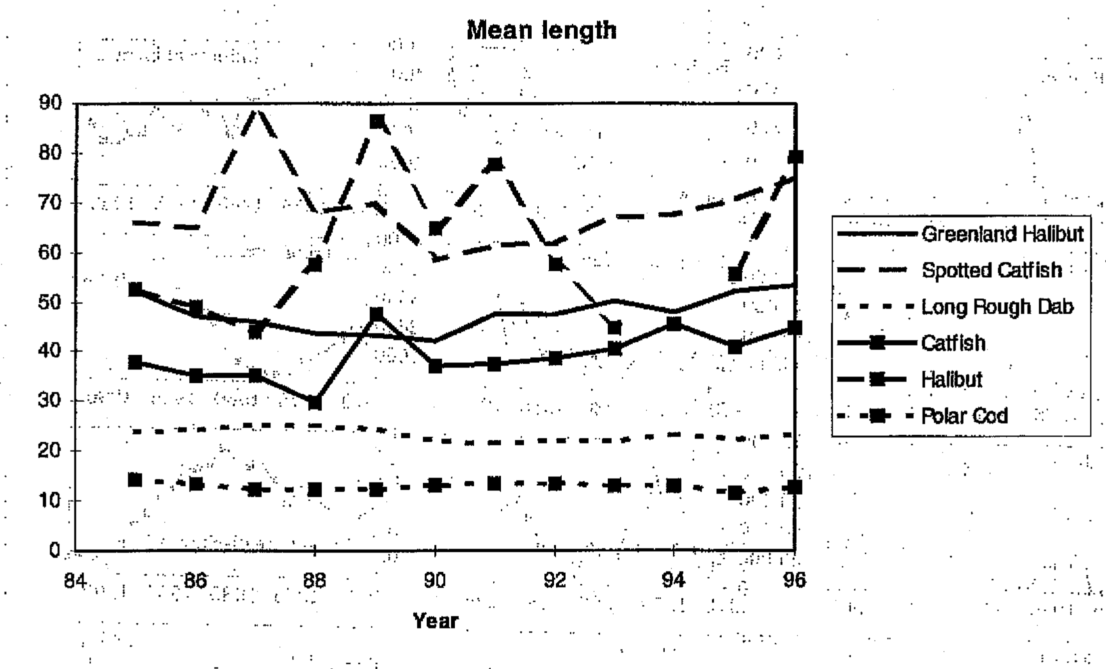


Figure 7.1.2.2. Trends in abundance indices for 12 species with very low or no commercial value. Indices are estimated using data from the Norwegian bottom trawl survey in the Barents Sea (subareas A, B, C, and D).

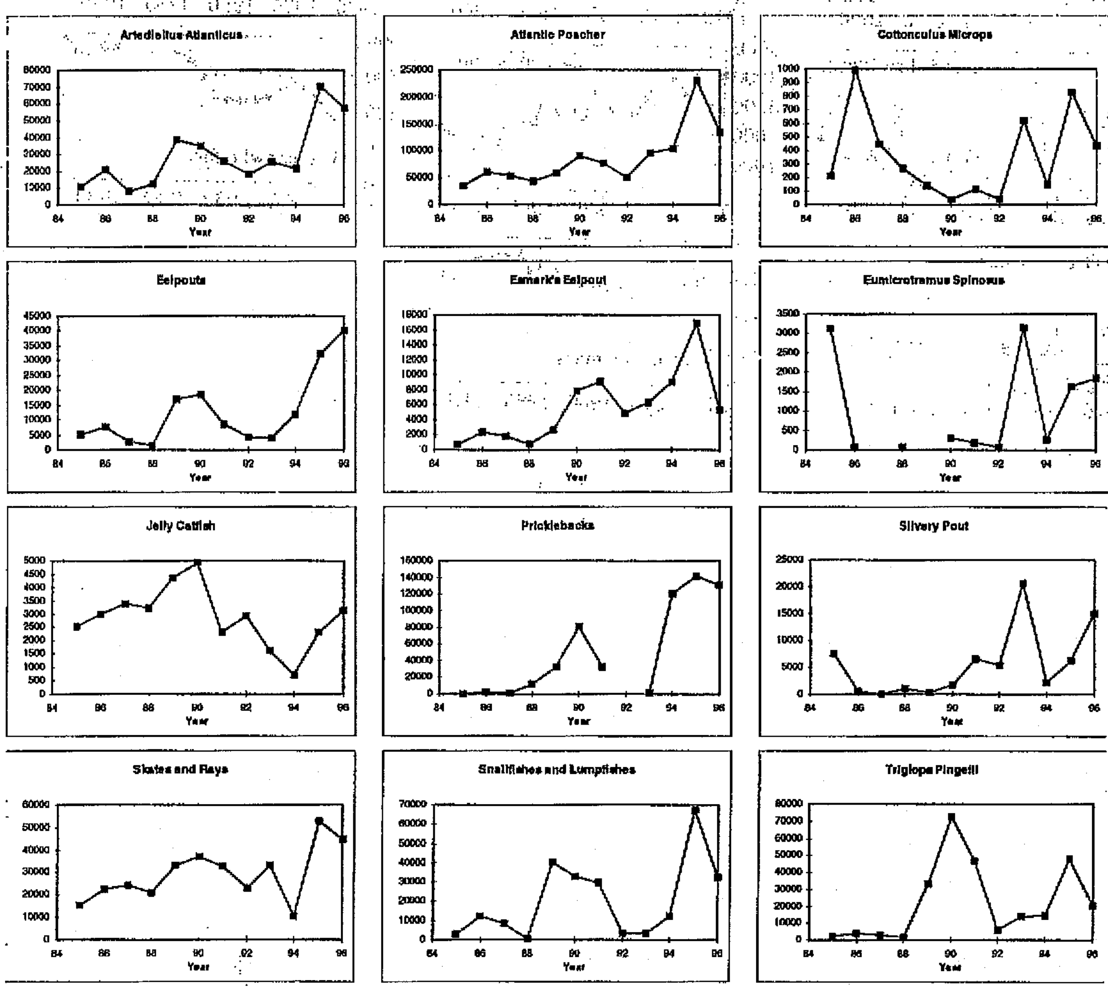


Figure 7.2.2.1. Trends in the abundance of ten non-target fish species in the North Sea and seven environmental time series (Heessen and Daan, 1996).

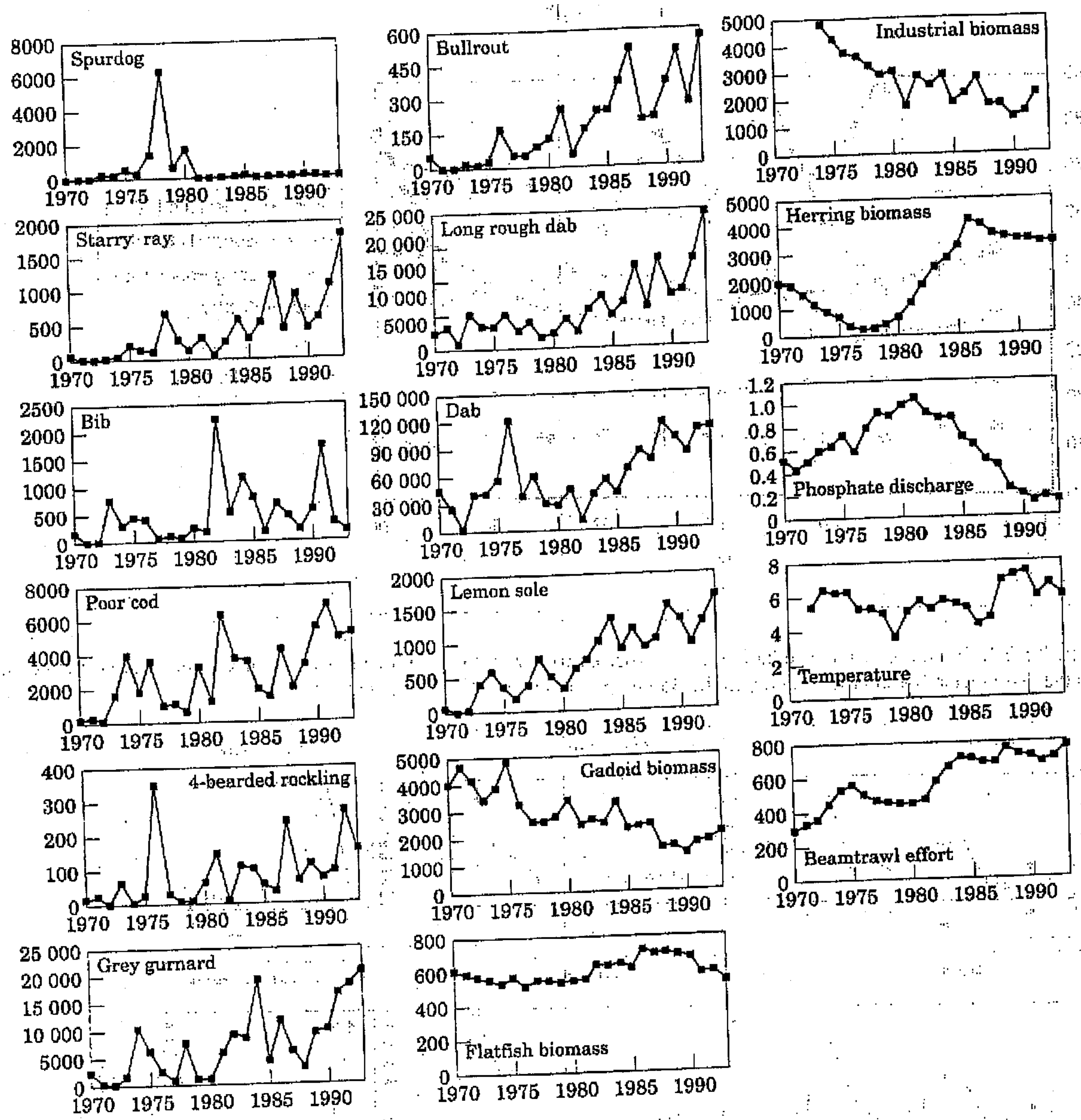


Figure 7.2.3.1. Trends in the abundance of 26 non-target fish species in the North Sea (Heessen, 1996).

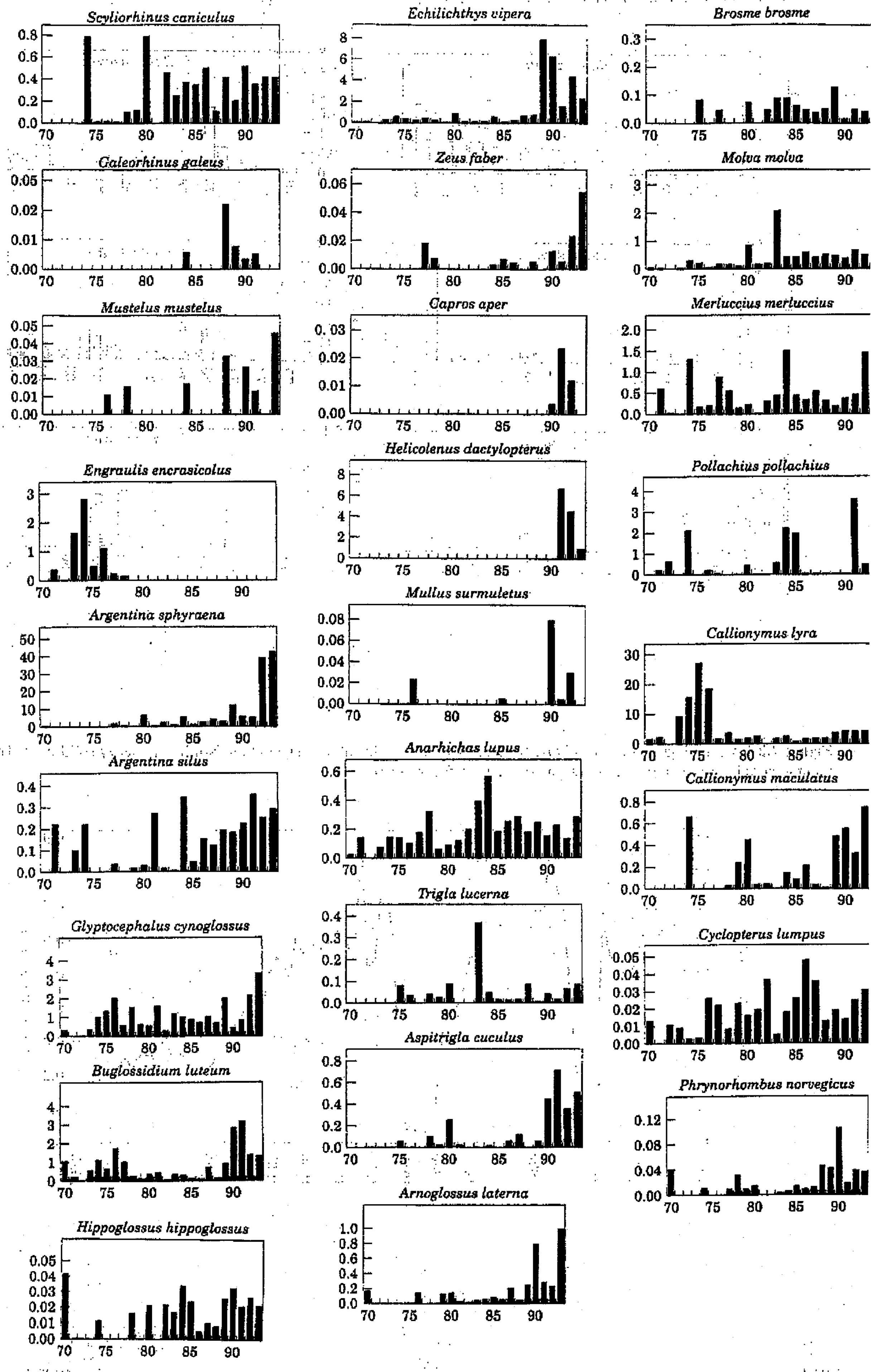


Figure 7.2.4.1. Trends in the catch rates of six ray species in the IBTS 1970–1993. A five year running mean is shown for the starry ray *Raja radiata* (Walker and Heessen, 1996).

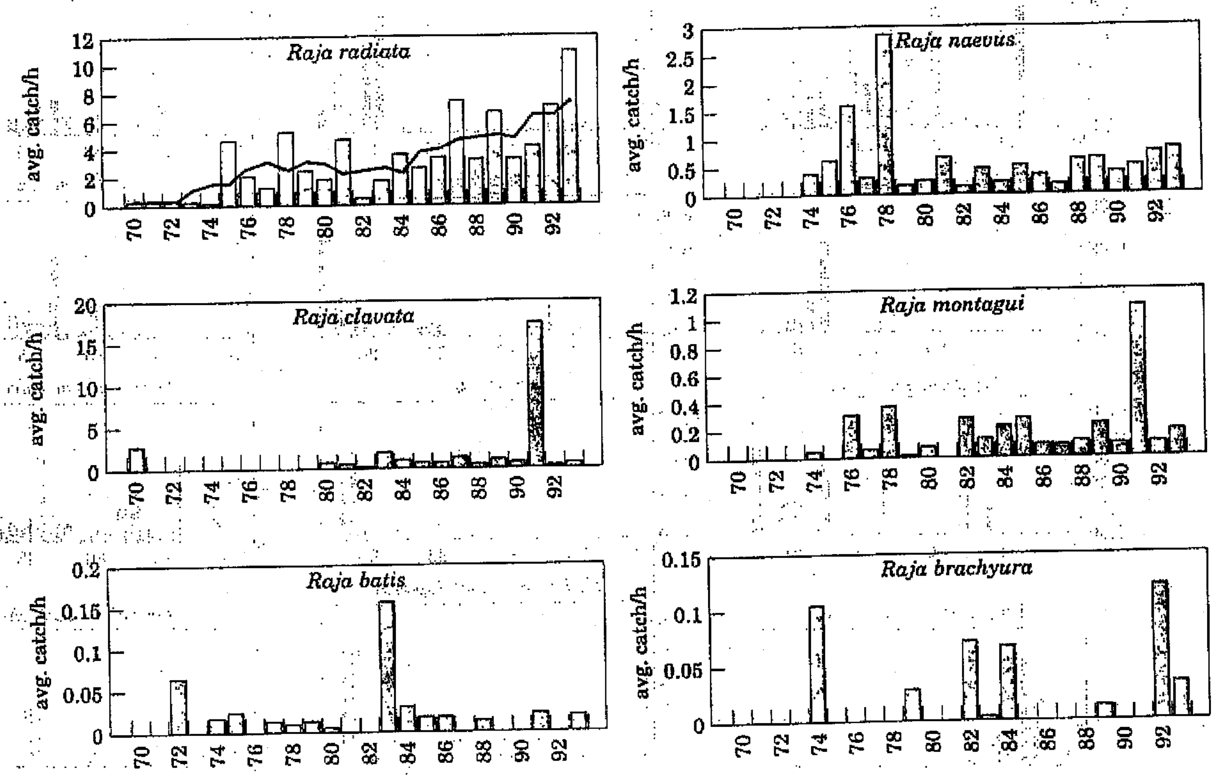


Figure 7.2.4.2. Landings of skates and rays from the southern (light solid line) and total North Sea (heavy solid line). Trend in plaice fishing mortality (dashed line) is shown as an index of fishing effort.

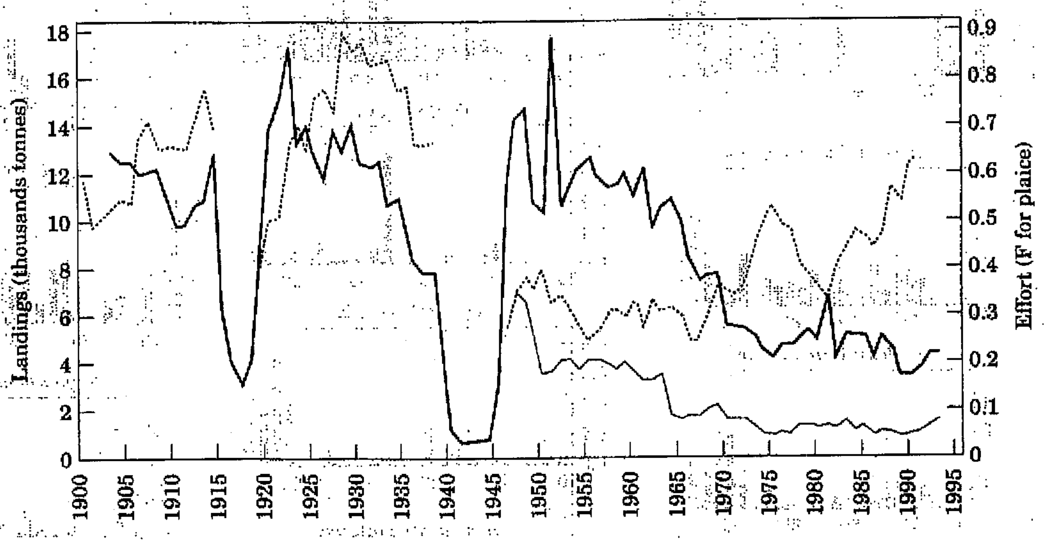


Figure 7.2.5.1. Long-term variation in the density of three target species and seven non-target species in the northwestern North Sea. Log catch per hour is shown and lines are fitted by LOWESS smooth.

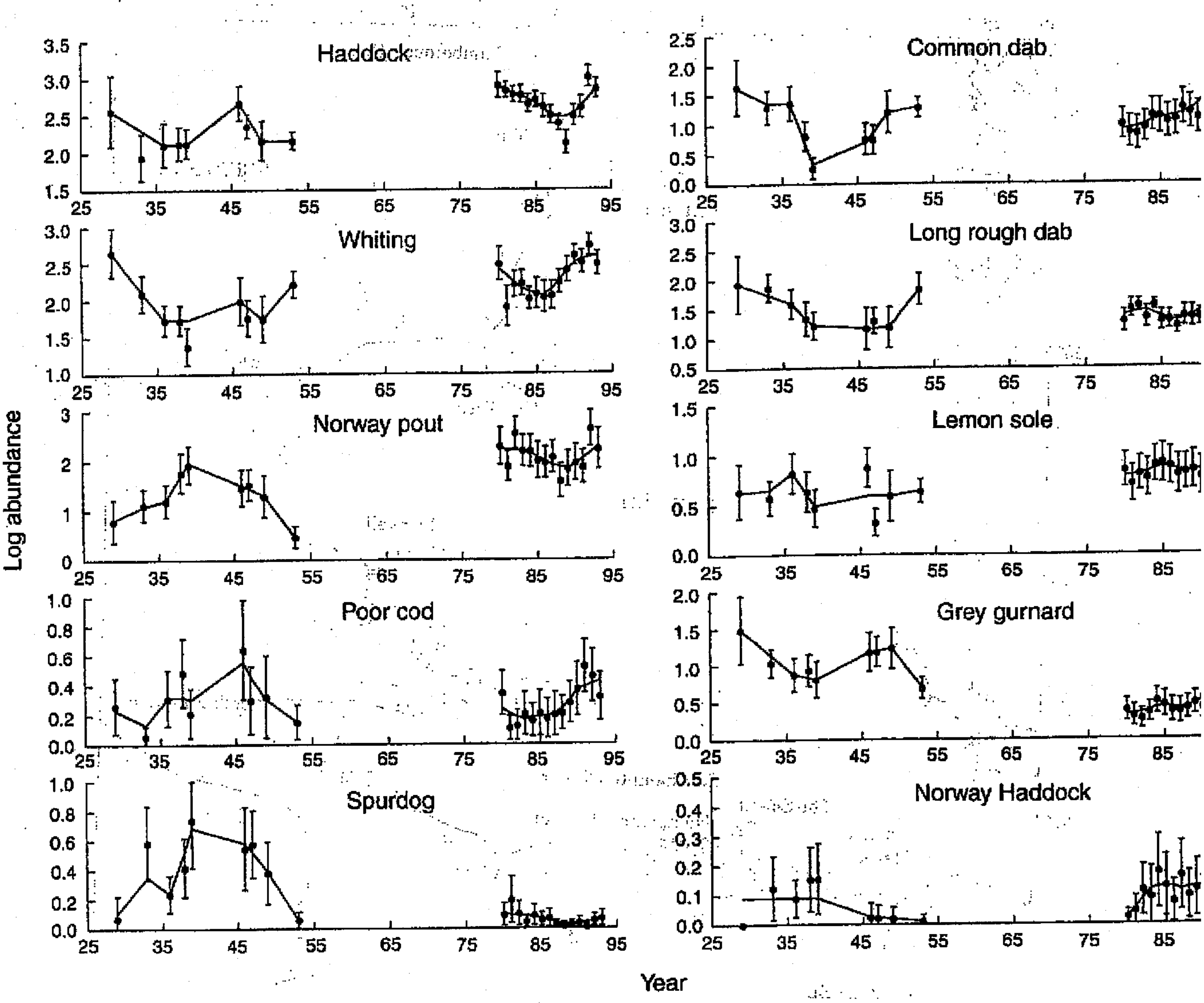


Figure 7.2.6.1. Map showing three areas included in the analysis (Rogers and Millner, 1996).

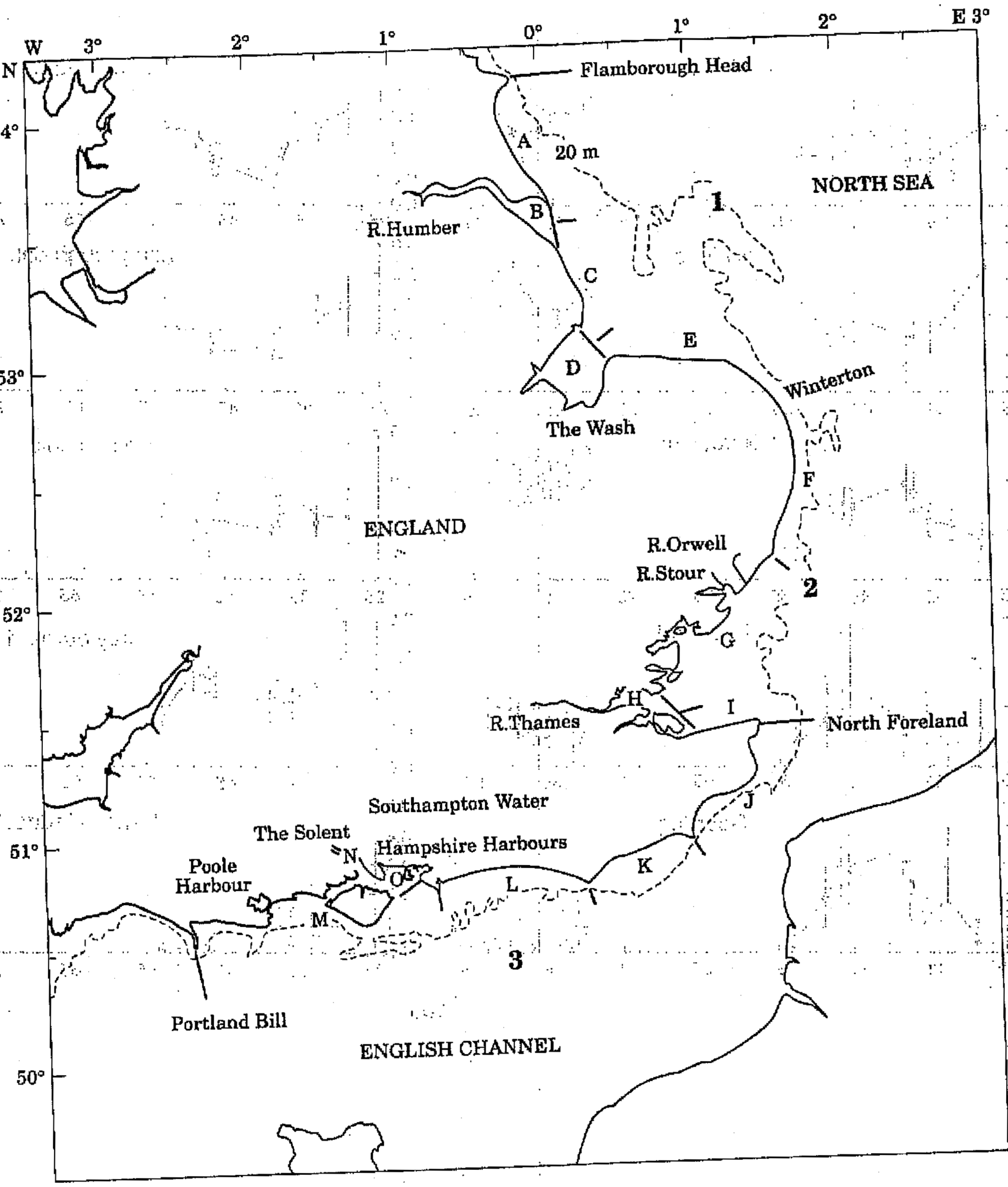


Figure 7.2.6.2. Mean catch (numbers per 1000 m²) of (a) hooknose, (b) butterfish, (c) sea snail, (d) eelpout, (e) balan wrasse, (f) spotted ray, (g) lesser weever, and (h) solnette in the period 1973–1995 for region 1 (continuous line), region 2 (dot dash line), and region 3 (dashed line) (Rogers and Millner, 1996).

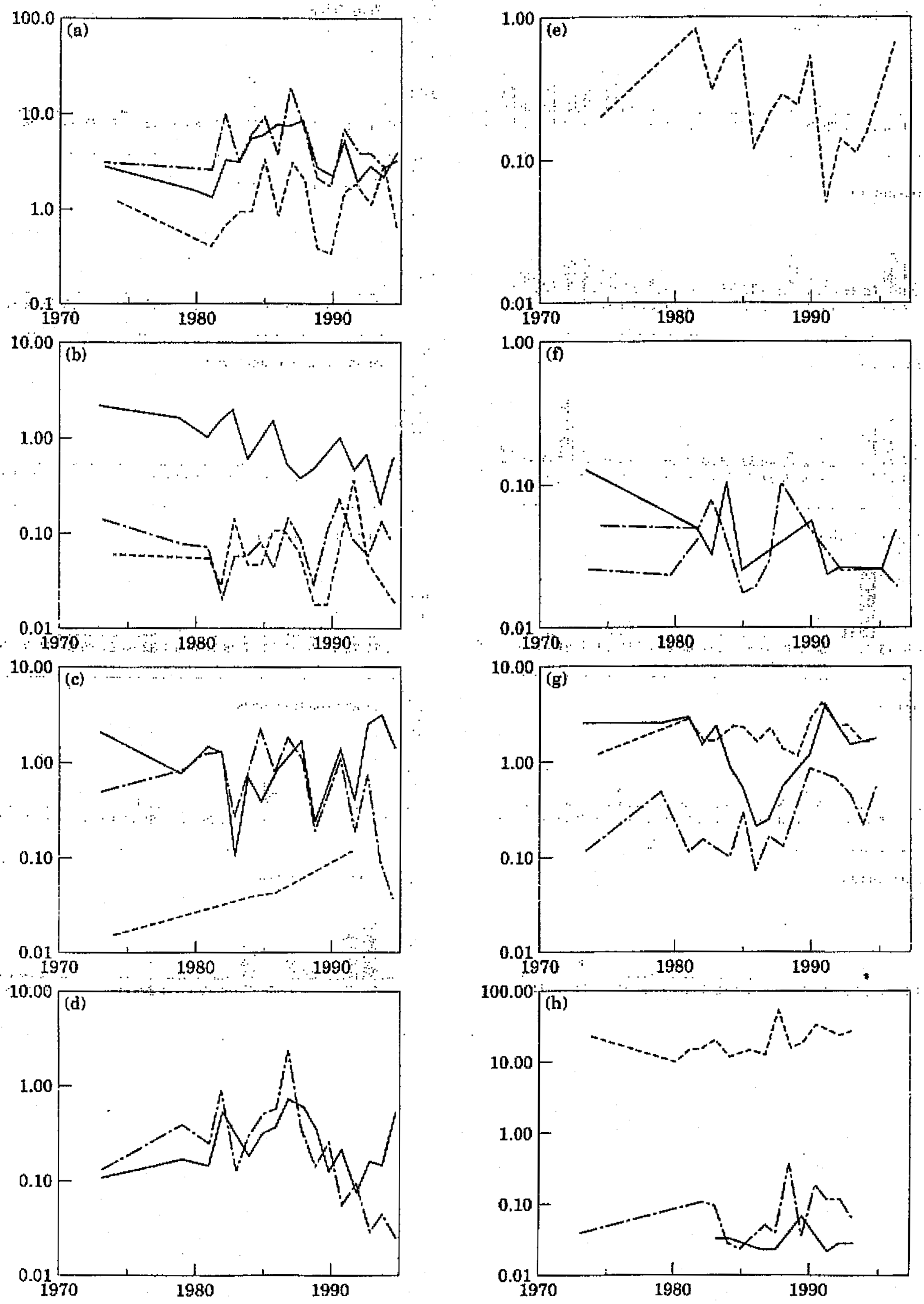


Figure 7.2.7.1. Average abundance of twelve fish species (numbers per hour fishing) in the southern North Sea over the period 1970–1994 (Corten and van den Kamp, 1996).

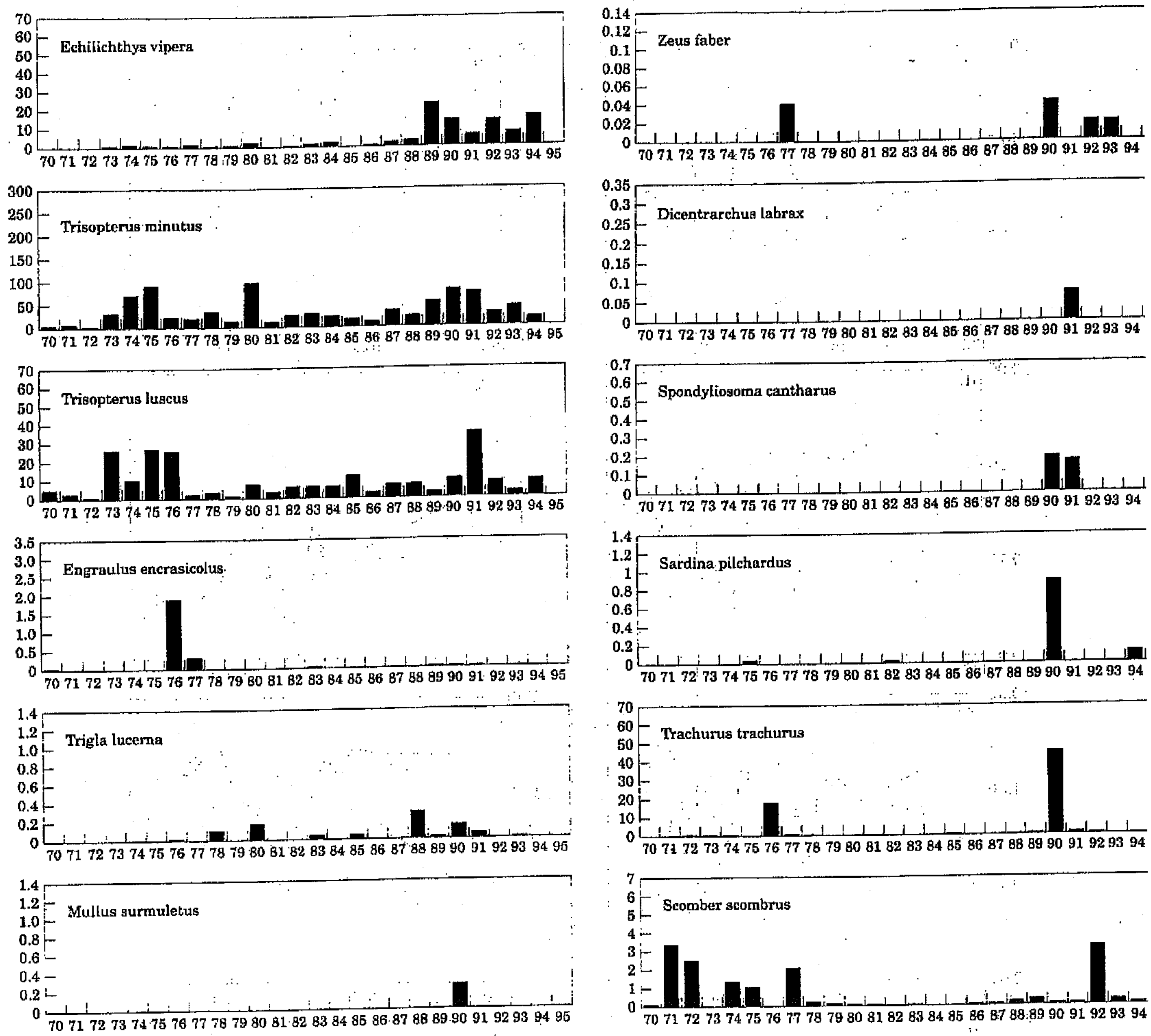


Figure 7.3.2.1. Mean annual abundance of selected non-target species sampled during the beam trawl survey in Division VIIa, 1989–1996.

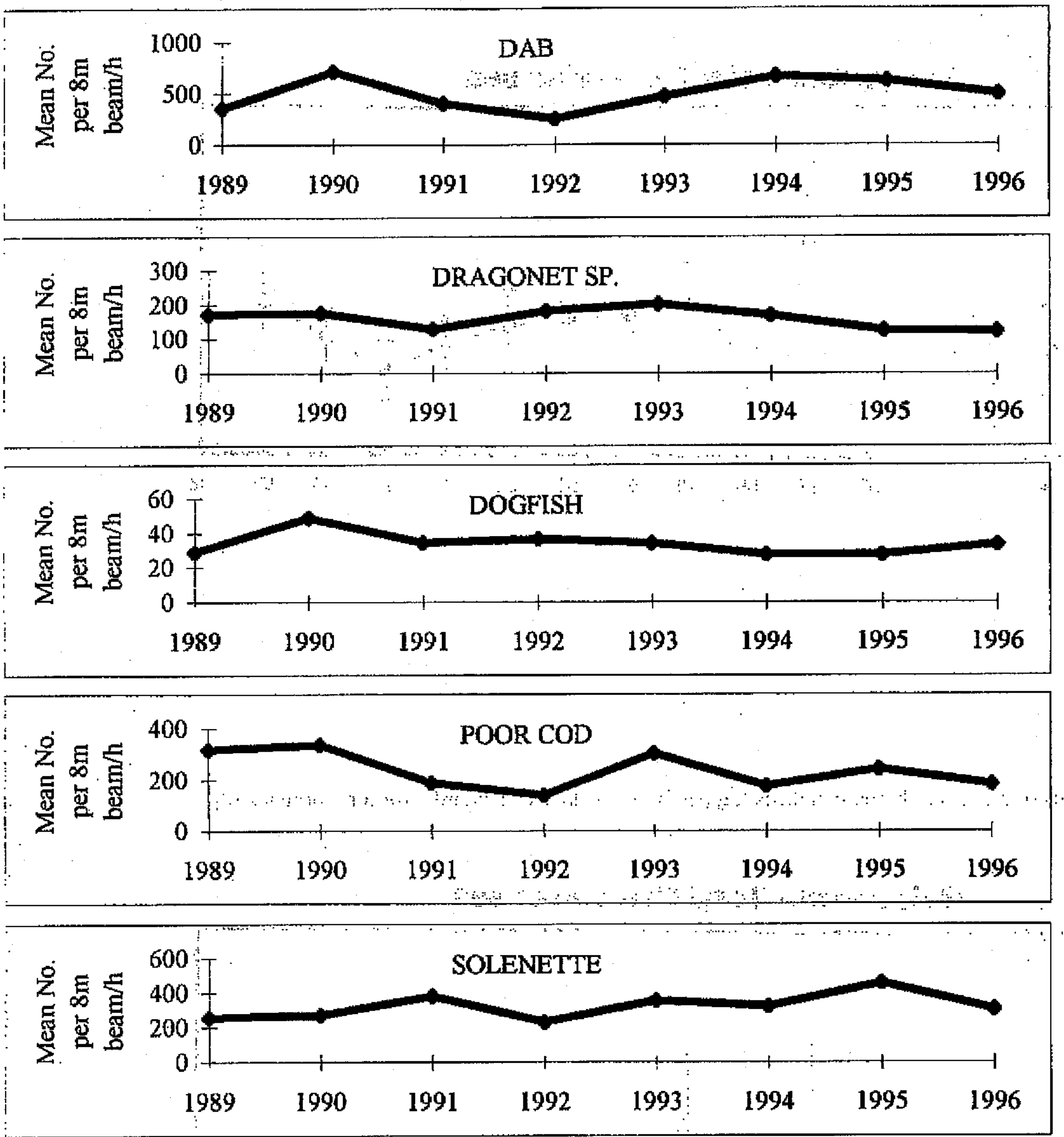


Figure 7.4.1.1.a. Historic series of biomass indices (kg per 30 minutes) and standard error for dogfish.

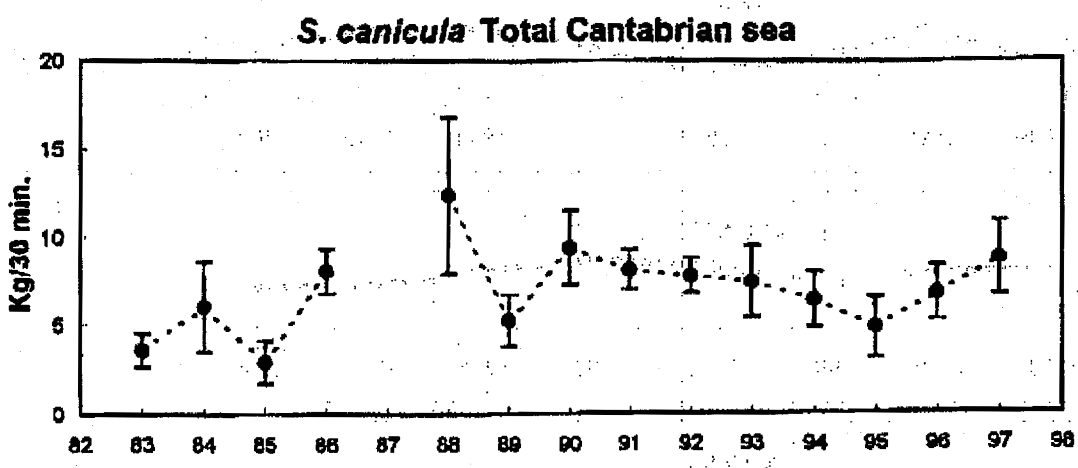
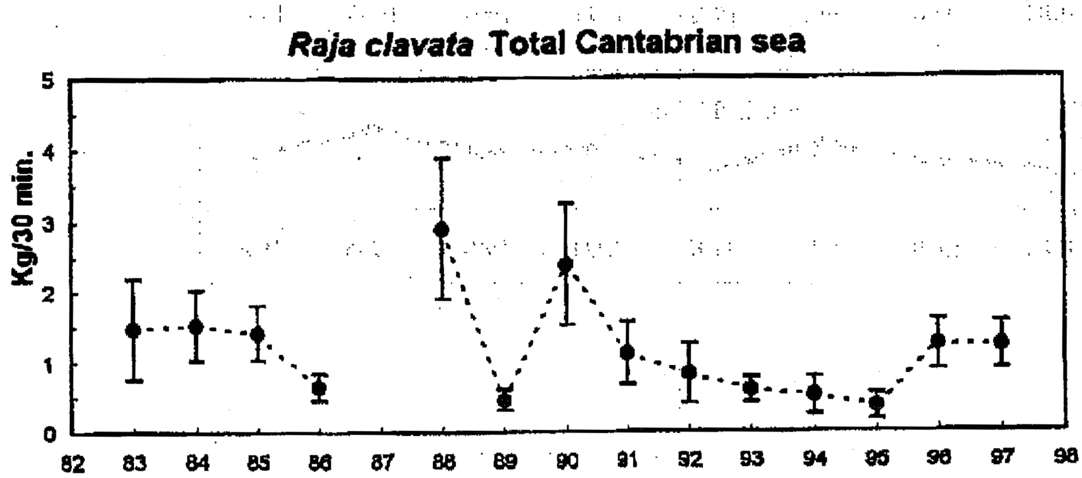


Figure 7.4.1.1.b. Historic series of biomass indices (kg per 30 minutes) and standard error for thornback ray.



8 REFERENCE POINTS INCLUDING ECOSYSTEM CONSIDERATIONS

Develop and examine potential reference points which might be used for including ecosystem considerations in relation to the precautionary approach.

This Term of Reference is specifically about reference points, and not about the broader management objectives for which the quantitative reference points are developed and used. The material which follows will be readily interpretable in the context of current approaches to fisheries. However, WGECO considered a much broader framework than just traditional fisheries management objectives. Many other types of objectives already influence fisheries practices, from very local scales (for example, the protection of specific bivalve beds close to shore-based viewpoints, because they attract concentrations of seaducks) to very large ones (the objective of protecting ecosystem diversity, for example). It is important that the following arguments are viewed as applying in all of these contexts, and not just as serving traditional fisheries management objectives. Likewise, it is important that specific objectives be discussed and set by society in many contexts, in addition to fisheries.

8.1 Statement of the Issue

The precautionary approach (FAO, 1995; Doulman, 1995; Garcia, 1996) has been accepted as a guiding principle in fisheries management. It covers biological, social, and economic aspects of fisheries. Up to now the practical implementation of the precautionary approach has led to the establishment of limit reference points and target reference points for commercial species. These reference points are recommended as quantitative management objectives. At the current exploitation pattern of fish stocks, the short-term objective is to have a low probability of fish stocks falling below limit reference points, to ensure a long-term sustainability (ICES, 1997). Target reference points are viewed as long-term objectives.

An additional aspect of the precautionary approach is the integration of fisheries management and ecosystem management. An ecosystem approach in the management and assessment of fisheries involves considering all relevant physical, chemical, and biological ecosystem variables (Anon., 1997). It thereby implies a widening of the current implementation of the precautionary approach. The question at stake is whether reference points being developed for commercial species are sufficient to ensure an effective ecosystem management? This chapter reviews the ecosystem considerations of different potential reference points, including single species reference points for target and non-target species, multispecies and ecosystem reference points, mass-balance perspectives, and community metrics.

Even though ICES acknowledges the need to manage fisheries in a manner which ensures ecosystems that are sustainable in the sense that no species becomes extinct, little work has been done thus far on how to define reference points in an ecosystem context. Naturally such definitions would not only be restricted to fish but would need to include other components of marine fauna such as benthos, seabirds, and marine mammals, for some of which reference points relating anthropogenic impacts to population status has either been defined elsewhere or are non-existent. In addition, it would need to consider not only how fishing mortality affects individual species and their genetic make-up, but also how discarding and physical seabed disturbances affect the system.

One of the great effects of fishing is the harvesting of target species. If it were the case that reference points were used as intended in management, fisheries would already be much further on the way to meeting any specified ecosystem objectives. On the other hand, commercially important species are by their nature often highly productive components of the ecosystem. Reducing their abundances through fishing may have great impacts on the dynamics of the food web. Also, because they often are less productive, non-targeted species can be much more vulnerable to mortality caused by fishing than many commercially important species. It has been proposed that these sensitive species could be useful indicators for determining the state of the ecosystem. With respect to the single species approach, attention is given to the usefulness of such signal species as a basis for additional reference points for fisheries management.

Multispecies models contain more ecosystem considerations than their single species counterparts. The multispecies models used by ICES account for predator/prey relationships. In work completed to date they have led to more conservative estimates of reference points and estimate lower fishing mortality rates for a sustainable fisheries than single species models (ICES CM 1997/Assess:16). In that sense they require more conservative fisheries to achieve an equal degree of precaution.

As discussed in Section 3, fisheries can affect community structures. Due to the high selectivity of fishing, certain community metrics may be altered. The question is, can metrics like shifts in size or productivity at different trophic levels also serve as potential ecosystem reference points. The results of the work in Section 3 will be used in the discussion of this section. The value of multispecies modelling, mass-balance models, MSVPA, and other alternatives are also reviewed with regard to their potential usefulness in providing possible ecosystem reference points.

Thus, to answer the question whether there is a need for extra reference points from an ecosystem perspective we will discuss the relevance of:

- reference levels assessed by various models;
- reference levels for community metrics and indicator species (target and non-target) on the basis of survey data.

8.2 Specific Reference Points Considerations

8.2.1 What ICES already advises

ICES considers a stock to be within safe biological limits if the spawning stock biomass (SSB) is above the minimum biologically acceptable level (MBAL) with high likelihood, and there is a low likelihood of SSB falling below MBAL in the medium term, at status quo fishing mortalities. MBAL plays a key role in ICES advice. It is estimated in a variety of ways, but is generally considered to be the SSB at which either the probability of poor recruitment is increased or the probability of good recruitment is decreased markedly. The total allowable catches (TACs) advised by ICES are based on fishing mortalities. ICES does not advise one TAC-level but gives[1] short- and medium-term forecasts (if possible) of the stock development at different exploitation levels. The responsibility of using a precautionary approach in setting the definitive level of a TAC is vested in the fisheries management agencies receiving advice from ICES.

Within ICES, several Working Groups and Study Groups are discussing biological reference points which can be used in the ICES advice in the near future. For a description of these discussions on reference points for commercial species, the reports of the Advisory Committee on Fisheries Management (ACFM) Study Group on the Precautionary Approach (ICES CM 1997/Assess:7) and the Comprehensive Fishery Evaluation Working Group (WGCOMP) (ICES CM 1997/Assess:15) are of interest. The ACFM Study Group addressed technical aspects of estimating suitable reference points given the uncertainty inherent in fisheries data and in biological systems. Subsequently, WGCOMP interpreted the precautionary approach to conclude that as limit reference points for commercial fish stocks, fishing mortality should be kept below F_{msy} and that the biomass should be kept above B_{msy}[2]. Even for some commercial stocks, these reference points cannot be assessed with a high accuracy. Nevertheless, they urge that fishing mortality rates and biomass limit reference points are required. Because of the uncertainty in the assessments, WGCOMP recommends to keep the probability of a stock exceeding the limit reference levels below 5 % for any given year.

In addition to the work produced by WGCOMP and the ACFM Study Group, the Multispecies Assessment Working Group (MAWG) compared the difference of the above recommendations between a single species approach and a multi-species approach. They concluded that the theory of a precautionary approach should be elaborated to multispecies fisheries management. Multispecies interactions will affect the biological reference points and responses of populations to rebuilding strategies. The multispecies considerations make the reliability of single species reference points more uncertain, and suggest even greater caution is necessary to achieve a low risk to the stock (ICES CM 1997/Assess:16).

These developments in the approach of reference points are considered very important and promising developments in implementing the precautionary approach in fisheries management. As a part of the dialogue needed, ICES will provide a summary of the progress made so far in developing a framework for precautionary reference points in 1998 (ACFM, 1997).

While ICES has made steady progress in developing precautionary reference points, the implementation of ICES advice on single-species harvesting has room for improvement. Recent advice from ICES to management agencies has included tabulations of precautionary TACs advised by ICES, TACs implemented by the competent agencies, and the actual estimated catches or landings. Although there are instances of encouraging trends towards TACs consistent with ICES advice in some fisheries, for many stocks TACs are set higher than ICES advises, and fished harder than managers intend. Because of the difficulties in reducing the present intensity of fishing in many areas, conservation even of individual targeted stocks is at risk in many fisheries. Therefore, discussion of the possible benefits of fisheries management using reference points based on the state of the ecosystem rather than the states of individual harvested

[1] F_{msy} is the fishing mortality which produces maximum sustainable yield - see Beverton and Holt, 1957.

[2] B_{msy} is the biomass of the stock which produces the maximum sustainable yield - see Beverton and Holt, 1957.

stocks is largely speculative. On the other hand, such a discussion might identify compelling reasons at the ecosystem level for fisheries management to practice greater caution.

To begin this speculative discussion the first question to pose is 'If all fisheries were managed so that there was a high probability of achieving conservation objectives for the target fish stocks, would there be a high likelihood of achieving conservation objectives for ecosystems?'. Current knowledge makes the answer to this question clearly 'No' for at least four reasons:

1) the genetic diversity of a target stock might be at risk, even in management regimes that complied with single species reference points for biomass and fishing mortality (Section 8.2.2.1);
2) the conservation of non-target species could be at risk due to direct mortality from fishing activities (Section 8.2.2.2);
3) the conservation of dependent predatory species could be at risk due to local depletion of prey aggregations, even if conservation of the prey stock were being achieved on a much larger spatial scale (Section 8.2.2.3);
4) the conservation of some species could be placed at risk through the abundance of scavenging species increasing due to discarding in fisheries (Section 8.2.2.4).

It is not a coincidence that in all four of these situations the reference points which must be added are still single species reference points. In those cases, the principles and criteria most closely parallel existing approaches to reference points for target stocks. However, WGECO stresses that the issue does not end with single species reference points. The weight of scientific evidence suggests that there are additional reasons at the ecosystem level why the answer would be 'No.'. Examples of these reasons include documented changes to nutrient cycling and remineralization rates and pathways caused by impacts of fishing gear on substrates (Rowe *et al.*, 1975; Prins and Smaal, 1990) and diverse consequences on food web structure and function, caused by fisheries changing the absolute and relative abundances of target and non-target species (see Sections 3, 4, and 9). These types of risks, and their implications for reference points, are discussed in Section 8.2.3.

8.2.2 Additional reference points for species, from an ecosystem perspective

8.2.2.1 Genetic reference points for exploited stocks

In some studies it has been demonstrated that even short periods of intensive exploitation can alter the genetic make-up of an exploited population. Longer periods of exploitation, possibly at rates sustainable with regard to target stock size, may induce genetic responses as well (Lande, 1993; Stokes *et al.*, 1994; Waples, 1995). On a case-by-case basis, however, it is often problematic to differentiate phenotypic responses of life history or morphological traits from loss of genetic characteristics in the population (e.g., Rijnsdorp, 1993). Nonetheless, the loss of genetic diversity is a possible consequence of sustained or episodic intensive fishing, and it is not addressed in existing biological reference points based on biomass and fishing mortality. The Convention on Biological Diversity explicitly recognizes the need for management to conserve genetic diversity of stocks, so additional single species reference points are necessary to fulfill this responsibility.

8.2.2.2 Reference points for non-target species

Despite a reduction in fishing mortality rate of commercial species, which would result from full implementation of the current management advice of ACFM, there may still remain unwanted effects for a number of reasons. Fisheries kill organisms other than the target species. The by-catch mortality can be unsustainable for a non-target species for two different reasons. First, direct exploitation may be too high. Some species, such as elasmobranchs and cetaceans and some structure-building benthos, may only be able to withstand much lower mortality rates than the target fishing mortalities for directed fisheries (see Section 8.2.2.3). Commercial species may, by their nature, be more resilient to exploitation. Specific management targets should be set for the more vulnerable components of the ecosystem. Even low levels of by-catch mortalities for some may require reference points for specific species such as some seabirds and marine mammals. This is because of their inability to withstand high mortality rates or their potentially high vulnerability to incidental mortality due to at least periodically forming very large aggregations. Secondly, because the EU management sets single species TACs, a fishery targeting a mix of commercial species, therefore, may continue fishing, and thus generate additional mortality on commercial species, as long as not all TACs are taken. ICES acknowledges this potential problem in the text of the annual advice. However the estimation of and application of single species reference points may have to include aspects of multispecies relationships explicitly to provide high likelihood of achieving conservation objectives of stocks taken in mixed fisheries. In the discussion below, these considerations will be developed for potentially relevant species.

Downward or upward trends in populations of many non-target species have been shown for the North Sea and other intensively fished areas (Heessen and Daan, 1997; Anon., 1997). Still not all these species are suitable as a potential reference point in an ecosystem consideration in fisheries management because, to be useful as a reference point, it is desirable to have a very well-defined and clear relation of stock status with fishing activities. Otherwise it will not be possible to formulate effective management measures. The status of top-predators, species which serve as main sources of food, structure-building organisms or representatives of a vulnerable group of species may be particularly useful as reference points. From recent ecosystem and fisheries research, two potential indicator species will be reviewed as an example of potential reference points, the harbour porpoise and the thornback ray.

The most abundant cetacean in the North Sea and the Baltic Sea is the harbour porpoise (*Phocoena phocoena*). They are distributed throughout the North Sea, but are no longer present in the Southern Bight of the North Sea, the English Channel, or in much of the Baltic Sea. Incidental catches of harbour porpoise have been reported from almost every type of fishery in the North Sea. But bottom-set nets generate the great majority of harbour porpoise by-catch in the ASCOBANS area. Vinther (1994) estimated the annual by-catch in the Danish gillnet fisheries in the central and southern North sea at slightly more than 4500 animals.

A large shipboard and aerial survey (Small Cetacean Abundance in the North Sea, also known as SCANS) was made in 1994. The abundance of harbour porpoises in the North Sea, including the Channel and the Kattegat, was estimated at 304,000 (242,000–384,000) animals in 1994 (Anon., 1997). Of this total, the North Sea population of 170,000 occur in the central and southern North Sea. Genetic studies indicate this unit should be treated as a separate management unit. The harbour porpoise is specially protected under a number of international agreements and directives. The International Whaling Committee (IWC) recommends that a by-catch mortality rate of 1 % should lead to research and expression of concern. Mortality exceeding 2 % should lead to immediate implementation of management actions in order to reduce by-catch. For the central and southern North Sea, a maximum allowable by-catch of 3400 animals per year would be a sound ecological reference point related to fisheries. If this reference point was already operational, the current estimated annual by-catch of just a part of the fisheries in this region would exceed this biological reference point and effective management measures would be required immediately. Recent by-catch studies in the Celtic Sea estimated the fraction of harbour porpoises caught in fisheries to be 6.2 % of the total population size which would also be nonsustainable (Tregenza *et al.*, 1997). Equal use of the 2 % by-catch of harbour porpoises in this area would lead to a maximum of 725 allowed by-catches per year for the Celtic Sea instead of the current estimated annual by-catch of 2200 animals (Tregenza *et al.*, 1997).

A second example of a potential species for which an ecological reference point could be described is the thornback ray (*Raja clavata*). Rays and skates have a cartilaginous skeleton and, together with the sharks, belong to the group of elasmobranchs. This group of species have life history strategies which fall in the realm of the so-called K-selected species of the classic r/K selection theory (Hoenig and Gruber, 19xx). This strategy consists of large adult size, late reproduction, and production of few, well-formed young (Table 3.8.1), which makes the species vulnerable to additional mortality such as mortalities caused by fisheries. Rays and skates are a by-catch of demersal fisheries and all species have a commercial value except for the starry ray (*Raja radiata*), which is invariably discarded. Landings of all skate and ray species together decreased from around 18,000 t after both World Wars to the low level of 5,000 t around 1975 and has remained at this level since (Figure 7.2.4.2). Taking into account the increase in fishing effort in the North Sea over recent decades, the decrease in biomass is even more severe (Rijnsdorp *et al.*, 1996). Not all ray species are equally affected by commercial fisheries and species can be classified according to their vulnerability to fishing based on information in age at maturity and fecundity (Table 3.8.1). Fisheries independent data confirm this. The common skate (*Raja batis*) has virtually disappeared from the North Sea between 1930 and the present, while the starry ray has increased in abundance and seems to stay within safe biological limits (Walker, 1996; Walker and Hislop, in press; ACFM, 1997). The thornback ray is the most common species at the fish market and although this species has virtually disappeared from Dutch and Belgian coastal areas (Walker, 1996), it is still resident along the British coast around the Wash and Thames estuary (Walker and Heessen, 1996; Rogers *et al.*, 1997). Historical tagging data has shown that this coastal area is important for mating and spawning (Walker *et al.*, 1997). The thornback ray may serve as a biological reference point because it is still abundant enough to collect statistically valid information.

Table 8.2.2.1. Life history characteristics of five resident North Sea ray species (table from ACFM, 1997).

	L_{inf}	L_{mat}	A_{mat}	Fec	$Z_{r=0}$	Z_{est}	Rank
Common skate *Raja batis*	237	160	11	40	0.38		1
Thornback ray *Raja clavata*	118	86	10	140	0.52	0.60	2
Spotted ray *Raja montagui*	79	62	8	60	0.54	0.72	3
Cuckoo ray *Raja naevus*	75	56	8	90	0.58	0.69	4
Starry ray *Raja radiata*	71	39	5	38	0.87	0.79	5

(L_{inf}: maximum length; L_{mat} and A_{mat}: length and age at first maturity, respectively; Fec: nr of eggs produced per year; $Z_{r=0}$: maximum mortality that species is able to withstand; Z_{est}: estimated level of mortality based on recent survey catches; Rank: ranking in decreasing order of vulnerability)

In the North Sea, the thornback ray is caught as by-catch in demersal fisheries. Fishing mortalities of commercial species are high, ranging from 0.5–0.8 or even higher. Since the catchability of rays is high for these kinds of fisheries, similar fishing mortality rates can be expected. But thornback rays are known to form local subpopulations (Walker and Heessen, 1996). These do not have to coincide with the areas where the demersal fisheries put their highest effort. A reference point for the thornback ray should take into account these spatial aspects.

Based on the life history strategy characteristics, the maximum total mortality the thornback ray population is able to withstand, $Z_r = 0$, is calculated at 0.52 (Table 3.8.1). In order to ensure the continued existence of the thornback ray in the North Sea, the total mortality in areas where sub-populations of thornback ray still occur should be kept below a level of 0.52. Tag experiments show that thornback rays are resident and do not migrate over large distances (Walker *et al.*, 1997). This supports the effectiveness of area-specific measures. ICES already advises to limit the impact of demersal fisheries particularly in those areas where the species still occurs, this may be necessary to protect the stock in the North Sea (ACFM, 1997).

Thus, area-specific maximum mortality seems a suitable and effective reference point for the thornback ray. For accurate estimation of fishing mortality, a major and controllable part of the total mortality, improved data on landings (species specific), discards (juveniles), and disturbance of eggs by demersal gears is necessary, and requested by ICES (ACFM, 1997). With this kind of information it is possible to formulate the most effective fisheries measures in the areas of concern.

8.2.2.3 Reference points for ecologically dependent species

For some years CCAMLR has explored the important role of krill in the Antarctic ecosysem. The breeding success and even survivorship of a number of predators, including several species of seabirds and marine mammals, is affected greatly by the status of krill (Laws, 1984; Croxall and Prince, 1987). Correspondingly, the requirements of these ecologically dependent predators plays a major role in the management of krill fisheries in that region (SCCAMLR, 1992). Recently the Scientific Working Group of CCAMLR reviewed what would be a precautionary approach to the management of krill fisheries, in light of the expanding ideas about the precautionary approach and progress in the development of reference points. The associated analyses indicate that although a precautionary overall catch limit is necessary for large geographic areas, that limit is not sufficient to safeguard some of the dependent predators. A management approach is proposed which requires geographic subdivison of the overall catch according to varying requirments of predator populations, and uses information on predator populations and their physiological needs in setting harvest levels (Everson and de la Mare, 1996). The proposal does not go as far as proposing specific biological reference points for the ecologically related species and relating those reference points directly to krill management. However, the approach lends itself directly to those developments, and such reference points may be forthcoming in future publications from CCAMLR scientists.

Closer to home, ICES has received requests for advice about possible management measures which might be necessary to protect local aggregations of sandeels near sensitive wildlife concentrations. This issue is discussed in depth in Section 9 and thus will not be reviewed here. The request clearly stems from the same concern; there may be ecologically related species whose conservation is not assurred by a management approach that places the stock being targeted at negligible risk overall. Also, the fishery for capelin in the Barents Sea is managed under an approach which gives the feeding requirements of cod (and other predators?) priority over human harvests.

Specific types of biological reference points have not been proposed for such ecologically related species, nor have the links between the reference points and specific management actions been specified. Nontheless, in at least a few cases, such as colonial seabirds and their prey fish stocks, cod and capelin, and Antarctic top predators and krill, the relationships have been studied extensively, and the management needs are recognized. The knowledge base might be an adequate foundation for development, testing, and implementation of such reference points linked among species.

8.2.2.4 Reference points for species affected by scavengers feeding on discards and offal

Populations of many scavenging seabirds have grown in recent years (e.g., Lloyd *et al.*, 1991). Some of this growth may be due to recovery following a long period of persecution which ended in the early part of the current century, but it is likely that much of the growth of the populations of some species is due to the increased food supply deriving from fishery wastes (e.g., Fisher, 1952; Furness and Barrett, 1985). This growth appears to be continuing in many populations.

Owing to the requirement of seabirds to breed in areas that are free (or virtually free) of mammalian predators that can take eggs or young, there is frequently competition for the limited habitat that meets this requirement. In many cases, this leads to displacement either into nearby suboptimal habitat or away from the area entirely (Howes and Montevecchi, 1993). This displacement in many cases may not be desired by local wildlife managers (and may locally reduce biodiversity). Many of the tern species have been shown to have been displaced by larger gull species (Theissen, 1986; Becker and Erleden, 1986). This has led in many instances to the culling of the large gulls in order to allow terns to return to their original nesting sites (Wanless, 1988; Wanless *et al.* 1996). In Shetland, the great skua population has grown rapidly and was feeding on both sandeels and fishery waste. The availability of sandeels has declined around the Shetland Islands (trends in discard amounts are not known), and the great skua population has now switched to depredating seabirds and their young (Heubeck and Mellor, 1994). Previous regulation of the availability of offal and discards might have limited the growth of the population of great skuas.

Fisheries managers might thus consider reference points addressing discards and offal deriving from fishing operations.

8.2.2.5 Summary of reference points at the species level

Suppose that biologically sound reference points for genetic diversity were added to the existing B and F reference points, and that reference points were also identified for all non-target species and for species ecologically dependent on aggregations being fished. Furthermore, suppose that fisheries complied with these reference points, such that there was a high likelihood of achieving all single-species conservation objectives. Would conservation and sustainability of the ecosystem be achieved with at least an equal likelihood? If the answer to this core question is 'No', there are two ancillary questions. First, what multispecies properties might still be at an unacceptable level of risk? Second, how should these properties be monitored and/or modelled, in order to identify and evaluate the effectiveness of actions taken to reduce the risk?

8.2.3 Biological reference points from an ecosystem perspective

The answer to the first question, raised in Section 8.2.2.4, is that we do not know if conservation and sustainability of the ecosystem as whole would be achieved. We do know that without question fishing has changed the size composition of fish in some, possibly many, exploited systems (Pope and Knights, 1982; Pope *et al.*, 1988; Dayton *et al.*, 1995), and in the North Sea in particular (WGECO Report 1996; Rice and Gislason, 1996). Regardless of the trophic model considered, changing the size composition of predators in the ecosystem has, with high likelihood, changed the way that predation pressure is distributed among lower trophic levels in the ecosystem. The uncertainty is in the magnitude of the change, and its consequences for the ecosystem. We also know that the flux and residency of nutrients within the system must also have changed, as the numbers and biomasses at different trophic levels as well as features of benthos have changed (Rowe *et al.*, 1975; Prins and Smaal, 1990). Again, it is the magnitude and ecosystem consequences which are uncertain. Even if present knowledge is inadequate to answer the first question, it is adequate to highlight that a truly precautionary approach with the possible consequences, as outlined below, should be of serious concern.

A number of multispecies or ecosystem models have been developed which can be used to investigate this question. At this time, though, different models make very different predictions about ecosystem consequences (or lack thereof) of changing the distribution of predation pressure among sizes (and undoubtedly species) of prey. We also know too little about the flux of nutrients at lower trophic levels, and among the benthic, pelagic, and demersal parts of the ecosystem, to know even how the flux of nutrients has changed as a result of reducing the numbers and biomasses of large predators, let alone the consequences of the changes. Therefore, it is premature to draw inferences about impacts of changes in size composition of predatory fish on the sustainability and conservation of the larger ecosystem as a unit, and on the larger question of the need for additional precautionary reference points.

Primary production in marine ecosystems away from the coastal zone are generally controlled by the availability of nutrients and usually nitrogenous forms. In stratified regions, the rate controlling step is the regeneration of nutrients by zooplankton and fish excretion of ammonia. In vertically well-mixed areas, the flux of nutrients from the benthos is also important, decomposers in the benthos being responsible for the ammonification of organic nitrogen, and the reduction of nitrate to ammonia (Sørensen, 1978). High productivity of coastal waters may be dependent on this benthic-pelagic coupling (Rowe *et al.*, 1975). The flux rate of this coupling is dependent on the biological activity in the sediments and, in particular, the nature of the benthic fauna (Prins and Smaal, 1990; Josefsen and Schlüter, 1994). Fishing has the potential to alter these rates by (i) alterations in the benthic fauna, (ii) re-suspension of benthic materials by towed bottom gears, (iii) alterations in the chemical status of bottom sediments, e.g., exposure of anoxic materials, and (iv) alterations in the size of the various food web compartments.

Although we cannot evaluate the likelihood of achieving ecosystem-level objectives using a strategy of achieving all single-species conservation objectives, we do note some important considerations with regard to ecosystem-level reference points. First, it is well established that the dynamics of individual stocks and populations connected trophicly contain time lags and buffers (e.g., age structure, density dependent growth) which can slow down the rate at which the consequences of perturbations of a food web may be manifest. Therefore, we may not yet be observing the full impacts on the ecosystem of past levels of fishing. Moreover, if there were to be changes in major ecosystem properties, most models suggest the changes could be difficult and slow to reverse, and would aggravate the loss in total yield of fish, beyond the yield already foregone due directly to overfishing the target stocks.

Although we are not in a position to recommend that ecosystem reference points are necessary, beyond the reference points which would assure sustainability and conservation of all populations killed directly by fishing, neither are we prepared to confirm that single species reference points are enough to ensure a precautionary approach. This is a complex problem, with important implications, and much more investigation of model (and ecosystem) dynamics is required. For example, although WGECO has clearly documented that the slope of the biomass spectrum of the North Sea has changed over the past 20 years, we cannot advise what a maximum tolerable slope would be, what a 'good' target slope would be, or even if these are reasonable concepts to consider.

A commitment to a precautionary approach to fisheries management and conservation of biodiversity has to include a commitment to pursue these types of questions much further. Relevant programmes would have to identify:

a) what ecosystem properties require more than just the conservation of the individual component species?

b) which of the properties in a) could be placed at risk by fisheries?

c) what management measures would be necessary to have a high likelihood of achieving conservation of the properties in b)?

d) how could the properties potentially at risk be measured and monitored?

Some of these questions have fueled research and debate among community ecologists for decades, and quick resolutions are unlikely. Future meetings of WGECO could address the state of knowledge on these questions more intensively, but would require attendance by diverse specialists, and the opportunity to focus significant time on these questions. However, WGECO stresses that the need for some ecosystem level reference points is real. Even if different theoretical frameworks suggest different properties for ecosystem level reference points (often just because the different frameworks use different biological 'currencies'), in internally consistent ways, every framework indicates that such properties exist (see Section 8.3).

8.3 Models that may give Insight

In relation to fisheries impacts, much of the discussion on the implications of using the precautionary approach has focused on how to define target and limit reference points using traditional single species fisheries models to make predictions of impacts on target species (e.g., ICES CM 1997/Assess:15; ACFM Report, 1997, Part I).

The International Whaling Commission uses single species models to provide advice on sustainable levels of harvest of cetaceans. The nominal catch limits derived by the revised management procedure (RMP) are based on a comprehensive specification of data requirements in terms of catch history and abundance estimates, the algorithm for calculating catch limits, including a specification of the population model to be used, how it is fitted to the data, and well defined rules specifying how uncertainty should be taken into account (IWC, 1993). A similar approach has been proposed for small cetaceans in the North Sea (Barrington *et al.*, 1997).

For seabirds and benthos, reference points have not been set and, in particular for benthos, the present knowledge has, with few exceptions, not yet crystallized into models which could readily be used to predict consequences of fishing for individual species or assemblages.

WGECO has previously used the concept of potential jeopardy as a common yardstick to identify particularly vulnerable species in relation to fisheries generated mortality. This approach is closely related to the approach followed in fisheries management where limit reference points in relation to spawning stock biomass such as MBAL have been used. Potential jeopardy is defined as the additional mortality needed to decrease the spawning stock biomass of a certain species to a specific level, say 5 % or 10 %, of its virgin unfished value. The concept can be applied to calculate the vulnerability of individual species across taxonomic groups. It depends only on life history parameters of the particular species, i.e., on growth, mortality, and age or size at first maturity. However, data to estimate the actual mortality imposed are seldomly available and little is known about how life history parameters for particular species would respond to changes in the physical environment, in the amount of food available, and in the abundance of their predators.

Less effort has been spent on investigating how reference points could be defined by models which allow the species to interact. Multispecies fish stock models include species interaction in the form of fish predation and are available for some areas, but have rarely been used for providing management advice. Some of the multispecies models have been extended to include marine mammals and seabirds. Often this has been in terms of the impact mammals and seabirds have on commercially exploited species, only very rarely has the reverse question been asked. At present, the models are therefore of limited use for defining reference points in relation to fisheries generated food limitation for seabirds and marine mammals. However, simpler models have been used to estimate exploitation levels on prey species which take the needs of their predators into account, e.g., the models used to arrive at precautionary catch limits for krill in the Antarctic (Everson and de la Mare, 1996).

Few models describe how community or ecosystem properties would change in response to fishing, and often the existing metrics, such as species diversity indices or slopes of size spectra, are difficult to connect to the perceived state of the affected system. For this reason, such metrics have not yet been used to define limit and target reference points. The models that are available describe either overall metrics such as the slope of the size composition of the fish assemblage, or consider energy flow among trophic compartments. Of the latter type, mass-balance models, such as ECOPATH (Section 8.3.2), offer a range of possible measures that could be used for defining reference points. Another possibility is to utilize more conceptual tools, such as trophic cascade models (see Section 8.3.3). However, in both instances, the challenge is not to derive the metric, but to relate it to changes in the affected system of relevance to society.

8.3.1 Extensions of MSVPA/MSFOR

At its most recent meeting, the ICES Multispecies Assessment Working Group (MAWG) discussed how to derive reference points in a multispecies context (ICES CM 1997/Assess:16). Several modelling approaches were investigated including classical Lotka-Volterra models, MSVPA/MSFOR approaches, and single species models with changes in natural mortality due to predation. The investigations demonstrated that reference points derived from single and multispecies models can be expected to differ and, in particular, that single species reference points often will tend to be less conservative (and less precautionary) than their multispecies equivalents.

At this meeting, an extended version of the Baltic multispecies spreadsheet MSFOR-type model used at the MAWG meeting was available. The model includes cod, herring, and sprat in the central Baltic and performs a 32-year prediction of the biomass and yield of the three species with an annual timestep. The relationship between spawning stock and recruitment is of the Ricker type, and the model includes a description of how growth and maturity of cod changes in response to changes in the amount of available food. The input data are derived from the database used by the Working Group on Multispecies Assessment of Baltic Fish (ICES, 1996/Assess:2, 1997/J:2, 1997/Assess:12) (residual natural mortality, fishing mortality, suitabilities, weight at age, maturity at age, recruitment). The model predictions should therefore be in reasonable accordance with similar predictions made by the MSFOR used by the Working Group on Multispecies Assessment of Baltic Fish even though this model operates with a quarterly timestep. However, the model parameters describing changes in growth as a function of available food have not yet been estimated from retrospective runs. At the present stage, the model is therefore intended as a conceptual tool which can be used to demonstrate how competition and predation will affect precautionary reference points and not as a model from which management advice can be directly derived.

The model is able to run in three different modes corresponding to the classical single species fisheries model (constant natural mortality and growth for all species), the ordinary multispecies model (MSFOR including cod as a predator on herring, sprat, and young cod, with constant weight at age for all species), and an extended multispecies model where the amount of herring, sprat, and other food available will influence cod food intake, growth, and maturity at age. The extended version was made in order to take the large changes in cod weight at age observed over the period 1977–1996 into account, assuming that these changes were due to changes in the food supply of cod. Figure 8.3.1.1 shows how the average weight at age for ages 2 to 4 changed from between approximately 30 % below the long-term average to 10 % to 30 % above the long-term average at the end of the period. Figure 8.3.1.2 shows the change in average weight at age versus cod biomass.

In the single species version, recruitment to all of the species are modelled by Ricker curves, with parameters estimated from historic values of stock and recruitment. Natural mortality is constant at values equal to the sum of predation and other natural mortality (M1) in the multispecies status quo situation. In the multispecies models, cod recruitment at age 0 is assumed to be directly proportional to spawning stock biomass. Subsequent changes in cannibalism changes the number of cod surving to age two. Survival is thus lower at high levels of adult cod biomass producing a stock recruitment relationship similar to the Ricker model used in the single species case.

In the ordinary multispecies model cod is predating on herring and sprat as well as on their own young. The amount of other food available to cod is assumed to be constant irrespective of a change in cod biomass and intake of other food.

In the extended multispecies model, the annual growth of cod is assumed to be directly proportional to the amount of food available. The biomass of other food is modelled by a surplus production model of the Fox type (Biomass of other food = $1/q$ * exp(a + b* Biomass of cod)) where the cod's intake of other food in the status quo situation and the value of other food assumed in the ordinary multispecies run (30 million t) are used to estimate the q and b parameters, and the constant, a, is fixed at a value producing a biomass of other food which is 10 million t higher in a situation without cod predation. The latter value was adopted because it produced what appears to be sensible values for cod weight at age at high biomasses. In the status quo situation, the parameters are such that the weight at age of cod corresponds to the weight at age used in the single species and ordinary multispecies models. Changes in weight at age will influence the proportion mature at age. Based on historic data on maturity and weight at age, the relationship between weight and maturity at age is modelled by Maturity = (1 – exp(– c*W))^d, where W is weight and c and d are constants.

The fishery is controlled by two variables, 'cod effort' and 'pelagic effort', that are used to multiply the fishing mortalities for cod and for herring and sprat, respectively. In the status quo situation where both effort variables are set to 1.0, the average fishing mortality for cod ages 3–7 equals .82, while for sprat ages 3–5 and herring ages 3–8 the status quo fishing mortalities equal .15 and .27, respectively.

The initial population numbers in the starting year are set equal to the long-term equilibrium population sizes in the status quo situation in order to ease comparisons between this situation and a change in the fisheries. The results from a run where both fisheries were closed (cod effort and pelagic effort both reduced to 0.001) are presented in Figures 8.3.1.3 to 8.3.1.5. A closure is predicted to lead to damped oscillations in spawning stock biomasses resulting in a long-term increase in the biomass of cod and a long-term decrease in the spawning stock biomasses of herring and sprat. Cod weight at age will decrease, and so will the proportion mature at age.

The average yield and spawning stock biomass of cod predicted in each of the three models are shown in Figure 8.3.1.6 for various levels of cod effort. Pelagic effort was fixed at 1.0 and the values presented in the figure are averages over the last 10 years of the 32-year prediction period. In the status quo situation, the predictions of the three models are identical. When cod effort is decreased from the present level, the biomass and yield of cod increases. This increase is most pronounced in the single species prediction, less so for the ordinary multispecies mode where recruitment is reduced by cannibalism, and even less for the extended multispecies prediction, where the increase in cod biomass is counteracted both by cannibalism and by reductions in weight at age with knock-on effects on maturity and recruitment.

The model was used to examine how biomass reference limits might be derived in a multispecies context. Figure 8.3.1.7.a–8.3.1.7.c show plots of the regions of combinations of 'cod effort' and 'pelagic effort' that produces spawning stock sizes for all three species above or below 10 % of their unfished levels (calculated by closing both fisheries) in each of the three modes of the model, e.g., an SSB for cod of 2.0, 1.4, and 0.9 million t in the single, ordinary multispecies and extended multispecies modes, respectively.

The single species results are shown in Figure 8.3.1.7.a. The area within which the spawning stock biomass of all three species is above 10 % of the unexploited level forms a rectangle in the lower right corner of the plot. The present situation (both effort multipliers = 1.0) is right at the upper border of the area. Pelagic effort can be increased to between two and three times its present level before the herring SSB will fall below the reference limit. Increasing both efforts above the limits will generate an area where only the sprat SSB is above the limit.

In the ordinary multispecies predictions, the limits for the pelagic species become curved, Figure 8.3.1.7.b. The amount of pelagic effort which can be exerted without reducing the pelagic species below the limit now depends on the effort in the cod fishery. If cod is reduced to low levels by an intensive fishery, it is possible to increase the effort in the pelagic fisheries to approximately four times the present level before herring falls below the limit. If the cod fishery is closed and the cod stock increases, the reference limit for herring is close to the present level of effort.

In the extended multispecies case, all of the reference limits are curved, Figure 8.3.1.7.c. In this case, the limits for cod become dependent on the amount of pelagic effort. If pelagic effort is high, cod can sustain less effort before it exceeds the limit. This is because of a reduction in cod growth and proportion mature. If there is plenty of food for cod, i.e., large stocks of herring and sprat, cod will grow faster and mature earlier, and hence tolerate a more intensive fishery.

Species interactions will alter reference points and limits. Reference points for fisheries on forage fish cannot ignore changes in the biomasses of predators feeding on these species. Reference points for fisheries on predators cannot be set without considering how the predators are influenced by the simultaneous exploitation of their prey.

8.3.2 Mass-balance models

A number of metrics were highlighted earlier in this report (Section 3.3.2) which are of potential relevance for developing reference points from a multispecies perspective, based on mass-balance models of trophic interactions in ecosystems. These include:

- the trophic level of the fishery in an ecosystem;
- the transfer efficiency between trophic levels;
- Finn's cycling index;
- the primary production required to sustain fishery catches;
- mixed trophic impact analysis of the ecosystem, with the fishery as impacting and impacted component.

At present, no recommendations can be made as to how these analytical tools can be used for the definition of reference points. However, their step beyond single-species fisheries management towards explicitly considering the multispecies context in which the fishery operates may contribute to future ecosystem management.

8.3.3 Trophic cascade models

The central role of fish in limnic ecosystems, especially their influence on food web structures, has been known since the early 1960s (Hrbacek, 1962; Brooks and Dodson, 1965; review in Hansson, 1885). In the 1980s, research in this field increased significantly (e.g., see Carpenter, 1988; review by Northcote, 1988) and the concept of cascading trophic interactions (Carpenter *et al.*, 1985) has been heavily discussed (e.g., Carpenter and Kitchell, 1988; McQueen and Post, 1988a, b; Leavitt *et al.*, 1989; Brönmark *et al.*, 1992; Martin *et al.*, 1992; Christoffersen *et al.*, 1993; Schindler *et al.*, 1993). By cascading trophic interactions, we mean that, e.g., a top predator like a piscivorous fish does not only influence the ecosystem by reducing the abundance of its prey, but also indirectly influences the food organisms of this prey. For example, if the prey fish is an important zooplanktivore, the predation by the top predator may reduce the predation pressure on zooplankton. The effects of the predation from the top predator cascades down the food web: the decreased predation on zooplankton may allow these to increase in abundance and hence increase the grazing pressure on phytoplankton. These trophic dynamics generally follow the classical food web interaction concept of Hairston *et al.* (1960).

Most of our present knowledge on the role of fish in aquatic ecosystems, in particular their significance as predators and their influence on trophic cascades, derive mainly from studies in lakes. For marine ecosystems, these ecological interactions are much less understood. This is probably because marine systems are more difficult to study than lakes. Compared to the seas, lakes are well defined and geographically delimited ecosystems. Furthermore, there are thousands of lakes with different food web structures that can be compared to evaluate the consequences of these differences. The relative lack in our understanding of the role of fish marine food webs does not, however, imply that the significance of fish predation is less than in freshwater. Nixon (1982) actually showed that at a given primary production, fish yields from marine systems are generally higher than those from freshwater systems. This implies that marine food webs are at least as tightly coupled as those of freshwaters and that fish predation are also central in structuring marine ecosystems.

Several studies have shown that fish predation on zooplankton is intensive in marine systems (e.g., Fulton, 1983; Kimmerer and McKinnon, 1989; Hansson *et al.*, 1990; Hassel *et al.*, 1991; Hopkins and Gartner, 1992; Rudstam *et al.*, 1992; Arrhenius and Hansson, 1993; Luo and Brandt, 1993). There is also a number of articles which describe ecological effects of fish predation on organisms other than their prey and hence supports the presence of cascading trophic interactions or other complex ecological population dynamics processes in marine ecosystems (Skjoldal, 1989; Springer, 1992; Rudstam *et al.*, 1994; Parsons, 1991, 1992, 1996; Anon., 1996; Verity and Smetacek, 1996; Shiomoto *et al.*, 1997; Hansson *et al.*, in press). A direct implication of these results is that the intensive fishery for many common marine fish species is likely to influence marine ecosystem structures, and not only decrease the abundances of the target fish species.

Trophic cascading models have been successful in describing the responses of lower trophic levels of lacustrine systems to perturbations at upper levels. With suitable development of this application in marine systems, this type of model might become useful as a tool for identifying fishing strategies which have a high risk of causing amplified perturbations at lower trophic levels than those being fished. The associated ecosystem reference points might be tolerance limits on perturbations that fishing could impose on any single trophic level or on the suite of levels in the system being modelled. An example of a possible ecosystem reference point is that the relationship between abundances of piscivorous fish and their forage species must be kept within certain limits. Hence, a goal in fisheries management should be to avoid not only growth and recruitment overfishing (Cushing, 1975), but also ecosystem overfishing (i.e., ecosystem changes that drastically change trophic interactions, food web structures, nutrient cycling, etc.).

8.4 Concluding Remarks

This section has been developed by starting from existing practice and asking what must be added. WGECO concluded that one necessary addition to present practice is reference points for non-target species, as developed in Section 8.2. WGECO also concluded that the task does not stop here. WGECO notes that, implicitly, present practice assumes that explicit conservation objectives have been set by management agencies, to justify the development of even the reference points used at present. As recent ICES advice makes clear, even that assumption is not absolutely true. Nonetheless, in endorsing the precautionary approach, governments and management agencies have clearly committed to conservation of all species directly or indirectly affected by fishing (FAO, 1995; Garcia, 1996). Much of the internal debate within WGECO centered on what additional commitments are implicit in this approach, because there are strong theoretical reasons to expect that certain ecosystem properties may be altered by fishing activities.

Will society (and biology) be served by objectives to conserve particular configurations of an ecosystem being fished? Do the diverse international agreements summarized by FAO (1995) require such objectives to be adopted? What does it mean for an ecosystem to be 'at risk', and can an ecosystem be 'at risk' if the species which comprise it are not? Although WGECO looks forward to exploring these fundamental questions at future meetings, it stresses that they must be discussed in many other fora as well, both within and outside ICES.

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Figure 8.3.1.1. Percentage deviation from average weight at age of cod in the Baltic Sea, 1977–1996. Data from ICES CM 1997/J:2.

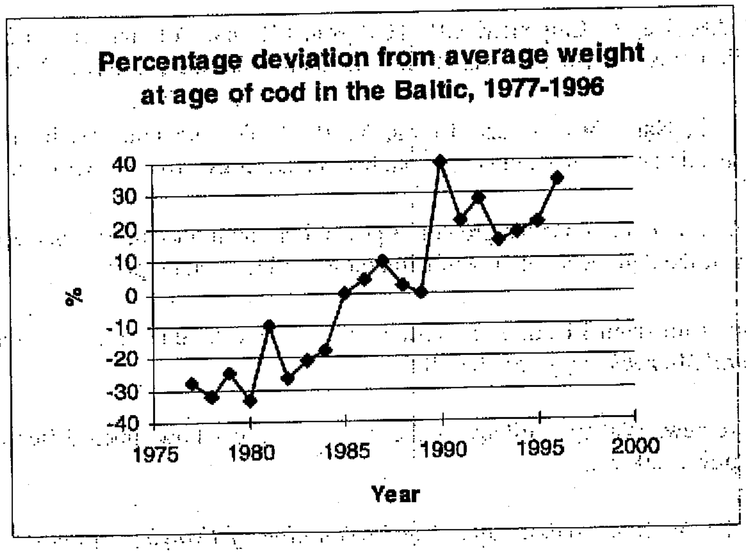


Figure 8.3.1.2. Percentage deviation from average weight at age of cod in the Baltic versus total biomass, 1977–1996. Data from ICES CM 1997/J:2.

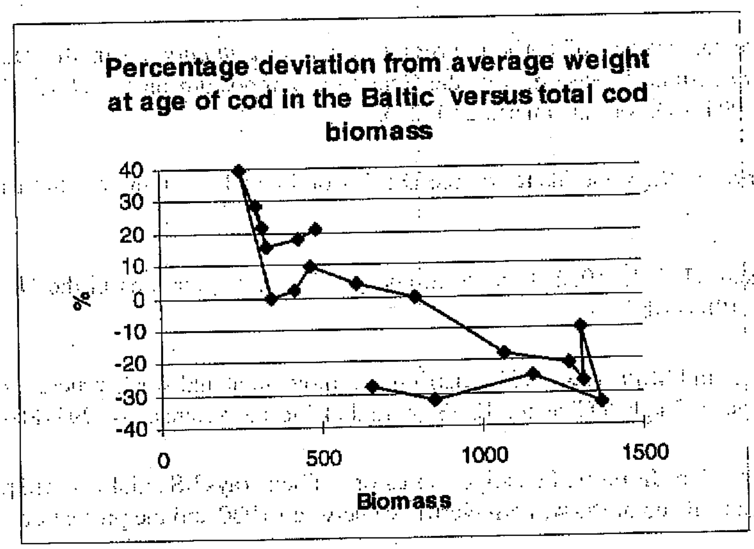


Figure 8.3.1.3. Predicted SSB of cod, herring and sprat after a closure of all fishing. Output from multispecies model with dynamic cod growth.

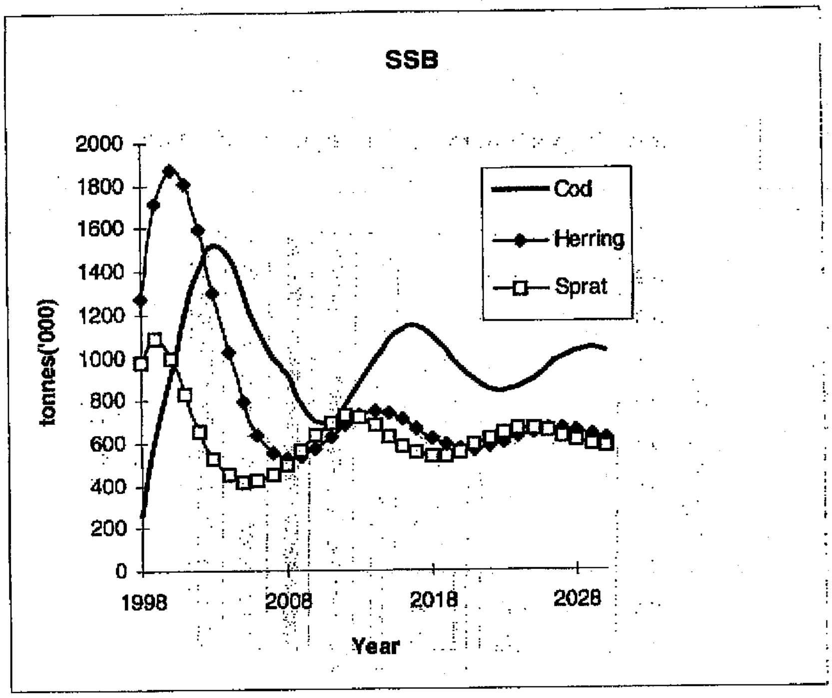


Figure 8.3.1.4. Predicted change in weight at age of cod after a closure of all fishing. Output from multispecies model with dynamic cod growth.

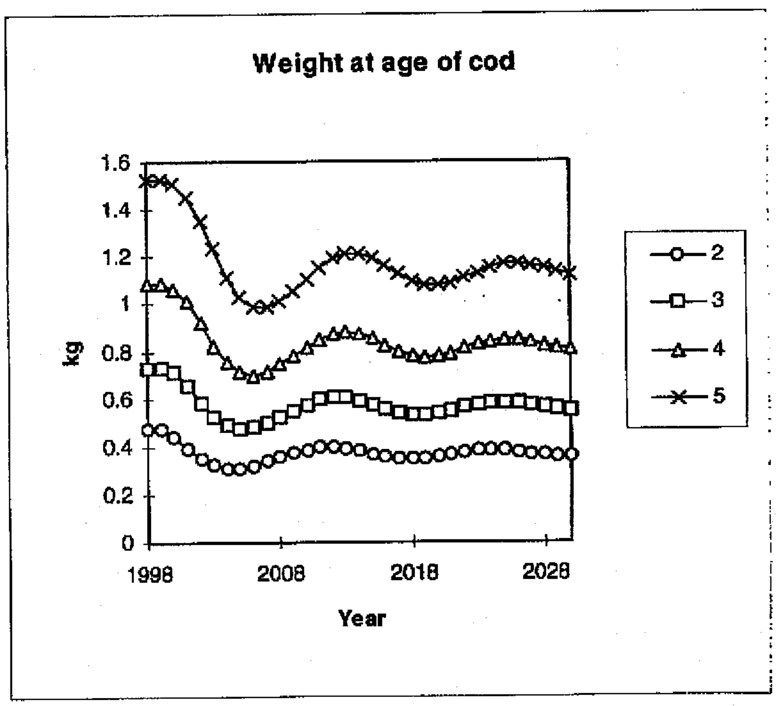


Figure 8.3.1.5. Predicted change in percent mature at age of cod after a closure of all fishing. Values for 1998 correspond to status quo fishing, values for 2030 to final year of prediction. Output from multispecies model with dynamic cod growth.

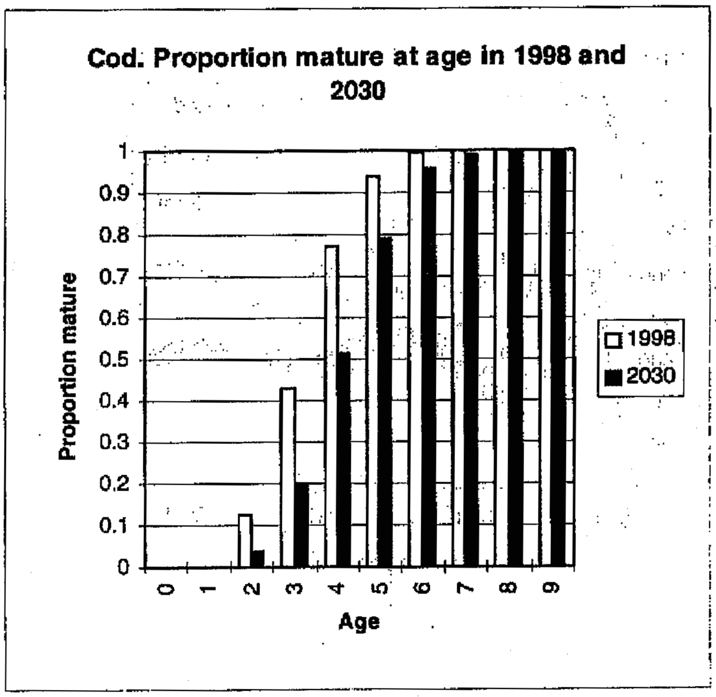
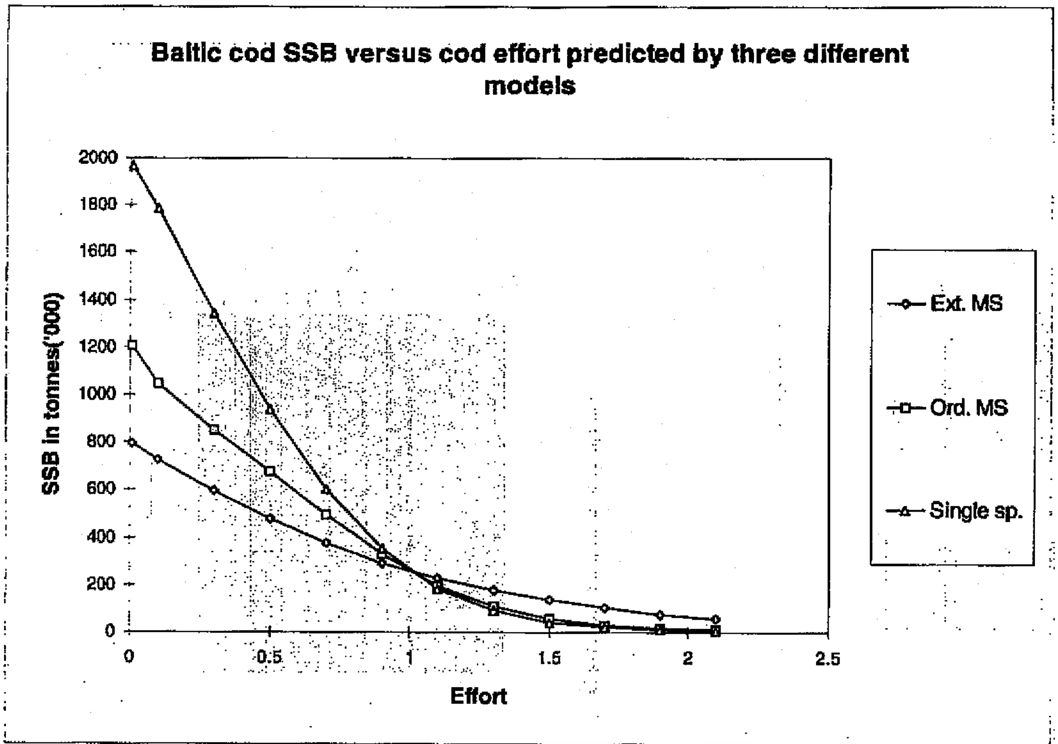


Figure 8.3.1.6. Average SSB and yield of Baltic cod predicted by the single species, ordinary multispecies and extended multispecies versions of the spreadsheet model.



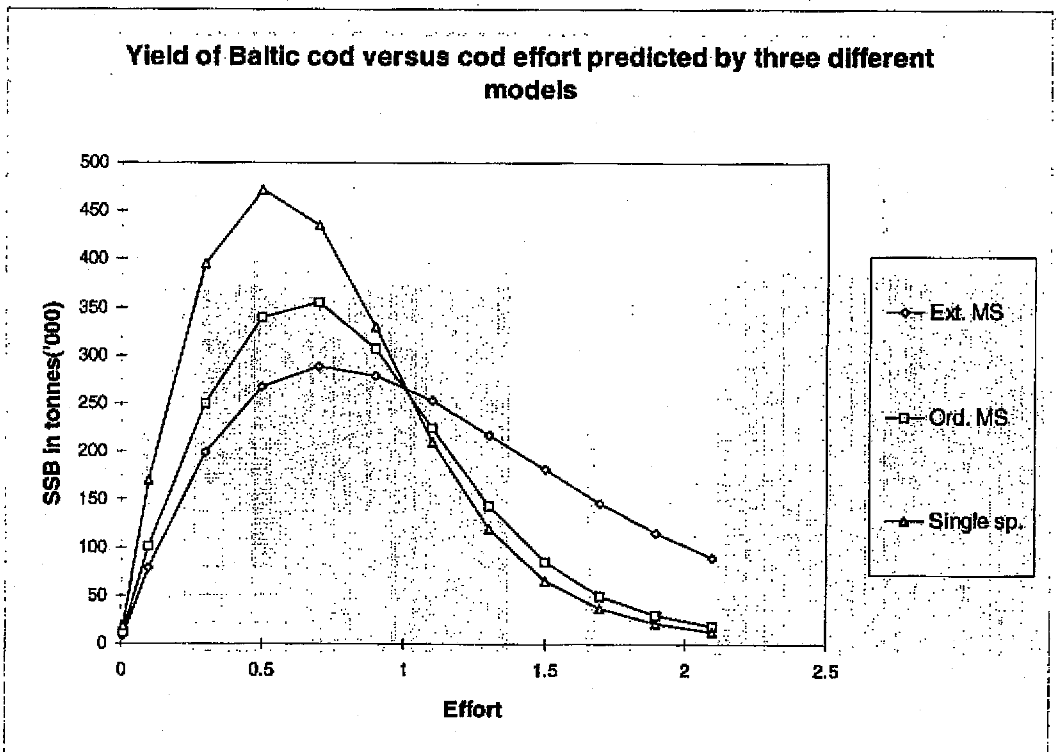
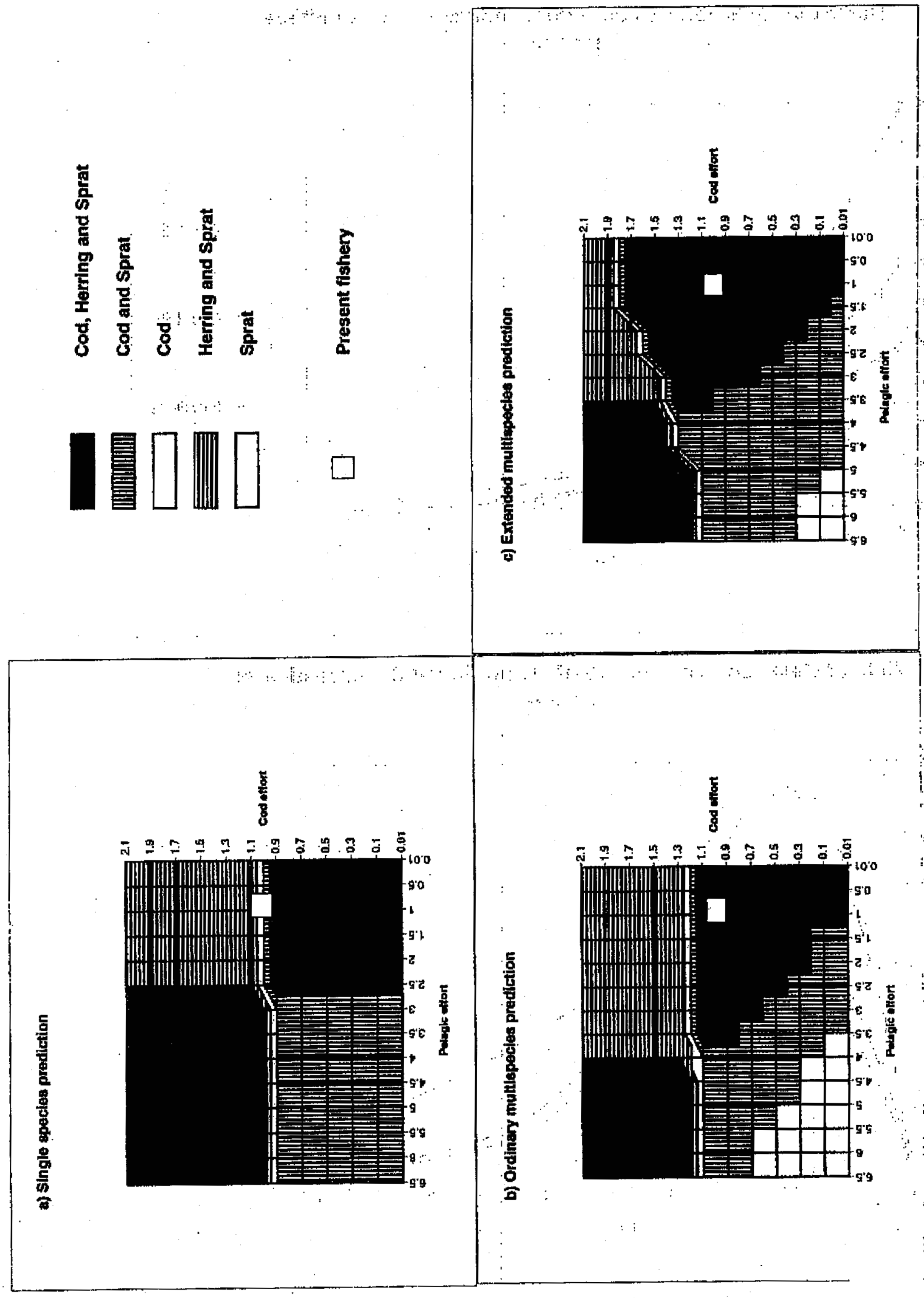


Figure 8.3.1.7. Combinations of effort levels in the cod and pelagic fisheries in the Baltic resulting in equilibrium SSBs above 10 % of the unexploited SSBs (a) for cod, (b) for herring, and (c) for sprat. Predictions assuming single species, ordinary multispecies and extended multispecies model structures.



9 PROTECTING LOCAL AGGREGATIONS OF SANDEELS FOR WILDLIFE ASSEMBLAGES

About one-third of the food taken by seabirds in the North Sea is comprised of sandeels (Tasker and Furness, 1996; Furness and Tasker, in press). Sandeels may form similar proportions of seabird diets elsewhere. These sandeels are not consumed uniformly across the North Sea; many more are taken in the north-western North Sea, and even within this area there are 'hot-spots'. In order to identify and define areas holding sandeels that need protection and to ensure that wildlife assemblages are adequately safeguarded, it is important first to identify the locations of important wildlife assemblages dependent on sandeels. This might be best ascertained by first determining which wildlife species, defined here as those of birds, seals, and cetaceans, are dependent on sandeels and then identifying the locations of important assemblages of these species. Knowledge on the local distribution and biology of sandeels in these areas may then need elucidating before management measures can be proposed.

9.1 Seabirds Reliant on Sandeels

Not all species or concentrations of seabirds rely on sandeels. Diets may also vary seasonally and geographically. Tasker and Furness (1996) assembled information on the diets of the more abundant species of seabird in the North Sea (Table 9.1.1). This compilation did not include some species also known to feed to a large extent on sandeels, but whose biomass in the North Sea is lower than that of the main species. These species include red-throated diver (Eriksson *et al.*, 1990), arctic skua (Furness, 1987), great skua (Furness and Hislop, 1981; Tasker *et al.*, 1985; Furness, 1987; Hamer *et al.*, 1991), Sandwich tern (Pearson, 1968; Dunn, 1972; Veen 1977), common tern (Becker *et al.*, 1987; Boecker, 1967; Dunn, 1972; Frank, 1992; Frick, 1993; Lemmetyinen, 1973; Massias and Becker, 1990; Stienen and Tienen, 1991; Uttley *et al.* 1989), arctic tern (Boecker, 1967; Dunn, 1972; Ewins, 1985; Frick, 1993; Hartwig *et al.*, 1990; Pearson, 1968; Lemmetyinen, 1973; Monaghan *et al.*, 1989; Stienen and Tienen, 1991; Uttley, 1991; Uttley *et al.*, 1989) and black guillemot (Ewins, 1990).

Table 9.1.1. Summary of seabird diets in the North Sea and adjacent areas for most abundant species at times of year when sandeels form an important component in the diet (from Tasker and Furness, 1996).

Species	Area	Period	Diet
Fulmar	North Sea	May-Aug	30 % sandeel (4–10 cm), 30 % offal, 30 % discards, 10 % zooplankton
Gannet	North Sea		30 % sandeels (0-1 group), 30 % herring, 30 % mackerel, 10 % discards
Shag	North Sea	All year	100 % sandeel (5–15 cm)
Great b.-b. gull	North Sea	Apr-Aug	60 % gadoid discards, 20 % sandeels (12 cm), 20 % other prey
Kittiwake	IVa(west)	May-Aug	100 % sandeels (6–14 cm)
	IVb, IVc, IVa(east)	May-Aug	60 % sandeels (6–14 cm), 20 % sprat (8 cm), 20 % zooplankton
Guillemot	IVa (west)	Mar-Aug	100 % sandeel (10–14 cm)
		Sep-Feb	33 % sandeel (10–14 cm), 33 % sprat (10 cm), 33 % gadoids (12 cm)
	IVa (east), IVb, IVc	Mar-Aug	80 % sandeel (10–14 cm), 20 % sprat (10 cm)
		Sep-Feb	40 % sandeel (10–14 cm); 30 % sprat (10 cm); 30 % gadoids (12 cm)
Razorbill	IVa (west)	Mar-Aug	100 % sandeel (6–10 cm)
		Sep-Feb	60 % sandeel, 40 % sprat
	IVa (east), IVb, IVc	Mar-Aug	70 % sandeel, 30 % sprat
		Sep-Feb	60 % sandeel, 40 % sprat
Puffin	IVa (west)	May-Aug	90 % sandeel (0-group), 10 % rockling
		Sep-Apr	30 % sandeel, 30 % gadoids, 30 % sprat, 10 % zooplankton
	IVa (east), IVb, IVc	All year	50 % sandeel, 30 % sprat, 20 % gadoids (all 0-group)

Off Iceland, sandeels form a very high proportion of the diets of razorbills, guillemots, black guillemots, puffins, kittiwakes and fulmars off western, southern and eastern Iceland (Lilliendahl and Solmundsson, 1997) in summer.

Sandeels form a relatively high proportion of the diet (>30 %) of great cormorants, double-crested cormorants, gannets, common and arctic terns, kittiwakes, guillemots, razorbills and puffins in the Gulf of St Lawrence (Chapdelaine *et al.*, 1985; Cairns *et al.*, 1991)

While most seabirds can be relatively flexible in their diets, the distributions of those that feed to a great extent on sandeels might best be examined to locate concentrations. Within the North Sea, these species would include red-throated diver (April to August), shag (whole year), arctic and great skuas (April to August), kittiwake (May to August), Sandwich, common and arctic terns (April to September), guillemot (March to August), razorbill (March to August, particularly in IVa west), black guillemot (whole year), puffin (May to August in IVa west).

9.2 Concentrations of Seabirds

In northwestern European waters, a number of publications have documented concentrations of seabirds (e.g., Lloyd *et al.*, 1991; Stone *et al.*, 1995; Skov *et al.*, 1995; Pollock *et al.*, 1997; Durinck *et al.*, 1994). There are fewer publications outside these waters with Brown (1986) and Powers (1983) being the most recent seabird atlases off eastern Canada and north-eastern USA, respectively.

The location of concentrations documented by these publications may be caused by a variety of interrelated factors. Two factors predominate: the location of colonies and the location of feeding grounds. Seabirds breed in a relatively restricted number of locations and their feeding grounds, by necessity, need to be close to these colonies. Wright and Begg (1997) found that the highest level of aggregation of the northern North Sea guillemot population occurred during the breeding season as opposed to other times of the year. Outside the breeding season, there is less of a requirement to be at the colony, but many species still require relatively high densities of food in order for efficient foraging.

Skov *et al.* (1995) provide the most accessible maps of concentrations of seabird species, these additionally take account of the relative importance of the concentration of each species in a biogeographic context. Unfortunately these maps cover only the North Sea, but there is no reason why their concept could not be extended in the future. Figure 9.2.1 is a compilation of internationally important (see Skov *et al.*, 1995 for definition) areas for shag, Sandwich tern, guillemot, razorbill and black guillemot for the periods when these species appear to utilise sandeels to the greatest extent. It can be seen that a broad area off the east coast of Scotland, along with areas immediately around Shetland, close to the west of Orkney and patches close to other coasts may be particularly important for these 'sandeel dependent' species. This may be further refined to indicate which areas hold more than one of these species at internationally important levels (Figure 9.2.2).

This approach might be further refined in future by using a 'sensitivity index' approach for each species/season similar to that employed by Carter *et al.* (1993) and Webb *et al.* (1995) in relation to sensitivity to oil pollution. Each species could be rated by its dependance on sandeels and the flexibility of its feeding niche at various times of year. An arctic tern would thus have a high sensitivity rating, while a fulmar would be much lower. Added to this rating would be a measure of the importance of the North Sea to the species in a global sense. This score could then be multiplied with an expression of species density in each area in order to factor in the variation in importance of different parts of the North Sea for each species. The sum of these species-area scores for each area would then give a geographic mapping of seabird community vulnerability to changes in sandeel food supply. If information was available on temporal (between year) variation in location of seabirds, this information could be included to the model and added to the maps.

9.3 Seals

There are at least two seal species in North-east Atlantic waters known to include sandeels in their diet (Hammond *et al.*, 1994; Prime and Hammond, 1990). Harbour seals occur on all coasts of the North Sea, on the western coast of Scotland and in northern Ireland, on west Norwegian coasts, in the Skagerrak, Kattegat and western Baltic. The main concentrations are in the Kattegat/Skagerrak, the Wadden Sea, Orkney, Shetland and the west coast of Scotland. Harbour seals remain in coastal waters for much of the year, so the above areas are those where reductions in sandeel stock might have an effect on this species.

Grey seals occur on nearly all coasts north of Brittany in western Europe. In contrast to harbour seals, the species form large breeding concentrations. Notable sites include the Farne Islands off north-east England, the Isle of May off eastern Scotland, some of the uninhabited islands of Orkney, North Rona off northern Scotland and the Monach Islands off the Outer Hebrides. Sable Island off eastern Canada is the largest grey seal rookery in the world, holding 85,000 animals.

Recent studies using satellite transmitters have indicated that grey seals may travel great distances while foraging, but have also shown that relatively small areas of sea close to major rookeries appear to be very important. One such area is in the Farne Deeps to the north-east of the Farne Islands off north-east England. The diet of seals using this area is the subject of current research.

9.4 Cetaceans

There is little published effort-related information on the location of concentrations of cetaceans in the ICES area. Hammond *et al.* (1996) surveyed the North Sea (including the Channel and Kattegat) and Celtic Shelf but sampled at a fairly large scale. While some of their data could be treated at a smaller scale, this would not be uniform for the whole area. Information held on the European Seabirds at Sea database includes data on cetaceans (Northridge *et al.*, 1995), but this is still in the process of being further analysed. Harbour porpoises are known to include sandeel in their diet both off north-western Europe in the Barents, North and Baltic Seas (Rae, 1973; Schulze, 1987; Lick, 1991a, 1991b) and off Canada (Smith and Gaskin, 1974). Aarefjord *et al.* (1995) recorded few sandeel in harbour porpoise diet, but this may have been because their samples were from areas that do not hold sandeels off Scandinavia.

The European Seabirds at Sea database shows that harbour porpoise occur across the whole North Sea and in north-west European shelf waters. The western North Sea holds generally more porpoises than other areas. Specific concentrations were observed off south-west Ireland.

Minke whales seem likely to consume concentrations of sandeels where they co-occur and have been recorded feeding on these to the west of Greenland and in the Moray Firth (Jonsgård, 1982; Santos *et al.*, 1994). Minke whales have been observed in areas with many seabirds that were feeding on sandeels. The only recent descriptions of diets have been from areas mostly outside the areas of occurrence of sandeel (Haug *et al.*, 1996). Schweder *et al.* (1992) found relatively large numbers of minke whales off eastern Scotland in July–August 1990. The European Seabirds at Sea database shows that minke whales occur most commonly in the North Sea over the sandbanks off eastern Scotland and over the north-western Dogger Bank. Other concentrations occur off western Scotland.

White-beaked dolphins are concentrated off eastern Scotland and in the northern Minch in north-west European seas (Northridge *et al.*, 1995, 1997) and bottlenose dolphins occur in nearshore concentrations in places such as the Moray Firth, Cardigan Bay and the Shannon estuary. Bottlenose dolphins have been recorded as consuming sandeels off Galicia and sandeel remains have been found in white-beaked dolphin stomachs off Scotland (Santos *et al.*, 1994), however it seems likely that sandeels are not of great overall importance to these cetaceans (Kinze *et al.*, 1997).

9.5 Sandeel Aggregations

ICES (1997) defined a sandeel aggregation as the sandeel present at a collection of adjacent grounds, some of which may be very small, within an area no larger than a single ICES rectangle. There is however very little published information on sandeel aggregations at this scale in the ICES area, but several projects are under way which should extend knowledge considerably.

At a larger spatial scale, Jensen *et al.* (1994) used information gained from the stomach contents of cod and whiting in the North Sea to describe the distribution of sandeel. These data gave relatively incomplete coverage of the whole North Sea but there tended to be greater quantities of sandeels in cod and whiting stomachs in the western North Sea off Scotland. Wright and Begg (1997) used an unpublished database on presence/absence of sandeels derived from enmeshed fish taken in large-mesh trawl surveys off Scotland. The distribution of sandeels was not described geographically, but Wright and Begg (1997) found that aggregations of guillemots corresponded in the breeding season with areas inhabited by post-settled sandeels, in other words, sandeels were commonest in the western North Sea.

Sandeel grounds have been described at a local scale around Shetland, both for fisheries and for seabird foraging. Seabird foraging areas are frequently smaller than those needed for profitable fishing in this area (Monaghan *et al.*, 1996; Wright, 1996).

Maps of sandeel catch, and catch per unit effort for the Danish fishery in the North Sea have been compiled as part of a wider project by S.A. Pedersen for each month of the fishery for the years 1988 to 1997. These maps show considerable variation both between years and between areas in catches per unit effort but indicate that overall some of the largest catches and highest values of catch per unit effort occur over the sandbanks off the south-eastern coast of Scotland and north-eastern coast of England. The population structure and distribution of the aggregations of sandeels in this area are being investigated by the ELIFONTS project, part funded by EC DGXIV.

9.6 Evidence for Impact of Sandeel Stock Size on Wildlife

The feeding niches of each of these species varies. This in turn will influence the susceptibility of each species to perturbations in their food supply. Some seabirds (kittiwake and the terns) can only feed in the top few metres of the water column, while others (guillemot, razorbill and puffin) can feed throughout the whole water column. Red-throated divers, shags and black guillemots feed near the seabed. Arctic skuas are obligate kleptoparasites on other seabirds carrying fish, while great skuas will take surface-occurring sandeels where available, will also rob other birds of their food and may kill both adults and chicks.

The sandeel stock near Shetland experienced a period of low recruitment in the late 1980s and early 1990s (e.g. Bailey, 1991; Bailey *et al.*, 1991). This change provided an excellent case example of the variation in susceptibility to changes in food supply. During the above period, those seabirds with a relatively narrow feeding niche suffered a total breeding failure, and in some cases declined in number, while the effects on those with a wider niche were fewer and generally less severe. For example, arctic terns frequently laid eggs, but could not rear their young (Monaghan *et al.*, 1989; 1992); their close sympatric species, common tern was able to utilise a wider range of feeding areas and in some cases reared young (Uttley *et al.*, 1989). Kittiwakes failed at most colonies in Shetland (Hamer *et al.*, 1993), while guillemots and shags bred successfully, but exhibited changes in behaviour indicating the foraging had become less successful (in fisheries terms, catch per unit effort had declined). The diet of gannets and puffins changed (Martin, 1989). There was a decline in breeding success of great skuas (Hamer *et al.*, 1991).

From the above, it should be possible to prioritise those seabirds species which may be susceptible to local changes in sandeel stocks, and identify those places where particular care should be taken to ensure that human activities, including fisheries, do not adversely influence this food supply.

Evans *et al.* (1993) identified areas off eastern Shetland that hold regular concentrations of harbour porpoises that appear to coincide with areas known to hold sandeels. Numbers of porpoises in these areas declined during the period when sandeel stocks were at a low level in the late 1980s and early 1990s.

9.7 Evidence for Effects of Fisheries on Sandeel Aggregations

There is little evidence at present to demonstrate any effects of fishing on sandeel aggregations. Partly this is because sandeel populations and aggregations are poorly known. Studies on the relatively isolated sandeel fishing grounds near Shetland in the late 1980s and early 1990s indicated that variations in recruitment, possibly caused by hydrographic variation, were considerably more important than local fisheries in influencing stock size (Wright, 1996; Wright *et al.*, 1996). The ELIFONTS study on the Marr Bank and Wee Bankie off eastern Scotland is attempting to examine the influence of sandeel fisheries in this area.

Two possible time-scales of effect could be caused by fishing. First, in the short-term, there may be local depletion due to harvesting. Depletion, whether natural or anthropogenic, would be likely to cause a decline in breeding success at nearby seabird colonies or a change in distribution, possibly ultimately leading to lower survival rates for seabirds at other times of year, or for the more mobile marine mammals. Decline in seabird breeding success of seabirds which consume sandeels has been noted at colonies in south-east Scotland and north-east England since 1986, but has been particularly low since 1990 (Harris and Wanless, 1997). Large-scale sandeel fishing started 1990 on a small part of the feeding grounds of these colonies in 1990, but has grown very rapidly since then.

Secondly, longer-term effects may depend on persistence of any cause of depletion, and the possibilities for recolonisation from elsewhere. These possibilities would depend to an extent on the population structure of sandeels in an area. There is some evidence of sub-structuring of sandeel populations in the North Sea. Genetic analysis provided no evidence of any population substructuring between sandeel concentrations from near the Outer Hebrides, the Northern Isles and the northern North Sea, but has shown that a sandeel concentration near Fraserburgh did belong to a separate reproductive stock (Verspoor *et al.*, 1994). This finding was consistent with both the knowledge of sandeel larvae distribution and of hydrographic isolation. These studies are being further extended in the North Sea by an EC-funded project (EC DG XIV/1810/C1/94/071) (P.J. Wright, pers. comm.).

9.8 Management Measures for Wildlife Aggregations

From the above, it is evident that there is reasonably good information in the North Sea to define the areas holding seabird aggregations that are dependent to a large extent on local aggregations of sandeels. These are often in the vicinity of seabird colonies, but there is also an important large area off the east coast of Scotland which holds large numbers of seabirds particularly in the third quarter of the year. This area corresponds with that found by Jensen *et al.* (1994) where there was significant positive correlation between sandeel abundance and auk distribution. Information on seabird aggregations dependant on sandeels outside the North Sea is less well developed, but it seems likely that the principles deriving from the North Sea will apply in these other areas. Information on aggregations of other, non-fish, sandeel predators is limited. Where such information is available, it is possible to have some knowledge as to where interactions between sandeel stocks and wildlife occurs, and therefore areas where any fishery management measures should take particular note of these interactions.

A precautionary approach on the basis of the above information alone would indicate that fisheries management measures would be required. These measures would be above and beyond the precautionary management that should be in place for the sandeel fishery as a whole (Gislason and Kirkegaard, 1997). Such measures would need to ensure that fisheries are regulated in such a way which does not deplete the food resource of the wildlife in the area. It should be noted that restrictions on the catch of both 0-, 1-group and older sandeels may be required in these areas in order to allow for the differing requirements of various parts of the wildlife resource (Wright and Tasker, 1996). Surface feeding seabirds such as terns and kittiwakes tend to take 0-group fish, while diving seabirds (such as guillemots and shags) and seals take 1-group and older fish.

Until further information becomes available, two types of precautionary measure might be justified in areas where there is a wildlife concentration dependant on a sandeel aggregation. These measures would correspond with the two time scales of possible effect. In order to avoid short-term effects, it might be justified to close specific areas for specific time periods to sandeel fisheries. This, for instance, has been applied to the waters near Shetland where sandeel fishing has been limited in recent years to months prior to July 1, in order to avoid an excessive take of 0-group fish. Such closure has been recommended also by the statutory nature conservation agencies in the UK for the sandeel fishery on and near the Wee Bankie (Vincent, 1996). Gislason and Kirkegaard (1997) considered this to be in line with the precautionary approach for areas near seabird colonies. If sufficient monitoring facilities and rapid response capabilities were in place, it might be possible to tune the boundaries and timing of area closures to the state of the local sandeel stock. It is also important that closures do not merely move a fishery from one sensitive area to another.

In the longer term, it is important to ensure that sandeels populations in areas of wildlife concentrations are not depleted by fisheries in other areas. Controls on overall sandeel take in a management area would undoubtedly help achieve this objective, as in the precautionary TAC applied to the Shetland sandeel fishery at present. A considerable amount of research is required to elucidate features such as the degree of population sub-structuring of sandeels in the broader area as described above. In addition to the above information on the relative isolation of stocks, information would be needed on the size of the local aggregation and on sandeel recruitment growth and mortality at each location, as well as on the degree of annual fluctuation in these parameters. A considerable amount of research on the link between fisheries and these stocks will also be required (ICES, 1997). In addition to such research, there would be a need for a political decision on the degree of risk that wildlife concentrations should be exposed to.

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Figure 9.2.1. Areas holding internationally important concentrations of seabirds particularly dependent on sandeels in the North Sea (from Skov *et al.*, 1995).

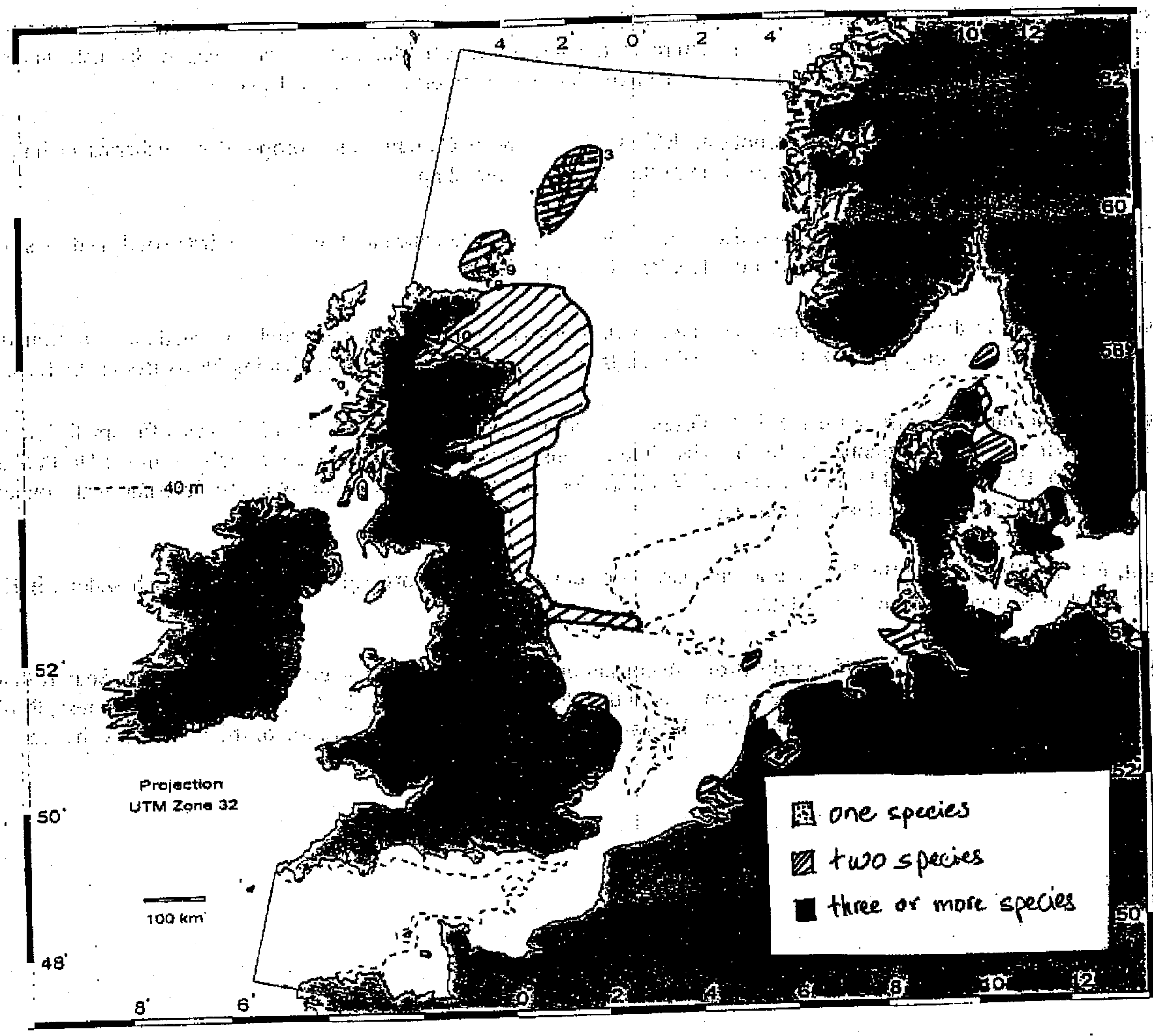
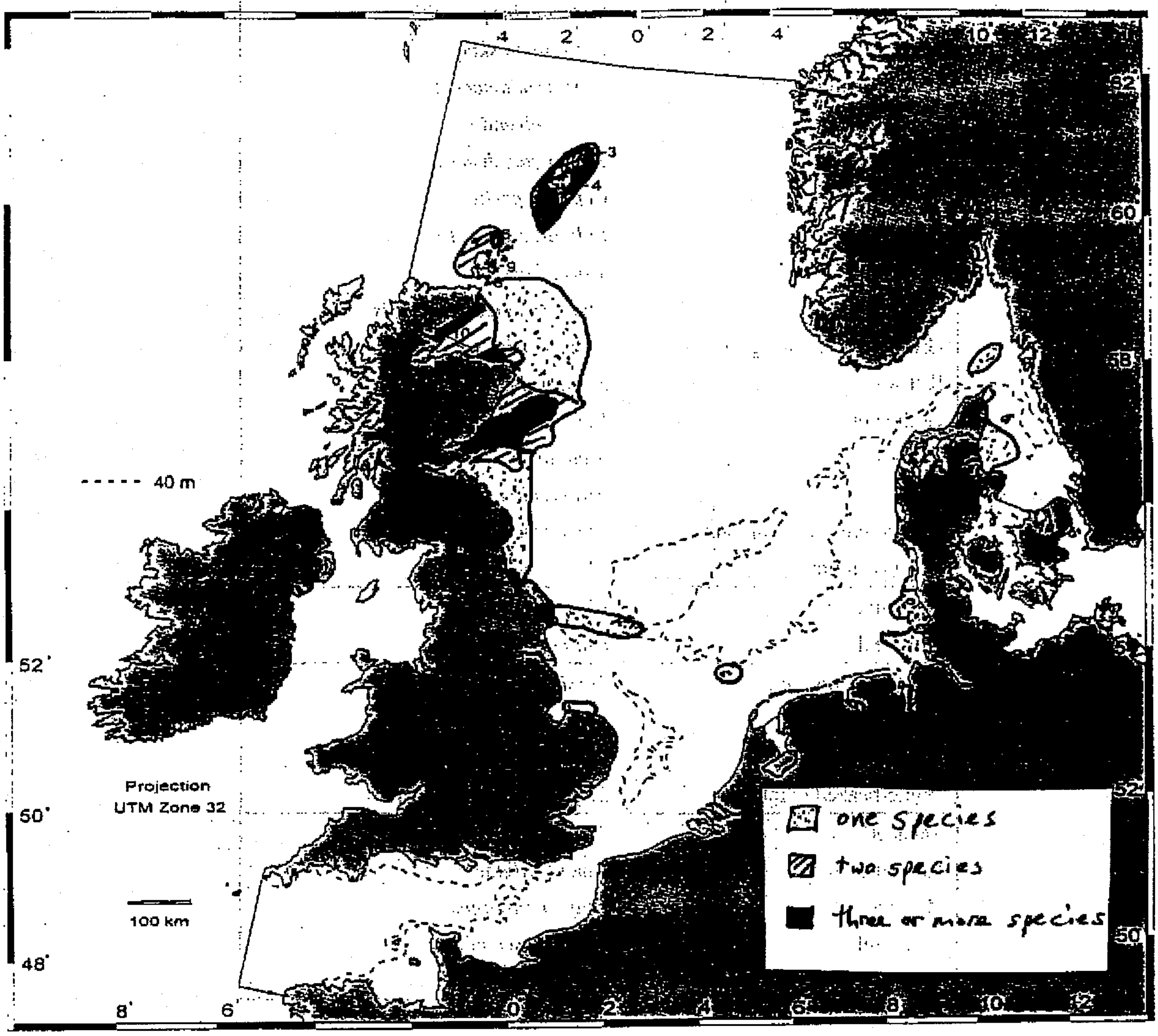


Figure 9.2.2. Number of seabird species at internationally important levels within the areas described by Figure 9.2.1 (from Skov *et al.*, 1995).



Appendix Table 9.1. Scientific names used in Section 9.

Red-throated diver	*Gavia stellata*
Arctic skua	*Stercorarius parasiticus*
Great skua	*Catharacta skua*
Sandwich tern	*Sterna sandvicensis*
Common tern	*Sterna hirundo*
Arctic tern	*Sterna paradisaea*
Black guillemot	*Cepphus grylle*
Fulmar	*Fulmarus glacialis*
Gannet	*Morus bassanus*
Shag	*Phalacrocorax aristotelis*
Great black-backed gull	*Larus marinus*
Kittiwake	*Rissa tridactyla*
Guillemot	*Uria aalge*
Razorbill	*Alca torda*
Puffin	*Fratercula arctica*
Double-crested cormorant	*Phalacrocorax auritus*
Great cormorant	*Phalacrocorax carbo*
Harbour seal	*Phoca vitulina*
Grey seal	*Halichoreus grypus*
Bottlenose dolphin	*Tursiops truncatus*
Minke whale	*Balaenoptera acutorostrata*
Harbour porpoise	*Phocoena phocoena*
White-beaked dolphin	*Lagenorhychus albirostris*
Sandeel	*Ammodytidae*
Rockling	small *Gadidae*
Herring	*Clupea harengus*
Mackerel	*Scomber scombrus*
Sprat	*Sprattus sprattus*

10 FOOD FOR THOUGHT - THE WAY FORWARD

10.1 More on Linking the Theoretical Frameworks for Studying Fishing Effects and Ecosystem Structure, Function, and Dynamics

Throughout the meeting, discussions brought in aspects of the theory of community structure and community dynamics. Even though there is great diversity of opinion in this complex field (See Sections 3 and 8), fisheries theory clearly needs to forge clear links to this other body of theory. Fortunately, there are enough points of potentially direct contact between the theoretical foundations of fisheries, and diverse theoretical views of community structure (and particularly community change, which is the issue with ecosystem effects of fishing), that it should be possible to build bridges with sound, recognizable foundations in each field.

As WGCOMP (ICES, 1997) points out clearly in its Section 4, the yield per recruit and stock-recruit functions are fundamental to fisheries theory and practice. These functions have parameters which are directly interpretable in terms of concepts from community ecology and general population dynamics. For example; the slope of the stock recruit function is interpretable as the *r* parameter of classic population models (e.g., logistic), whereas the asympotic parameter of a Beverton Holt model (or peak of a Ricker model) can be linked directly to *K* (Cushing and Shepherd, 1984). Recent work to link growth models to size spectra (Gislason and Lassen, 1997) provide another illustration of taking a model with traditional fisheries parameters and forming quantitative links to community level attributes.

10.2 How to Focus on Theoretical Frameworks with Greatest Promise

We can distinguish two types of ecosystem effects of fishing, direct and indirect. Amongst the direct effects are the changes in the target stocks, effects on non-target fish populations, direct mortality of benthos and other non-target organisms (wildlife impacts). Indirect effects include changes in levels of predation of fish, benthos and plankton resulting from changes in the size, and size structure, of the fish assemblage, changes in benthic productivity arising from changes in species composition, changes in the carbon mass balance of the system due to transfer of marine production to terrestrial systems (man) and avian scavengers, and alterations in the fluxes of nutrients due to changes in the food web and direct effects on the benthic–water column flux rate.

At present there exist a multiplicity of models of ecosystem function which have been applied to various marine communities. The majority of these are likely to apply in some circumstances, but most can be seen as being on a continuum between stochastic dynamics, biotically structured with strong stabilizing feedbacks and systems with strong non-linear dynamics (chaotic) (DeAngelis and Waterhouse, 1984). To date, no one has assessed the most widely accepted models to explicitly derive the predictions they make with regard to ecosystem effects of fishing. A critical comparison of such predictions should provide guidance on which parameters could form useful measures of ecosystem effects of fishing.

We are now in a position where considerable information has been assembled on the direct effects of fishing on target species, non-target fish and wildlife and on benthos (IMPACT II; Lindeboom and de Groot, 1997). These can be fed into a review of models of ecosystem function in order to test the predictions derived from the models.

Some progress has been made at this meeting at looking at the more tractable indirect effects—predation rate and ecosystem mass balance models. Further progress requires guidance from model constructs as to the likely indirect effects of fishing.

The time would therefore seem ripe to review the direct effects studies in a holistic manner and against a background of ecosystem function models. This would then allow us to specifically address our levels of understanding and gaps in knowledge of the indirect effects. The production of such a review would be extremely timely in that it could significantly contribute to the discussion of ecosystems effects of fishing in the *QSR 2000* and in setting priorities for the next round of European funding in this area.

10.3 Possible Future ToRs

To provide:

a) a review of the principal models of ecosystem dynamics and to develop specific predictions based on each of these for the ecosystem effects of fishing;

b) a synthesis of the findings of recent studies on the direct effects of fishing on marine ecosystems and to critically assess the possible indirect influences of fishing on marine ecosystem function with a consideration of current levels of understanding of them;

c) based on a) and b), suggestions of appropriate areas for the development of measures of the indirect effects of fishing on marine ecosystems.

10.4 Questions that Need Directed Thought

10.4.1 How should we determine minimum stock size?

As explained more fully in Section 8, to maintain reasonably 'natural' ecosystem structures, stocks of different animals must not be reduced too much. This applies not only to target species, but to by-catch species, benthos and wildlife as well. 'Minimum biologically acceptable levels' (MBAL) are hence in principle needed for all species influenced by the fishery. But actually, what does it mean that a population must not be reduced 'too much'? Which reference points should we have? How should MBALs be developed to include precautionary ecosystem aspects?

Although there are a few exceptions, generally the MBALs used by ICES today are determined from recruitment properties of single target species without considering broader ecological concerns. Being precautionary means taking the ecosystem concerns into account early in the process of developing limits for abundances of all species; target and non-target alike.

10.4.2 What can we know about the state of ecosystems before they were fished?

To do a complete job of evaluating the ecological effects of fisheries, we must have knowledge about what the ecosystem structures looked like before human fisheries had much influence. The knowledge needed should include, for example, data on relative and absolute abundances, species distributions, growth rates, size and age compositions. We also need knowledge about natural variation in these parameters, to evaluate if the variations observed today could be caused by natural factors alone or most likely by anthropogenic factors, including: How should we approach the task of hindcasting ecosystem structures and dynamics? How should we best use the few data from much earlier times which are known to exist?

10.5 Models We Want to See More Of

Modelling will continue to play a very important role in evaluating the possible ecosystem effects of fishing. At this meeting several models were presented briefly, but are presently under development, or otherwise unavailable for careful review at this time. WGECO feels that they may be useful in our tasks, and hope to have them available at future meetings, for evaluation and possible use. Note that this is not to suggest a lack of faith in the approaches considered in Section 3, but one may learn quite different things from using different tools. Also, this list is not exhaustive; it is what people at the meeting had references for.

10.6 Taxonomic Diversity

Warwick and Clarke (1995) and Clarke and Warwick (in press) have defined three biodiversity indices quantifying the taxonomic diversity and taxonomic distinctness of a faunal assemblage. These indices incorporate information on taxonomic relationships within a sample into an index measuring the dominance of species abundances. Variations of this family of indices also a) remove the effects of species abundance patterns to show pure taxonomic relatedness of individuals, and b) consider only the special case where abundance information is not available or is ignored (presence/absence data). Clarke and Warwick (in press) show that, as mean values, these statistics are largely independent of sampling effort, which makes their use attractive in large-scale spatial studies where total sampling effort in different areas cannot be standardised.

Warwick and Clarke (in press) have applied these measures to literature data on marine benthic nematodes from various intertidal/subtidal and coastal/estuarine sites in the UK, and also to coastal habitats in Chile. They demonstrate a decrease in taxonomic distinctness at sites known to be subject to pollution impacts, such as the Clyde estuary and Liverpool Bay, in comparison with 'expected' levels of taxonomic distinctness at putatively 'clean' sites in the Exe estuary, the Scillies and the Northumberland coast.

The range of published studies in which these indices have been applied appears, as yet, to be limited to the soft-sediment macro-and meiobenthic analyses cited above. An inital examination of taxonomic distinctness over a large number of samples from beam trawl surveys over a wide spatial scale (Rogers *et al.*, 1998), has been attempted using a comprehensive phylogeny of the demersal fish found in North-east Atlantic coastal waters. Initial results suggest that the technique may be more sensitive to habitat structuring forces (i.e., deep water/estaurine faunas) rather than identifying assemblages. More work needs to be done, particularly in identifying the environmental features at individual fishing station positions which may contribute to the observed patterns.

Hall and Greenstreet (in review) have also applied these measures to a long-term groundfish survey dataset collected from the northwestern North Sea. They found that the two taxonomic diversity/distinctness indices tracked the more traditional indices of species diversity very closely. However, these data also suggested that at a given level of species diversity, taxonomic distinctness was higher during the period 1929 to 1953 than between 1980 and 1993.

10.6.1 References

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Clarke, K.R., and Warwick, R.M. 1998. A taxonomic distinctness index and its statistical properties. Journal of Applied Ecology. (in press).

Hall, S.J., and Greenstreet, S.P.R. (in review) Taxonomic distinctness and diversity measures: responses in marine fish communities. Marine Ecolgy Progress Series.

10.7 Mass Balance Models

Several recent expansions of the Ecopath II approach underlying the analyses in this report make it possible to (i) address the uncertainty around impact variables for balancing the model and deriving system-level metrices (Christensen and Pauly, 1995), and (ii) to simulate changes in fishing pattern and intensity through time in an ecosystem framework (Walters *et al.*, 1997). It was not possible at this meeting to address the potential of these expansions. However, the simulation of the effect of different fishing regimes on the flow structure of an ecosystem is likely to be an interesting tool for the exploration of management options.

10.7.1 References

Christensen, V., and Pauly, D. 1995. (Announcing Ecopath 3) NAGA.

Walters, C., Christensen, V., and Pauly, D. 1997. Reviews of Fish Biology and Fisheries.

ANNEX 1

AGENDA

a) Opening of the meeting.

b) Adoption of the agenda.

c) Terms of reference (see Annex 2).

   i. Plenary discussion.

   ii. Identification of subgroup topics.

d) Communities and assemblages. [tor 'a']

e) Predation on benthos. [tor 'b']

f) Size/age composition and spatial distribution of target fish populations in OSPAR regions. [tor 'c']

g) Discards by gear type in OSPAR regions. [tor 'd']

h) Non-target fish populations. [tor 'e']

i) Reference points. [tor 'f']

j) Sandeels. [tor 'g']

k) Size and spectra. [tor 'h']

l) Any other business ('food for thought').

m) Proposals for a further meeting.

n) Consideration and approval of the report.

ANNEX 2

TERMS OF REFERENCE

ICES C.Res. 1996/2:15:6

The **Working Group on Ecosystem Effects of Fishing Activities** [WGECO] (Chairman: Dr J. Rice, Canada) will meet at ICES Headquarters from 11–19 November 1997 (inclusive; Sunday being a non-working day) to:

a) continue to develop the underlying theory on the behaviour of community metrics in relation to changes in fishing activities by:

   i. integrating information on fish assemblages sampled by different North Sea surveys,

   ii. carrying out comparative analyses on fauna assemblages from different ecosystems,

   iii. investigating spatial differences in relation to long-term trends in fishing impact by area and gear;

b) collaborate with MAWG to estimate changes in levels of predation on benthos by fish in relation to changes in exploited North Sea fish species;

c) collate and provide information on the impact of fishing activities on the size distribution/age composition and spatial distribution of the target fish populations of commercially exploited stocks of fish and shellfish (five specific species to be named) in the five OSPAR regions [OSPAR 1997/5.1];

d) collate information on quantities of discards by gear type and OSPAR regions for commercially exploited stocks of fish and shellfish provided by AFWG, WGNPBW, HAWG, NWWG, WGNSSK, WGMHSA, WGNSDS, WGSSDS, WGNAS, WGPAND, WGNEPH, SGDEEP, and SGASSO [OSPAR 1997/5.3];

e) collate and provide information on changes in abundance of individual species of non-target fish owing to fishing activities in the OSPAR regions [OSPAR 1997/5.4];

f) develop and examine potential reference points which might be used for including ecosystem considerations in relation to the precautionary approach;

g) identify and define any requirements to protect local aggregations of sandeels in sensitive areas close to important wildlife assemblages such as seabird colonies;

h) continue development of 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.

ANNEX 3

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ANNEX 4

LIST OF WORKING DOCUMENTS

WGECO 1997/1	Agenda
WGECO 1997/2	Ehrich, S., and Stransky, C. 1997. Monitoring of changes in small-scale fish assemblages in the North Sea.
WGECO 1997/3	Hansson, S., Arrhenius, F., and Nellbring, S. In press. Food web interactions in a Baltic Sea coastal area.
WGECO 1997/4	Hansson, S. 1997. Temporal variation in the Baltic Sea (draft manuscript).
WGECO 1997/5	Rogers *et al.* In press. Spatial distribution and diversity of demersal fish in UK western waters and the North Sea.
WGECO 1997/6	Fifas, and Boucher, J. 1997. Analysis of fishing activity effects on growth: the scallop population in the St Brieuc Bay (France) example.
WGECO 1997/7	Rochet, M.-J., Poulard, J.-C., and Boucher, J. 1997. Analysis of the spatial and temporal variability of the size spectrum of the fish community in the Bay of Biscay 1987–1995. 14 pp.
WGECO 1997/8	Poulard, J.-C., and Boucher, J. 1997. Spatial distribution species assemblages in the Celtic Sea and the Bay of Biscay. 19 pp.
WGECO 1997/9	Gaertner, ?. 1997. Spatial structure and temporal stability of the demersal fish communities in the Gulf of Lion (draft manuscript).
WGECO 1997/10	Gomes, M.C., and Serrão, E. 1997. Spatial patterns of groundfish assemblages on the continental shelf of Portugal. (Submitted to the ICES Journal of Marine Science.)
WGECO 1997/11	Greenstreet, S.P.R., Reeves, S.A., and Hislop, J.R.G. 199?. Indirect effects of fishing: long-term changes in the consumption of different prey by four gadoid predators in the North Sea. ICES Journal of Marine Science.
WGECO 1997/12	Gislason, H., and Lassen, H. 1997. On the linear relationship between fishing effort and the slope of the size spectrum. ICES CM 1997/DD:05. 10 pp.
WGECO 1997/13	Borges, M.F., Groom, S., Pestana, G., and Santos, A.M.P. 1997. Is the decreasing recruitment of pelagic fish (sardine and horse mackerel) on the Portuguese Continental Shelf (ICES Division IXa) induced by a change of the environmental conditions? ICES CM 1997/T:25. 11 pp.
WGECO 1997/14	Poulard, J.-C. 1997. Distribution of hake (*Merluccius merluccius*, Linnaeus, 1758) in the Bay of Biscay and the Celtic Sea from the analysis of French commercial data.
WGECO 1997/15	Connolly, P.L., and Kelly, C.J. 1996. Catch and discards from experimental trawl and longline fishing in the deep water of the Rockall Trough. Journal of Fish Biology (Supplement A), 49: 132–144.
WGECO 1997/16	Piet, G.J., Rijnsdorp, A.R., Bergman, M.J.N., and van Santbrink, J.W. 1997. Estimated fishing mortality due to beam trawling in the Dutch sector of the North Sea. 12 pp.

WGECO 1997/17	Cardador, F., Sanchez, F., Pereiro, F.J., Borges, M.F., Caramelo, A.M., Azevedo, M., Silva, A., Pérez, N., Martins, M.M., Olaso, I., Pestana, G., Trujillo, V., and Fernandez, A. 1997. Groundfish surveys in the Atlantic Iberian waters (ICES Divisions VIIIc and IXa): history and perspectives. ICES CM 1997/Y:08. 36 pp.
WGECO 1997/18	Ojaveer, H. 1997. Application of experimental trawl data for estimation of fish stock dynamics in the Gulf of Riga (Baltic Sea). 22 pp.
WGECO 1997/19	Rogers, S.I., Maxwell, D., Rijnsdorp, A.D., Damm, U., and Vanhee, W. 1997. Comparing diversity of coastal demersal fish faunas in the Northeast Atlantic. 39 pp.

Appendix Table. Index of species names.

English Name	Scientific Name
Alfonsine	*Beryx decadacylus*
Anchovy	*Engraulis encrasicolus*
Anglerfish	*Lophius piscatorius*
Balan wrasse	*Labrus bergylta*
Bass	*Dicentrarchus labrax*
Bib	*Trisopterus luscus*
Black scabbard fish	*Aphanopuus carbo*
Black sea-bream	*Spondyliosoma cantharus*
Blonde ray	*Raja brachyura*
Blue whiting	*Micromesistius poutassou*
Bluemouth	*Helicolenus dactylopterus*
Boar-fish	*Capros aper*
Brill	*Scophthalmus rhombus*
Bullrout	*Myoxocephalus scorpius*
Butterfish	*Pholis gunellus*
Capelin	*Mallatus villosus*
Common catfish	*Anarchichas lupus*
Cod	*Gadus morhua*
Common dragonet	*Callionymus lyra*
Common skate	*Raja batis*
Conger eel	*Conger conger*
Cuckoo ray	*Raja naevus*
Common dab	*Limanda limanda*
Dragonet	*Callionymus* sp.
Eelpout	*Zoarces viviparus*
Five-bearded rockling	*Ciliata mustela*
Four-bearded rockling	*Enchelyopus cimbrius*
Golden-eye perch	*Beryx splendens*
Greater argentine	*Argentina silus*
Greater forkbeard	*Phycis blennoides*
Greater weever	*Trachinus draco*
Greenland halibut	*Reinhardtius hippoglossoides*
Roundnose grenadier	*Coryphaenoides rupestris*
Rough grenadier	*Macrourus berglax*
Grey gurnard	*Eutrigla gurnardus*
Gulper shark	*Centropgorus granulosus*
Haddock	*Melanogrammus aeglefinus*
Hake	*Merluccius merluccius*
Herring	*Clupea harengus*
Halibut	*Hippoglossus hippoglossus*
Hooknose	*Agonus cataphractus*
Horse mackerel	*Trachurus trachurus*
John Dory	*Zeus faber*
Kitefin shark	*Dalatias licha*
Lemon sole	*Microstomus kitt*
Lesser argentine	*Argentina sphyraena*
Lesser weever	*Echiichthys vipera*
Lesser-spotted dogfish	*Scyliorhinus canicula*
Ling	*Molva molva*
Long rough dab	*Hippoglossoides platessoides*
Lumpsucker	*Cyclopterus lumpus*
Mackerel	*Scomber scombrus*
Megrim	*Lepidorhombus whiffiagonis*

Appendix Table. Index of species names.

English Name	Scientific Name
Norway haddock	*Sebastes viviparus*
Norway pout	*Trisopterus esmarki*
Norwegian herring	*Clupea harengus*
Norwegian topknot	*Phrynorhombus norvegicus*
Orange roughy	*Hoplostethus atlanticus*
Pilchard	*Sardina pilchardus*
Plaice	*Pleuronectes platessa*
Polar cod	*Boreogadus saiidi*
Pollack	*Pollachius pollachius*
Poor cod	*Trisopterus minutus*
Red gurnard	*Aspitrigla cuculus*
Red mullet	*Mullus surmuletus*
Red sea-bream	*Pagellus bogaraveo*
Ocean perch	*Sebastes marinus*
Redfish	*Sebastes mentella*
Saithe	*Pollachius virens*
Sandeel	*Ammodytes* sp.
Scaldfish	*Arnoglossus laterna*
Sea snail	*Liparis liparis*
Smooth hound	*Mustelus mustelus*
Sole	*Solea solea*
Solenette	*Buglossidium luteum*
Spotted catfish	*Anarhichas minor*
Spotted dragonet	*Callionymus maculatus*
Spotted ray	*Raja montagui*
Sprat	*Sprattus sprattus*
Spurdog	*Squalus acanthias*
Starry ray	*Raja radiata*
Thornback ray	*Raja clavata*
Three-bearded rockling	*Gaidropsarus vulgaris*
Tope	*Galeorhinus galeus*
Tub gurnard	*Trigla lucerna*
Turbot	*Scophthalmus maximus*
Tusk	*Brosme brosme*
Whiting	*Merlangius merlangus*
Witch	*Glyptocephalus cynoglossus*
Wreckfish	*Polyprion americanus*