ICES WGFE Report 2005

Report of the Working Group on Fish Ecology (WGFE)

21-26 February 2005

Santander, Spain

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

Contents

Executive summary

This report details the work undertaken during the third meeting of the ICES Working Group on Fish Ecology (WGFE). The work areas undertaken addressed the following issues: Ecological Quality Objectives (EcoQOs) for fish communities (Section 2), fish habitat (Section 3), summarising important fish and fisheries data sets for the North sea and of potential interest to the REGNS project (Section 4), estimates of maximum gear-specific effort levels that North sea elasmobranchs that can be exerted without exceeding the sustainable mortality rates (section 5), the relative catchability of fishes in different survey gears (Section 6), abundance-range size relationships (Section 7), examination of the decline criteria used by conservation agencies (Section 8), further studies on the prey composition, food rations and gastric evacuation of North Sea fishes (Section 9), reviewing the feeding ecology of fish in the North sea and adjacent waters (Section 10), initiating broadscale studies of southern and western surveys (Section 11) and reviewing the updated IBTS manual with reference to the quality assurance for fish identification (Section 12).

Ecological Quality Objectives (EcoQOs) for fish communities are required by OSPAR, and analyses on various size-based metrics have been undertaken by WGFE. In this report (Section 2) we have summarised the analyses undertaken in previous years, undertaken similar analyses for the Cantabrian Sea, explored the utility of a "predation size spectrum", and used simulation environments to examine the sensitivity of selected indicators. Future studies will provide a protocol for such studies, explore the categorisation of "large" fish and undertake further evaluation of potential EcoQOs using simulation tools.

Various aspects of fish habitat were described (Section 3), and included a brief summary of the habitat requirements for some deep-water fish species, a description of the Le Danois Bank off the Cantabrian continental shelf, the relationship between physical environmental parameters and fishes in the Barents Sea, and mapping fish diversity in the Canadian Atlantic, with special reference to wolf-fishes.

The main ICES data sets that provide useful information on the distribution, relative abundance, biology and feeding habits of North sea fishes were summarised for the REGNS project (Section 4), with a brief overview of the kinds of effort and discards data that are available also provided.

Preliminary studies to explore the gear-specific catchability of elasmobranchs in the North Sean were undertaken (Section 5 and Section 6), with provisional estimates of the maximum gear-specific effort levels that can be exerted without exceeding the sustainable mortality rates for selected species provided (Section 5). This work area will be continued and expanded by the Working Group on Elasmobranch Fishes later this year. Additional studies on the relative catchability of fishes (Section 6) were undertaken for different vessels/gears in the Gulf of St Lawrence (Western IIA otter trawl *vs.* a fine mesh shrimp trawl URI 81/114), Barents Sea (Norwegian Campelen 1800 trawl *vs.* Russian research trawls type 2283–02) and Galician waters (Baca trawl *vs.* beam trawl).

Further analyses of inter-specific and intraspecific abundance-occupancy relationships were undertaken (Section 7), and it was considered that this area of research had potential utility to the conservation and management of fishes, and that further studies should be continued. Case studies examined interspecific relationships in the Barents Sea and North Sea, and intra-specific relationships for thorny skate in the Canadian Atlantic and Norway pout in the North Sea.

The decline criteria used by various conservation fora were examined (Section 8), with results broadly consistent with the outcomes of more rigorous assessment methods (with respect to a

matrix of hits, misses and false alarms). In terms of other nature conservation issues for marine fishes, a brief summary of the Canadian Species at Risk Act (SARA) was also provided.

Revised estimates of the prey composition, gastric evacuation and food rations of selected North Sea fishes are in progress (Section 9), though it was recommended that this TOR be addressed in greater detail during 2006. A summary of studies on the feeding ecology of North Sea fishes was also collated (Section 10).

Studies of the fish communities along the continental shelf of the North-eastern Atlantic were summarised, with emphasis on the Cantabrian and Celtic Seas (Section 11). Future studies will further integrate data sets from these regions and WGFE will liaise with IBTS accordingly. The revised IBTS manual was also discussed and methods for ensuring high quality for the identification of fishes proposed (Section 12).

1   Introduction

1.1   Terms of Reference

The Working Group on Fish Ecology [WGFE] (Chair: J. Ellis, UK) will meet in Santander, Spain, from 21–26 February 2005 to:

High priority:

     a.   continue the development of EcoQOs for the fish communities by:

     i.   conducting further studies on size-based indicators,

     ii.   developing a suite of indicators that address specific issues of the fish communities on the relevant spatial scales;

     b.   continue the descriptions of essential fish habitat, to support studies on threatened, commercial, and selected non-target species;

     c.   with reference to the request from REGNS for data on fish individual abundance at length, weight at length, age at length and maturity at length, for all species (both commercial and non-commercial), discards data for all gear types and all fleets, effort data for all gear types and all fleets based on logbook data at the scale of ICES rectangle across the North Sea for the period 1984–2004. Further to this, extract and compile all available stomach content data for diet analysis.

     i.   determine what parts of the data request can be met within the time frame specified by REGNS;

     ii.   identify data gaps and impediments to data access;

     iii.   interact with the Bureau Working Group on Data and IT issues with a view of contributing to developing a longer term strategy to address issues on data gaps and impediments to data access;

     d.   working with the Working Group on Elasmobranch Fishes and the Working Group on the Assessment of Demersal Stocks in the North Sea, explore the feasibility of estimating gear-specific catchability (for example with the methods reported by WGFE 2004) to various species of skates, rays, and sharks in the North Sea. Use the results to provide estimates of maximum gear-specific effort levels that can be exerted without exceeding the sustainable mortality rates for those species or species groups. Coordination of the work and presentation of the results should be led by WGEF.

     e.   continue studies on the relative catchability of fishes, including the effects of fish size, in survey gears, and examine the implications of gear effect on:

     i.   descriptions of the structure and function of fish assemblages, and

     ii.   associated fish community metrics;

Lower priority:

     f.   undertake analyses to examine abundance-range size relationships in marine fishes, including:

     i.   the effect of sampling issues on the form of the abundance-occupancy relationship,

     ii.   intra- and interspecific abundance-occupancy relationships of marine fishes,

     iii.   the processes and mechanisms underlying marine abundance-range size patterns, and

     iv.   explore the utility of abundance-occupancy relationships in fisheries and ecosystem management issues;

     g.   evaluate the decline criteria used by existing nature conservation organisations, and address any upcoming nature conservation issues for marine fishes;

h. continue studies on food rations and prey composition of North Sea fishes by:

i. re-evaluating predation mortalities of the MSVPA prey fish populations, and examine the consequences by relevant runs of MSVPA/FOR when using food rations of MSVPA predators obtained by application of a new mechanistic gastric evacuation model rather than food rations used at present by the ICES, and

ii. estimate food rations and prey compositions of grey gurnard, horse mackerel, and mackerel in the North Sea, applying new information about gastric evacuation rates;

i. review existing literature and available data sources for the diet, feeding habits, and foraging behaviour of target and non-target fishes in the North Sea and adjacent waters;

j. initiate studies on the broadscale temporal changes in fish assemblages along the European continental shelf of the eastern North Atlantic (ICES Divisions VI–IX);

k. review the updated IBTS Manual and assess whether the protocols and quality assurance procedures of IBTS data are sufficiently robust to provide the data appropriate to examine the various issues relating to fish communities.

WGFE will report by 30 April 2005 for the attention of the Living Resources, the Resource Management, and the Diadromous Fish Committees, as well as ACE.

1.2 Participants

The following scientists attended the Working Group meeting. Full contact details are given in Annex I.

Tom Blasdale (JNCC, UK)
Andrey Dolgov (PINRO, Russia)
Nick Dulvy (CEFAS, UK)
Daniel Duplisea (DFO, Canada)
Jim Ellis (CEFAS, UK)
Helen Fraser (FRS, UK)
Geir Huse (IMR, Norway)
Dave Kulka (DFO, Canada)
Lena Inger Larson (ICES, Denmark)
Ignacio Olaso (IEO, Spain)
Izaskun Preciado (IEO, Spain)
Are Salthaug (IMR, Norway)
Francisco Sanchez (IEO, Spain)
Alberto Serrano (IEO, Spain)
Yunne Shin (IRD, France)
Francisco Velasco (IEO, Spain)

The following working group members contributed by correspondence, and submitted background documents, data or text.

Niels Gerner Andersen (DIFR, Denmark)
Julia Blanchard (CEFAS, UK)
Verena Trenkel (IFREMER, France)
Siegfried Ehrich (Germany)
Ivone Figueiredo (IPIMAR, Portugal)

1.3 Background

The Working Group on Fish Ecology first met in 2003 (ICES, 2003). The rationale behind the formation of the group was to support ICES on issues of fish community metrics and to provide advice on threatened marine fishes. OSPAR and HELCOM had requested advice in these

areas from ICES, and ICES had been unable to respond. Until 2002, fish community issues were considered by WGECO, but as the demands on WGECO increased the establishment of WGFE enabled a more focussed consideration of fish community issues. WGFE met again in 2004, and continued ecological studies, including the development of Ecological Quality Objectives (EcoQOs) for fish communities, abundance-range size relationships, and the relative catchability of fishes in different survey gears (ICES, 2004). WGFE has addressed issues on non-commercial fish species, including species of conservation importance, fish communities and assemblages, and other aspects of fish ecology (e.g., feeding habits and prey rations, habitat requirements), so that ICES can provide advice in these areas in relation to ecosystem, biodiversity and nature conservation issues.

1.4　References

ICES. 2003. Report of the Working Group on Fish Ecology. ICES CM 2003/G:04; 113pp.

ICES. 2004. Report of the Working Group on Fish Ecology. ICES CM 2004/G:09; 257pp.

2   Developing EcoQOs relating to fish communities and associated indicators and reference levels

2.1   Introduction

In 2005 we addressed the terms of reference for EcoQO's by summarising the considerable information presented in case studies from the 2003 and 2004 reports. Accordingly, we developed a table summarising the indicators examined in these studies and added the results of a new case study from the Northern Spanish Shelf conducted at the present meeting. This summary proved useful both to show the general trends in indicators across the systems studied as well as some of the difficulties found in studies and our ability to compare indicators across systems. This has led to some particular questions which should be considered in future case studies and the recommendation that WGFE, in the coming years, presents a series of guidelines or a protocol for constructing, analysing and presenting EcoQO studies. A protocol for indicator case studies should make comparisons between studies more rigorous and allow a more systematic evaluation of the indicators so they can be classified according to utility under various circumstances.

A new size-based indicator, predation size spectra, is presented in this report and the evaluation of some indicators was conducted using fish community models as simulation environments. We feel that simulation environments could be useful to evaluate various metrics in a systematic manner and examine their sensitivity to fishing effort and robustness to errors and uncertainties.

2.2   Size-based indicators

2.2.1   Case study: Northern Spanish Shelf (ICES Divisions VIIIc and north IXa)

2.2.1.1   Introduction

This is an important new case study because it provides additional information from southern waters in ICES area, and also because the Portuguese shelf studies from the previous reports appeared to show patterns different from most other systems. Therefore it would help to clarify if this is a more general pattern in southern areas.

2.2.1.2   Material and methods

Data analyzed come from the time-series of ground fish surveys in the Northern Spanish Shelf (SPGFS) carried out yearly every autumn by the IEO. These surveys follow a random stratified sampling design (ICES, 2002) with three bathymetric strata and five geographic ones, with a number of hauls per strata proportional to the strata trawlable area.

To address the variation over time of the size-related EcoQOs the yearly mean stratified length distributions per 30 min haul were used. The survey series commenced in 1983, was not performed in 1997, and until 1992 the length distribution of only the commercial species was measured, thereafter all species were measured. Therefore, our size-based community indicators could not be calculated for the first nine years of the survey.

The indicators investigated were:

1. Changes in the combined length distribution in number of all fish species: Both ends were cut (15–90 cm) to avoid sizes not thoroughly sampled by the gear, and outliers with scarce large sizes. Two different data sets were analyzed, one including all species and another one excluding species with high inter-annual variability in their abundance (silvery pout, boarfish, snipefish and horse mackerel)

and also the red-band fish, which also has a high inter-annual variability and at the same due to its morphology has an important effect in the perception of the right-hand of the distribution. Length distributions are presented as natural logarithm of the number *vs.* natural logarithm of the midpoint-length in 5 cm length classes. Linear regressions are fitted to the resulting annual length distributions looking for historical changes in the slope using the ratio between each year value and the value from the first year of the time series.

2. Percentage of large fish: two approaches to define "large fish" were explored,

   a. For the overall fish community the 0.6, 0.85 and 0.95 quantiles of the length distribution were estimated using the two data sets mentioned in the previous paragraph.

   b. For some commercial species (hake, blue whiting, megrim and four-spotted megrim) different multipliers of their Linf were used as threshold to define "large fish" and the consistency of the results with the different values is assessed.

To explore possible time-trends results are presented as the ratio between each year-value and the value from the first year of the time series.

2.2.1.3   Results

Figure 2.1 presents the variation along the time series of the length composition of the whole fish assemblage sampled in the bottom trawl surveys between 1992 and 2004. No clear time trends related are evident in the size spectra of the fish caught in SPGFS surveys, although there is a slight trend in increasing (shallowing) slope between 1999 to 2004 it is not important when considered within the variability found in the whole time series studied.

The differences between the two data sets studied are small as it is evident from the evolution of slopes shown (Figure 2.2), and applying a paired Wilcox-test to both sets no significant differences are found (p=0.12). This lack of differences is partly due to excluding fish smaller than 15 cm in the analysis, since the catches of all the species excluded (except red-bandfish and horse mackerel) are comprised mainly by individuals smaller than this size. Additionally, in the case of horse mackerel the large fluctuations in abundance are due to blooms of small recruits also smaller than 15 cm, like in 1994 (Sánchez *et al.*, 2002).

All Fish Species

All Fish Species but (*)

(*) Silvery pout, boarfish, snipefish, horse mackerel & red bandfish
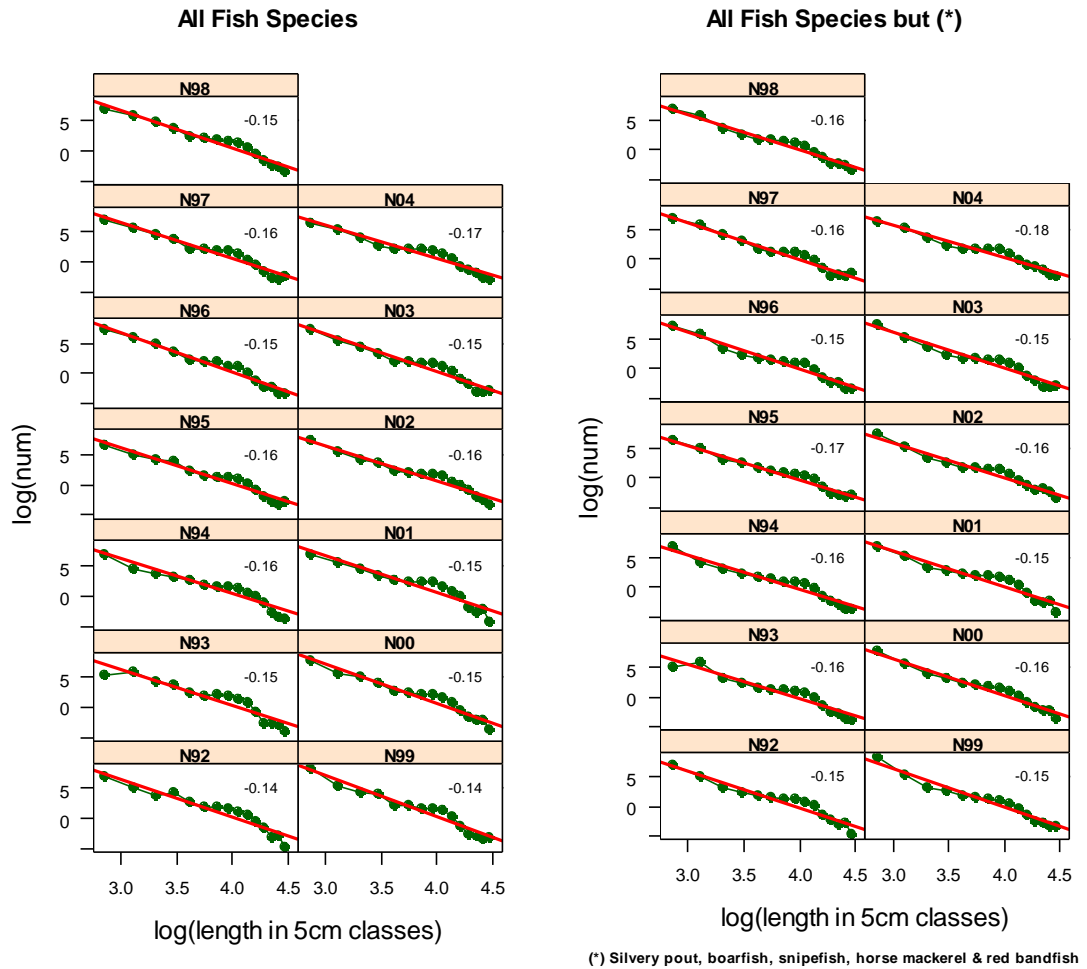
Figure 2.1: Length distributions of fish larger than 15 cm and smaller than 90 cm.

All Fish Species **All Fish Species but (*)**
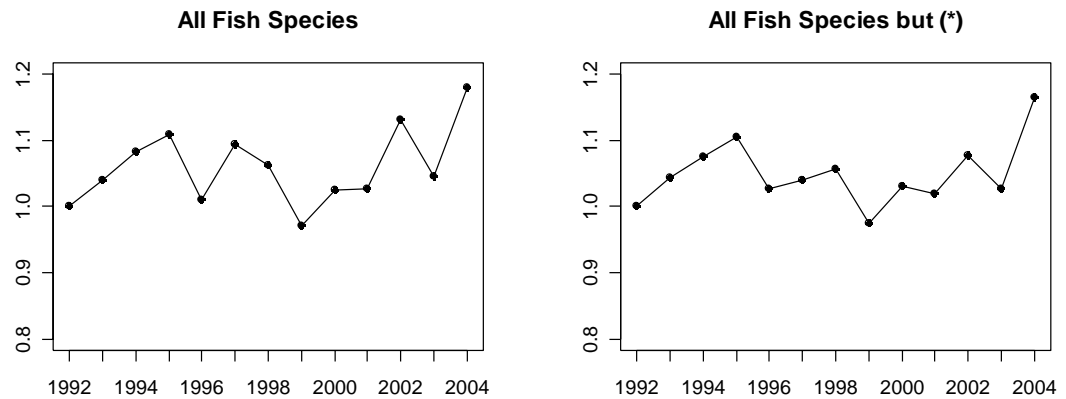
(*) Silvery pout, boarfish, snipefish, horse mackerel & redband fish

Figure 2.2: Size spectra slope evolution (slope/slope first year) along the time series available 1992–2004. (See Figure 2.1 for complete length distributions: 15–90 cm).

The differences between including and excluding some species and/or individuals smaller than 15 cm are evident comparing the results on the percentage of large fish (Figure 2.3) where four different options have been explored:

All fish species and the whole size range: there are important differences depending on which quantile of the length distribution is chosen to define "large fish", especially between using quantile 0.6 and the larger ones (> 15 cm). But in spite of these apparent differences among the three series, they are not found statistically significant applying paired Wilcox-tests in pairs to the three time series.

Selected fish species and the whole size range: by removing highly variable species the results are not so dependent on what quantile of the length distribution is chosen to define large fish. Nevertheless it has to be taken into account that the majority of the species excluded are small ones and therefore mainly fishes smaller than 15 cm are excluded. Paired Wilcox-tests show significant differences between the time series obtained with 0.6 quantile and the two larger ones, but not among the larger ones. In this case the large percentage of large fishes found in 1992 is due to the low recruitment of blue whiting in this particular year that produced an important decrease of fishes smaller than 15 cm.

All fish species fish but only those larger than 15 cm: the results with this set of data are similar to those obtained with the previous data set but for the oscillations produced by the variability of blue whiting recruitment, as it is clearly shown in the decrease of the effect of these recruits found in 1993. In this case the paired Wilcox-tests found statistically significant differences between 0.95 quantile and the two smaller ones, while they are not significant between 0.6 and 0.85 (p = 0.11).

Selected fish species and only fish larger than 15 cm: taking the most restrictive data set gives the least consistent results between the three quantiles considered since the Wilcox-tests only show not significant differences between quantiles 0.6 and 0.95 and with a probability close to the 0.05 significance level (p=0.08).
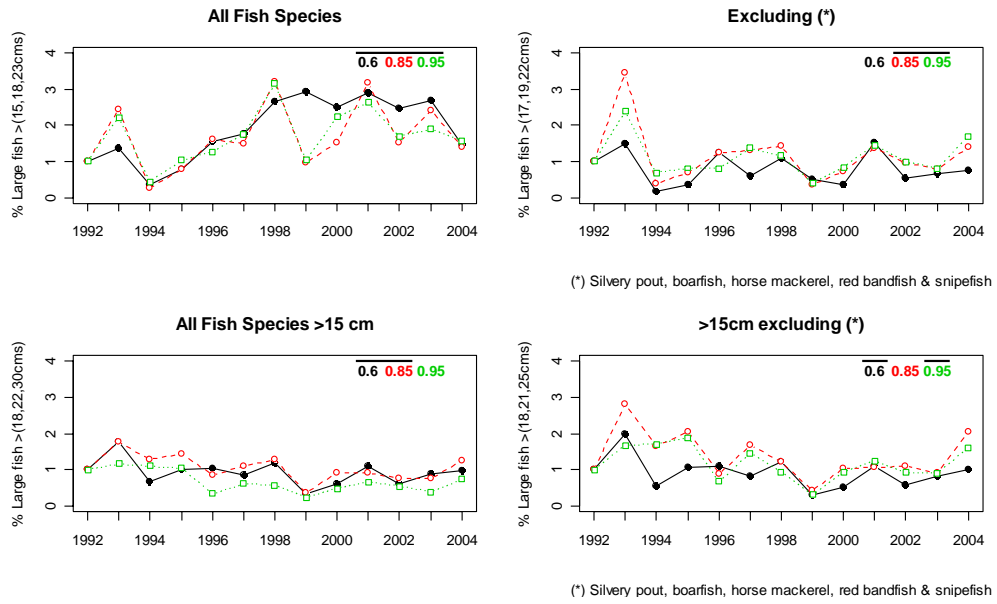
Figure 2.3: Percentage of large fishes in the community of demersal fish from the Cantabrian Sea. Using to define "large fish" the quantiles 0.6 (● black), 0.85 (○ red) and 0.95 (□ green) of the length distribution in the assemblage.

Figure 2.4 shows the proportion of large fishes for some important commercial target species in Northern Spanish Shelf using three different multipliers of the correspondent species Linf to define the "large fish". The proportion of large fishes has a strong dependence on the multiplier of Linf used and it not only depends on the multiplier but also on the species studied, for example in the case of hake the larger multiplier that did not produce any missing data was 0.45, while in the case of blue whiting it was 0.7 and in both megrims species 0.8, this is probably related with span of the length distribution of each species, e.g., it is much wider for hake than the other species considered.

Considering the different species no clear historical trends on the proportion of large fish are evident for three of the four species considered, the only exception is four-spotted megrim that shows an increasing trend in the proportion of large fish, mainly from 1988 and onwards, and when using the two smaller multipliers of Linf, trend that is less clear when the larger multiplier is used. A further exploration is needed in order to detect possible driving factors for the changes in the percentage of large fish for the species considered.
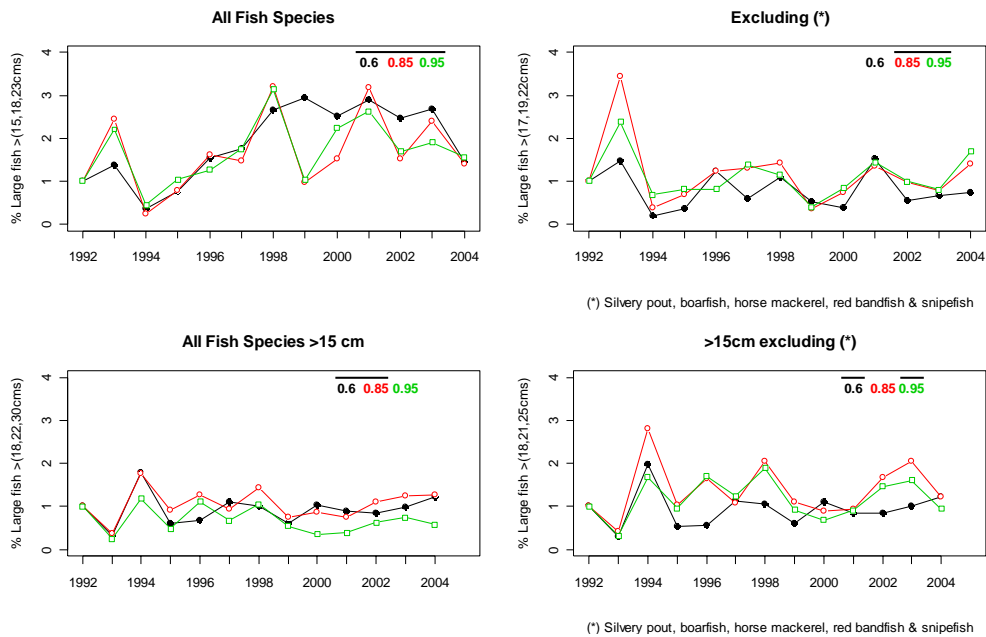
Figure 2.4: Evolution (as ratio of initial year) of the percentage of large fishes for several demersal fish species in Northern Spain shelf. Using to define "large fish" different multipliers of Lmax (● black < ○ red < □ green). The line over the multipliers indicates no significant differences between them.

2.2.2   Review of the case studies of WGFE 2003–2005

The working group considered survey series within twelve geographic areas. Much experience has been learnt from constructing time series of indicators in all these case studies, that helped us to identify gaps and establish some recommendations for the future meetings of the WGFE (see Section 2.4). Table 2.1 summarises the results obtained in terms of identified trends in five different size-based indicators. The trends were generally negative for the longer time series, while absent in the shorter series. The only exception was an increase in the percentage of large fish on the Portuguese shelf. Considerable work has been done for providing consistent time series of some indicators. Before constructing indicators, data are usually filtered in various ways to remove biases in data. Regretfully, the multiple choices made are not always documented or justified: cut-off of the size distribution or time series, exclusion of some particular species etc. If data filtering is not well documented, then it makes it difficult to compare different case studies and derive general characteristics of the indicators. It was also noticed that the emerging pattern in any metric could strongly depend on the taxonomic groups or species included in the analyses. The selection of the species for indicators analyses was based on various criteria (undersampling problems, dominant species in the ecosystem, morphological differences, habitat- or community-specific sampling gear, and availability of species parameter such as Linf) that are not easy to defend if they are to represent "the fish community" in terms of ecological quality. A checklist or series of questions should be constructed to aid in the consistent development of empirical indicator studies so that data filtering decisions are explicitly considered and documented.

Globally, the data presented appear to confirm that higher effort is related to a lower average weight, a lower average maximum length, a lower proportion or abundance of large fish, and a steeper slope of the size spectrum. However, it was not easy to derive a consistent interpretation of the temporal trends in the indicators presented by the various data sets, primarily because a suitable metric of the exploitation rate of each community is lacking for most areas. One of the reasons we are developing EcoQO's is that we hope they will summarise a com-

munity state and that they will reflect the change in state in response to fishing. Unfortunately, most studies on indicators fail to include information on fishing effort. It is therefore difficult to know if temporal trends in the values of indicators are a result of fishing or some other process, either natural or human induced. It is often very difficult to have a measure of total fishing effort on a community though community F values weighted by biomass of constituent and major species can sometimes be determined in some ecosystems (see method proposed by Daan *et al*. in ICES, 2003). For the North Sea and the Celtic Sea, enough information appears to be available to estimate some overall trend for the last 30 years as well as to classify rectangles according to average recent effort. If even rough measures of fishing effort can be constructed for communities, it could be very useful for interpreting trends in EcoQO indicators. The best measure of community F would probably be the ratio of total biomass removed per year by all fisheries (including discards) to average biomass present. But these data are hardly available.

When comparing case studies, it was difficult to draw general conclusions on indicators sensitivity because indicators were not standardised. In order to make comparative analyses possible and to aid in selecting a panel of useful indicators for an EAF, indicators should be represented in terms of relative variations or coefficients of variation. This method is in accordance with the fact that directions of change (with associated directions of reference) rather than absolute values of indicators are suggested to be more appropriate for assessing fishing effects at the community level (Link *et al*., 2002, Shin *et al*., 2005).

Table 2.1: Summary of trends in the size-based indicators obtained within the different case studies of the WGFE (ICES, 2003–2004 and the present one). /: not reported or not calculated, *: Northern Spanish shelf, case study in the present report.

Area	time span	source of data	spatial analysis	Nb of species (species excluded)	size cut off	fishing mortality	TRENDS IN SIZE-BASED INDICATORS				
							Mean weight	Max length	Proportion of large fish (PLF)	Number of large fish	Slope of size spectrum
North-western North sea	1925–1996	Scottish August Groundfish Survey	yes	56 (herring, sprat, sandeel) (28 species excluded for max length)	/	rectangles classified into low, medium and high f levels	significant negative trend in medium and high f rectangles	significant negative trend in medium and high f rectangles	significant negative trend in high f rectangles	/	/
North-western North sea	1983–1996	Scottish August Groundfish Survey	yes	56 (herring, sprat, sandeel) (28 species excluded for max length)	/	rectangles classified into low, medium and high f levels	significant difference between zones with different levels of f	no significant difference between zones	significant difference between zones with different levels of f	/	/
North Sea	1980–2001 1985–2001 1969–2001	IBTS BTS SNS	/	/	/	multispecies F estimate increased til 1985 and stabilized	significant negative trend	significant negative trend	/	significant negative trend	/
North Sea	1977–1999	English groundfish survey	/	107	15–150 cm	/	/	/	/	/	decrease
North-Norway West-Spitsbergen	1995–2002	Norwegian bottom-trawl survey	/	(herring, capelin, invertebrates)	/	/	no significant trend	/	no significant trend	/	/
Celtic Sea	1987–2003	CEFAS Celtic Sea groundfish surveys	yes	117 (33 excluded for max length)	/	multispecies F estimate	significant negative trend	significant negative trend	slight decreasing trend	significant negative trend	significant negative trend

Table 2.1 Continued

Area	time span	source of data	spatial analysis	Nb of species (species excluded)	size cut off	fishing mortality	TRENDS IN SIZE-BASED INDICATORS				
							Mean weight	Max length	Proportion of large fish (PLF)	Number of large fish	Slope of size spectrum
Portuguese continental waters	1990–2002	IPIMAR autumn demersal surveys	/	(snipefish, boarfish)	/	landings increased till mid-1970s then steadily decrease	no trend	no trend	significant increasing trend for 95th percentile	/	/
Barents Sea	1980–2003	Norwegian bottom trawl survey	/	/	/	/	no trend	no trend	no trend	/	no trend
Northern Gulf of St Lawrence	1990–2001	DFO groundfish surveys	/	66 (no invertebrates but squid)	15–150 cm	/	/	/	/	/	no trend
Cabot Strait	1970–2001	DFO groundfish surveys	/	43 species (no invertebrates but squid)	15–150 cm	/	/	/	/	/	decreasing trend
Eastern Scotian Shelf	1970–2001	DFO groundfish surveys	/	51 species (no invertebrates but squid)	15–150 cm	/	/	/	/	/	decreasing trend
Western Scotian Shelf	1970–2001	DFO groundfish surveys	/	49 species (no invertebrates but squid)	15–150 cm	/	/	/	/	/	no trend
Georges Bank	1986–2001	DFO groundfish surveys	/	45 species (no invertebrates but squid)	15–150 cm	/	/	/	/	/	linear decreasing trend
Northern Spanish shelf*	1992–2004	SPGF survey	/	185 species (silvery pout, boarfish, horse mackerel, redband fish, snipefish)	15–90 cm	/	/	/	no trend	/	no trend

2.2.3 What is a large fish? Consequences for the sensitivity to fishing of the proportion of large fish

In the last two years, the WGFE has mainly documented three size-based indicators: the mean size of fish, the maximum length of fish and the proportion of large fish in a community. The proportion of large fish (PLF) in a population or in a community is a size-based indicator which is one of the most concrete and easily understandable by non-scientists; however, the proportion of large fish at the population or at the community level is difficult to objectively define among the size-based indicators. Some criteria must be explicitly considered to determine the size beyond which a fish is "large" (hereafter called MSLF: minimal size of large fish). Before a MSLF is adopted for further work, the consequences of choosing a MSLF must be evaluated for sensitivity of the PLF indicator. In some cases, different MSLF can lead to divergent responses of the PLF to fishing mortality which is an undesirable behaviour for an indicator upon which we assess community state.

We distinguished three different ways to choose the MSLF:

1- statistical criterion: In the previous reports, the size beyond which a fish is considered to be large was defined according to percentiles of size distributions which were aggregated over a time series. Depending on the region considered, these percentiles correspond to different MSLF. The following Table 2.2 summarizes the size limits chosen to calculate the proportion of large fish and documents if the trends obtained are similar (c.f. ICES, 2003) and Section 2.2.1 (Northern Spanish shelf case study in this report). When the trends in the proportion of large choosing the percentile.

Table 2.2: fish are divergent while using different percentiles for determining the MSLF, the possible causes must be explored and documented as results are sensitive to the subjective decision of Percentiles used for the calculation of the proportion of large fish in the different case studies of the WGFE. Corresponding minimal sizes of large fish are reported, and temporal trends obtained for the different corresponding proportions of large fish.

	MINIMAL SIZE OF LARGE FISH			
Area	60th percentile	85th percentile	95th percentile	trends in PLFs
North-western North sea	/	/	30 cm	/
North-Norway	19 cm	36 cm	52 cm	similar
West-Spitsbergen	15 cm	31 cm	59 cm	divergent
Celtic Sea	21 cm	28 cm	33 cm	similar
Portuguese continental waters	20 cm	25 cm	33 cm	divergent
Northern Spanish shelf	15 cm	18 cm	23 cm	similar

2- biological criterion: At the population level, a more biological criterion can be adopted, assuming that a large fish is a mature fish. To test the robustness of this criterion, several multipliers of length at maturity (>1) or several multipliers of maximal length (<1) can be systematically applied to determine the MSLF and calculate the proportion of large fish. By so doing, it may be informative to identify for which species the trend in the proportion of large fish is not sensitive to the MSLF size threshold chosen.

The Osmose model was applied to the southern Benguela fish community (see Section 2.3.1 for more details) and the robustness of the proportion of large fish was tested using simulations with increasing community fishing mortality. Simulations results suggest that for most species, including anchovy, deep water hake, kingklip, lanternfish, round herring, silver kob, snoek- Figure 2.5 a-d-f-g-i-k-l), the choice of the MSLF does not have much influence on the sensitivity of the PLF to fishing mortality. For some other species, the trends obtained for the PLFs based on different MSLFs are divergent (chub mackerel, horse mackerel, lightfish – Figure 2.5 b-e-h) such that opposite directions of change can be found (shallow water hake – Figure 2.5 c).
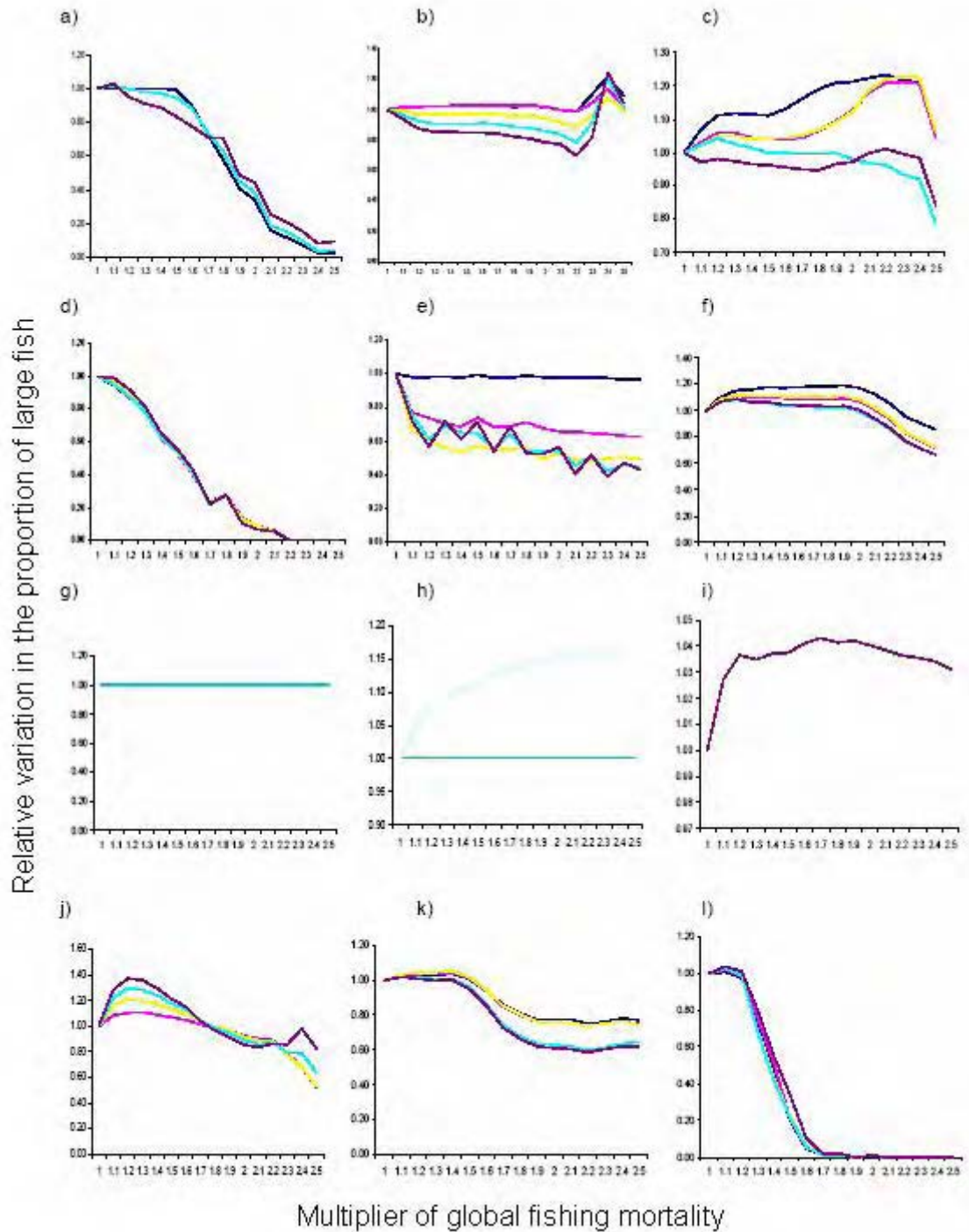
Figure 2.5: Relative variations of the proportion of large fish resulting from simulations of increasing global fishing mortality on the southern Benguela fish community, using Osmose model. The proportion of large fish is represented for each species, and for different minimal sizes of large fish (multiplier of the length at maturity ranging from 1 to 1.5). a: anchovy, b: chub mackerel, c: shallow water hake, d: deep water hake, e: horse mackerel, f: kingklip, g: lanternfish, h: lightfish, i: round herring, j: sardine, k: silver kob, l: snoek.

3- fishing criterion: In previous WGFE reports, many size spectra were not characterised necessarily by a slope and intercept, but by the number of fish (abundance index) in each size class. It was the representation adopted for the North Sea (ICES, 2003, § 2.3.1).
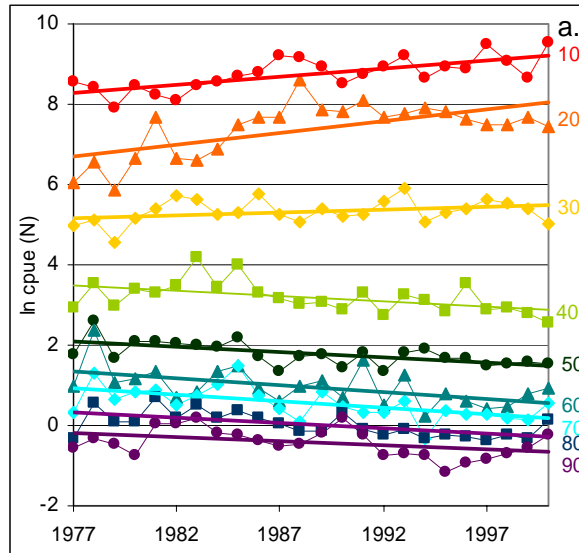
Figure 2.6: Ln cpue by size class in IBTS (taken from ICES, 2003)

There are some exploited communities for which it may be possible to define a pivot size class beyond which fish are affected by fishing and below which fishing indirect effects may lead to increased abundances of small fish (between 30 and 40 cm in Figure 2.6). Therefore, at the level of the community, the PLF may be defined as the proportion of fish that are targeted.

The PLF is an intuitive indicator and as such is one of most compelling size-based indicators for development within an EAF perspective; however, to be useful and correctly interpreted, further work must be done in order to clarify its response to ecological, statistical and fishing processes. Furthermore, the sensitivity of the PLF to fishing effects and robust to the choice of the MSLF must be evaluated. Given both the potential usefulness of the PLF, and the caveats associated with its calculation, we recommend that criteria used for defining the MSLF should be evaluated for the next meeting of the working group. This can be addressed through systematic exploration of data series and model output.

2.2.4   A new indicator: predation size spectra

2.2.4.1   Introduction

Predator–prey body size ratio for fish can be modelled using a log-normal distribution (Floeter and Temming 2003, Benoît and Rochet 2004). Metabolic rate and energy demand (i.e., food consumption rate) also are a function of body size (Peters 1983). These two simple and generally accepted allometric relationships, after parameterization, allow us to determine what prey sizes a predator is most likely to target to satisfy its demand. This information, combined with data on the abundance of predators of different size in a community, permits the calculation of a predation size spectrum (PSS) for the community (Duplisea 2005). Specifically, this PSS can be plotted with body size of prey on the x–axis, and the potential consumption rate of each prey size by all predators in the community on the y–axis. Such a PSS may reveal the prey sizes in a system most heavily targeted by predators, given the predator size distribution.

Time-series of predator size and abundance can be used to show how a PSS changes over time. A PSS models community predation with only few assumptions, compared with complicated community and ecosystem models.

An annual PSS for the northern Gulf of St Lawrence was calculated using predator size and abundance data for the fish community from surveys. Data were included for harp seals (*Phoca groenlandica*), because they are by far the most abundant marine mammal in the system, and include a large proportion of fish in their diet. The fish and seal PSS allow us to examine if changes in the Gulf of St Lawrence system characterized by declines in predatory fish and increases in seals led to changes in the sizes of prey targeted over time.

2.2.4.2   Methods

An annual summer stratified random sampling survey (Doubleday 1981) has been conducted in the northern Gulf of St Lawrence since 1984. Gear and vessel were changed in 1991, after a calibration year in 1990. Data used here are from 1984 to 2003, and corrected data from 1984 to 1990, for differences in sampling efficiency between the two vessels (Harley and Myers 2001, Savenkoff *et al.*, 2004). Harp seal data were updated from Hammill and Stenson (2000). This data set (1960–2002) is a reconstruction of harp seal abundance, but taking into account harvest and modelled pupping rates.

Predator size spectra were constructed by first calculating the food required by a single average predator in each size class, then multiplying the abundance of predator size to obtain an estimate of the total food required. The fish predator size range is taken as that caught in the survey. The proportion of consumption derived from a single prey size is then determined from the log-normal predator–prey size ratio, where the sum of preference over the prey size range is normalized over the prey size range considered. This is tantamount to saying that a predator eats only within this prey size range. Predation on each prey size by all predators is then calculated by summing the prey consumption in each size class over all W. Finally, a relative PSS is obtained by dividing size class predation values by the total predation. The relative PSS can easily be transposed into an absolute PSS if reliable estimates of absolute predator abundance at size are available.

A $\log_2$ predator–prey weight ratio for fish used with was 6.64 (= $\log_2$ 100) and s.d. = 2 which is consistent with the observed range for North Sea cod (Floeter and Temming 2003). For harp seals, a predator/prey weight ratio of 10 was used (= $\log_2$ 1000; Hammill and Stenson, 2000), with s.d. = 3.32. Daily consumption rate for fish was based on an allometric formulation (Magnússon 1995) used in MSVPA for North Sea cod (ICES 1998). Daily consumption rate for harp seals was taken from (Hammill and Stenson 2000).

2.2.4.3   Results

Most predation by fish is targeted at two prey size classes, spanning sizes of about 0.5 g to 2 g (Figure 2.7a), which roughly corresponds to 4–6.5 cm for cod-like fish. Before the collapse of the northern Gulf of St Lawrence cod stock in 1993 and the fishing moratorium in 1994, predation pressure on the small fish size range was considerably greater than thereafter. Total predation by fish peaked in 1987 and, though steeply declining, remained quite high until about 1990.

Most seal predation is targeted at prey of 60g–125 g (Figure 2.7b), roughly corresponding to 18–23 cm for cod-like fish. This accords well with the cod sizes reconstructed from otoliths found in harp seal stomachs (Hammill and Stenson 2000). Figure 2.7b also suggests though that harp seals would inflict a substantive though lesser predation pressure on larger and smaller prey.
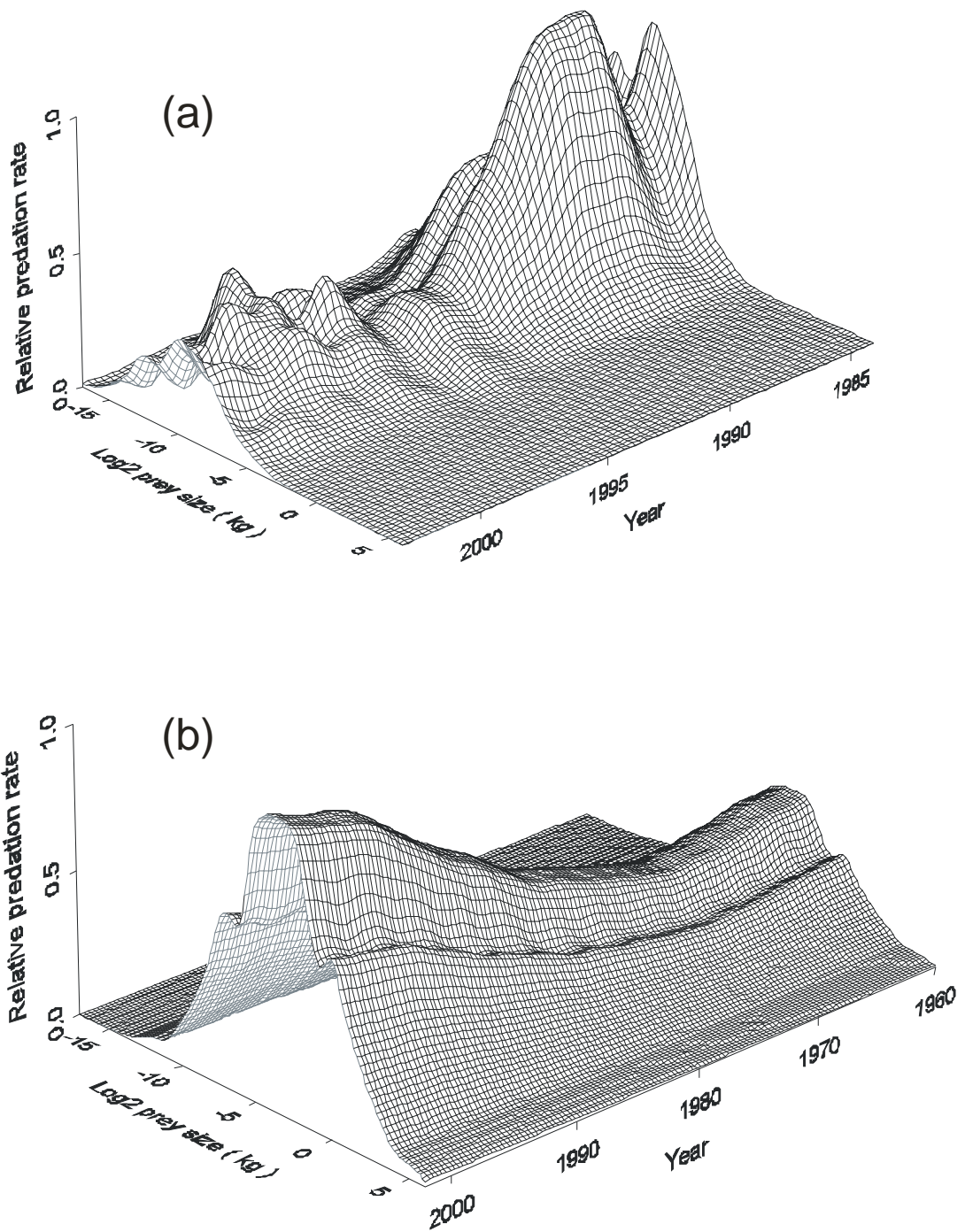
Figure 2.7: Relative predation size spectra calculated for (a) fish predators and (b) harp seals in the northern Gulf of St Lawrence.

2.2.4.4   Discussion

The PSS method provides an alternative to detailed modelling of predation, and may aid directly in single species assessment by informing decisions on changing input natural mortality in cohort reconstruction approaches. Furthermore, predation size spectra represent a general indicator of the predation environment experienced by all fish that can be used to uncover major changes in the predation patterns of an ecosystem. This could be a valuable tool for

evaluating if and when a community has made a large-scale state transition (Scheffer *et al.*, 2001), because the predation environment would likely change in conjunction with state.

In this particular application, however, there are three main players, cod, harp seals, and the fisheries. The northern Gulf of St Lawrence cod fishery was closed in 1994, and only small fisheries have been allowed ever since (CSAS 2003), although F may still be high, because the stock biomass is small. For the period from 1991 onwards, recruitment has been consistently low, even though spawning stock biomass (SSB) varied almost sixfold over the period (CSAS 2003), suggesting that recruitment is a function of more than SSB. In stock assessment, natural mortality of post-recruit cod is assumed to have doubled since 1986, to M = 0.4 (CSAS 2003). It is conceivable that the reasoning behind an increased M for post-recruits also applies to pre-recruits. For instance, the elevated predation owing to increasing seal numbers could potentially create a significant increase in pre-recruit mortality. Although predation by fish on smaller pre-recruits has probably decreased considerably since 1984, owing to general declines in predatory fish abundance, the increase in the numbers of harp seals could have shifted the predation mortality to larger pre-recruits.

Cod begin to recruit to the fishery only at age 3, some 400–500 g, and are not fully recruited until age 8. Therefore, the high potential predation on pre-recruits shown by the PSS in the late 1980s may have contributed to depressed recruitment when the stock collapsed in 1993. Exacerbating the large F on an already depleted stock, predation of pre-recruit cod could have precluded the stock from making any recovery since the 1994 moratorium.

Fisheries are size-selective, and one of the most common observations on exploited stocks is that average size of fish decreases with increasing exploitation (Hilborn and Walters 1992). Because predation is generally concentrated on the smaller sizes, the proportion of a fish stock that is vulnerable to predation is likely to increase with exploitation. This could be especially true when a major predator is a seal targeting relatively large pre-recruits, that would have a high potential to recruit to the fishery. In systems with a great abundance of predators targeting smaller pre-recruits, the chance that those recruit to the fishery is anyhow lower, because they must endure a high mortality environment for a longer time. Hence, different units of predator biomass may affect recruitment disproportionally, and a nominally equal amount of predation changing from small to large prey is likely to reduce recruitment.

The effect of the distribution assumed for the predator/prey size ratio used to derive the PSS has not been evaluated. It is assumed to be log-normal in accordance with empirical observations taken from stomach sampling. The stomach content of fish, however, does not represent only active prey selection by fish but are also a function of passive selection, i.e., the abundance of prey of different body sizes in the environment. Therefore, what appears to be log-normal active prey selection from stomach data could result from a more uniform (non-selection) of prey within a size range combined with different abundance of prey of different size. Additionally, gastric evacuation rates will vary by prey species and size for a predator and that can affect what is sampled from stomach contents. Furthermore, some species groups will consume prey outside these ranges and even larger than themselves, therefore the distribution used here is not universal but is likely to be broadly robust. Given the potential confounding factors determining the active prey size selection distribution for a predator, various distributions are plausible and sensitivity of the result evaluated in relation to the distribution assumption.

If the prey-size preference of harp seal is constant from year to year, and prey is always selected between 60 and 125 g, such peaks in predation could very well create a selection pressure on prey size. This suggests that it would be evolutionarily advantageous for prey to either avoid growing to the most intensively preyed upon size range, or to grow quickly out of that range. As fish predators strongly target prey between 0.5 and 2 g, there would be a small respite in predation mortality between 2 and 60 g (though both seals and fish will eat prey in this

range, with lower preference). Therefore, the best strategy is for a fish to grow out of the pre-recruit size range as quickly as possible. Fisheries, on the other hand, create a selection pressure for fish that mature early and remain small (Olsen *et al*., 2004). Locked between the threats of a natural predation environment and selective fisheries, fish such as cod experience a variety of mortality gauntlets during their lives that may influence patterns of growth, size, and maturity in different directions.

2.3 Using models as simulation environments to test the sensitivity of indicators to fishing

Simulation environments for testing the efficacy and robustness of management strategies and harvest control rules are rapidly becoming a key area of research in the ICES community (e.g., SGMAS 2005). These environments are seen as one of the only methods for systematically evaluating how various biological indicators of stocks and management responses to these indicators are likely to behave in practice. In the context of ecosystem based management initiatives, science must provide advice to help managers fulfil objectives related to community or ecosystem state. It is widely acknowledged that community and ecosystem indicators will be essential for doing this and accordingly suites of metrics have been proposed (SCOR/IOC symposium "Quantitative Ecosystem Indicators for Fisheries Management" held in Paris in April 2004 www.ecosystemindicators.org) with much focus on size-based indicators. Size-based indicators are some of the best studied and understood indicators of community state yet the sensitivity and robustness of these indicators to changing fishing effort are known only roughly, usually from empirical studies with limited ranges of effort in a few northern systems with long multispecies survey time series. As a result, we can only infer how indicators might respond to changing effort outside of the empirically observed ranges or in a different fish community. It is likely that at different effort levels, some indicators may be more appropriate than others to show fish community changes to fishing or in communities with different characteristics such as species composition. Clearly, we need to explore these aspects of community indicators before we can know their sensitivity and behaviour under various circumstances and be comfortable providing management advice based on their assessment of community state.

Here we present evaluation of community metrics from two different models of fish communities. Various indicators (species-based, size-based and trophic) in a modelled Southern Benguela community at both the species and community level at different levels of F were evaluated using Shin's (Shin and Cury 2001, 2004) individual based modelling approach. Changes in equilibrium size and diversity spectra, and k dominance curves were evaluated at different long term community F strategies for Hall *et al*.'s (Hall *et al*., manuscript) length-based MSVPA type of model parameterised using life history data and characteristics from the George's Bank fish community. These two models are vastly different in structure and therefore evaluating indicator behaviour between them is a test of indicator robustness. Additionally, models of this sort, if parameterised based on a real community, can suggest the value of various indicators in the unexploited state. We recommend that further work along these lines would be very useful, especially if conducted in parallel such that indicators can be directly compared over model.

2.3.1 Sensitivity of ecosystem-based indicators to fishing using a simulation approach for the southern Benguela

2.3.1.1 Introduction

The use of simulation models may help evaluating the robustness of ecological indicators and understanding their response to fishing by systematically exploring different fishing scenarios and by providing a virtual laboratory in which all biomass and abundance values are accessi-

ble. Although this case study concerns the southern Benguela fish community, the conclusions and recommendations may apply to other exploited communities.

The SCOR/IOC symposium "Quantitative Ecosystem Indicators for Fisheries Management" held in Paris in April 2004 (www.ecosystemindicators.org) reviewed studies about ecosystem indicators and it appears that there are many indicators, probably with some redundancy, that could be used to evaluate fishing impacts on ecosystems (Cury and Christensen, 2005). The aim of this section is to study the sensitivity of a set of indicators to fishing, to help in selecting a few indicators on the basis of their response to fishing. A simulation approach is adopted, based on the application of the individual-based model Osmose (Shin and Cury, 2001, 2004; Shin *et al*., 2004) to the southern Benguela ecosystem during the 1990s. Different fishing scenarios and levels of fishing mortality are simulated so as to analyse changes in the indicators in a wide range of possible configurations of the southern Benguela ecosystem.

2.3.1.2   Materials and methods

2.3.1.2.1   Indicators

The sensitivity to fishing of the main ecological indicators reviewed in the SCOR/IOC symposium is evaluated in the southern Benguela ecosystem (Cury and Christensen, 2005). Three main categories of ecological indicators can be distinguished, and are considered in this study: size-based, species-based and trophodynamic indicators. All indicators used in this study are described in Tables 2.3–2.4. They were chosen according to their common use, their ecological meaning and their theoretical sensitivity to fishing pressure.

Table 2.3: Population indicators. L represents fish body length in cm, Ni the abundance of species i without age-class 0, Ci is the number of fish of species *i* in the catches.

INDICATOR	NOTATION	DESCRIPTION	UNITS	KEY REFERENCES
Mean body length	L_i	$L_i = \dfrac{1}{N_i} \sum_{N_i} L$	cm	Haedrich and Barnes 1997; Babcock *et al.* 1999
Mean body length in catches	Lc_i	$Lc_i = \dfrac{1}{C_i} \sum_{c_i} L$	cm	Rochet and Trenkel 2003

Table 2.4: Community indicators. L_i represents the mean body length of species i, Ni and Bi its abundance and biomass without age-class 0, N is the total abundance of the community without age-class 0, C is the number of fish in the catches, Y their biomass in the catches, pi is the proportion in abundance of species i in the community, S is the number of species in the community.

INDICATOR	NOTATION	DESCRIPTION	UNITS	KEY REFERENCES
Mean body length	L	$$L = \sum_i L_i * \frac{N_i}{N}$$	cm	Rochet and Trenkel (2003)
Mean maximum body length	L_{max}	$$L_{max} = \sum_i L_{\infty,i} * \frac{N_i}{N}$$ $L_{\infty,i}$ is the infinity length of the von Bertalanffy model	cm	Jennings *et al.* 1999
Size spectrum - Slope - Curvature	α γ	$$Ln(N_k) = \alpha + \beta Ln\left(\frac{L_k + L_{k-1}}{2}\right)$$ $$Ln(N_k) = \alpha + \beta Ln\left(\frac{L_k + L_{k-1}}{2}\right) + \gamma\left[Ln\left(\frac{L_k + L_{k-1}}{2}\right)\right]^2$$ With k the size index	-	Rice and Gislason 1996; Gislason and Rice 1998; Bianchi *et al.* 2000; Shin and Cury 2004;
Shannon-Wiener index	H'	$$H' = -\sum_i \frac{N_i}{N} * \log \frac{N_i}{N}.$$	-	Magurran 1988
Slope of Diversity size spectrum - Slope	$\alpha Slope_{DSS}$	$$Log(H') = \alpha DS + \beta DS\left(\frac{L_k + L_{k-1}}{2}\right)$$ With k the size index	-	Rice and Gislason 1996, Gislason et Rice 1998
W-statistic Area between abundance and biomass curves (ABC curves)	*W-statistic*	$$W = \sum_{i=1}^{s} \frac{Bcum,i - Ncum,i}{50(S-1)}$$ with *Bcum* the cumulative biomass and *Ncum* the cumulative abundance	-	Warwick 1986 Yemane *et al.* 2005
Mean Trophic Level	TL	$$TL = \sum_i TL_i * \frac{B_i}{B}$$ with TL_i the species trophic level	-	Pauly *et al.* 2001
Fishing-in-Balance Index	*FiB*	$$FiB = \log \frac{Y_t}{TE^{TL_t}} - \log \frac{Y_0}{TE^{TL_0}}$$ with TE the trophic efficiency, equals here to 0.1	-	Pauly *et al.* 2000

2.3.1.2.2   OSMOSE model

The individual-based model Osmose is based on the hypothesis that predation is an opportunistic process depending on relative body sizes and the spatio-temporal co-occurrence of a predator and its prey. This size-structured and spatial model deals with the main processes of

fish life cycle: predation, growth, reproduction and mortality. The basic unit of the model is the fish school or "super-individual", defined as a group of fish from the same species that have the same size, the same spatial coordinates and which require similar food. The model is structured by means of different classes with each fish school belonging to a "cohort" class, which in turn belongs to a "species" class. This hierarchical structure allows output variables to be tracked at different levels of aggregation (Shin and Cury, 2001; 2004; Shin *et al.*, 2004).

The time step adopted for the application to the southern Benguela ecosystem is a six-month period (January-June, July-December), corresponding to a compromise between the need to take into account seasonal variations (e.g., migration of fish) and the available data. Fish schools move in a two-dimensional grid divided in square cells (40x40) with closed boundaries, which represents the southern Benguela ecosystem extending from the vicinity of Orange River mouth (25°40'S 16°E) to East of East London (37°40'S 28°E).

Initialization

Each species modelled in Osmose is defined by several input parameters: survival parameters such as longevity, fishing mortality rates and age of recruitment, growth parameters such as von Bertalanffy model parameters and Fulton's condition factor, reproductive parameters such as age at maturity and relative fecundity. The last type of model input is the mean spatial distribution of each age class in each semester.

Carrying capacity constraint

In OSMOSE, the carrying capacity corresponds to the upper limit of the total biomass of non-piscivorous fish, which can include larval and juvenile stages but also adult fish. At the beginning of each iteration, and locally in each cell, the biomass of non-piscivorous fish is compared to the carrying capacity. If this biomass exceeds the carrying capacity, it is reduced in proportion, causing mortality for non-piscivorous fish. Furthermore, additional mortality is added in the model for larval and juvenile stages (classes of age 0) to account for the higher natural mortality they undergo (Shin *et al.*, 2004), and serves as a tuning parameter for applying the model to the southern Benguela ecosystem.

Foraging and Predation

These stages are only applied to piscivorous fish in the model. School displacement is directed by the search for the adjacent cell with the highest biomass of potential preys. Once it has moved, each piscivorous fish school feeds on fish groups of a suitable size, causing an explicit predation mortality for each fish school preyed upon. At the end of this stage a predation efficiency is calculated for each fish school. This coefficient is determined by the ratio between the food biomass ingested by a school and the food biomass required for the predator to fulfil its vital functions.

Growth and Starvation mortality

Mean growth rates in length of fish are calculated from the von Bertalanffy model. This rate is readjusted according to the amount of food ingested by each fish school during each time step. If the predation efficiency is above a critical value, the growth rate in length increases linearly with the predation efficiency. If it is below this critical predation efficiency, the food ration is too low to provide the basic maintenance requirements of fish. In this case fish schools are affected by a starvation mortality which increases when the predation efficiency decreases. Details are provided by Shin and Cury (2001; 2004).

Fishing mortality

Using Pope's approximation (Pope, 1976), fishing mortality is applied to the different species at the middle of the year, in other words between the semester January-June and the semester July-December. Fish catches are simulated uniformly across the spatial grid, therefore the spatial distributions of fishing effort is not taken into account in this model.

Reproduction

At the end of the year, reproduction of fish is simulated by adding a certain number of eggs per species in the system. The number of eggs spawned per species is calculated assuming that the sex-ratio is 1:1 and based on input parameters such as the relative fecundity of the species and its age at maturity (Table 2.5).

2.3.1.2.3   Parameterization of the model for the southern Benguela ecosystem

Considering that biomass, catches and food consumption (as it is linked to species interactions) are three main criteria in the choice of species, twelve fish species have been selected to represent the southern Benguela fish community: anchovy *Engraulis encrasicolus*, sardine *Sardinops sagax*, round herring, also called redeye, *Etrumeus whiteheadi*, horse mackerel *Trachurus trachurus capensis*, chub mackerel *Scomber japonicus*, shallow-water Cape hake *Merluccius capensis*, deep-water Cape hake *Merluccius paradoxus*, kingklip *Genypterus capensis*, snoek *Thyrsites atun*, silver kob *Argyrosomus inodorus*, lanternfish *Lampanyctodes hectoris* and lightfish *Maurolicus muelleri*. This set contains pelagic and demersal species, representing at least 76.2% of the biomass and 93.8% of the catch of all fish species, according to a study using an Ecopath model of the southern Benguela ecosystem during the 1990s (Shannon *et al.*, 2003). In terms of consumption, these species are responsible for at least 84% of the total consumption by fish species and top predators and 72% of the consumption of small pelagic fish production (Shannon *et al.*, 2003). Furthermore, these twelve species are representative of functional groups. For example anchovy, sardine and round herring constituted 97% of the biomass and 99.9% of the catch of small pelagic fish in the 1990s (Shannon *et al.*, 2003). Hake, horse mackerel, kingklip, snoek and chub mackerel represented about 70% of the biomass of commercially important demersal fish species surveyed in 2001 on the West and South coast of the Benguela and 90% of the reported demersal commercial catch over the period 1992–1999.

At model initialization, these twelve species are created from a number of input parameters describing growth, survival and reproduction (Table 2.5). Initial abundances of fish are deduced from biomasses estimated from a study applying an Ecopath model of the southern Benguela during the 1990s (Shannon *et al.*, 2003) (Table 2.5). The tuning of the model was undertaken by allowing larval mortalities to vary in order to obtain mean species biomasses in the range estimated for the 1990s.

Table 2.5: Input parameters of Osmose model for the 12 fish species modelled explicitly. L∞, K and t0 are the parameters of the von Bertalanffy growth model, c is the Fulton's condition factor, φ is the relative fecundity, amat is the age at maturity, amax is the longevity, arec is the age of recruitment, Madd is an additional mortality (due to predation by other species of the ecosystem which are not explicitly modelled), F is the annual fishing mortality rate, and B1990s is the mean biomass for the 1990s.

	GROWTH				REPRODUCTION		SURVIVAL				INITIALIZATION
	L_∞ (cm)	K (year^{-1})	t_0 (year)	c (g.cm^{-3})	ϕ (eggs.g^{-1})	a_{mat} (year)	a_{max} (year)	a_{rec} (year)	M_{add} (year^{-1})	F (year^{-1})	B_{1990s} (tons)
Anchovy	14.8	1.37	-0.03	0.007	8000	1	5	1	0.403	0.23	786 143
Chub mackerel	68	0.207	-0.98	0.005	300	3	8	2	0.143	0.07	100 000
Hake (shallow water)	270.6	0.039	-0.73	0.005	500	4	15	3	0.228	0.23	388 300
Hake (deep water)	219.4	0.049	-0.914	0.005	500	4	15	3	0.174	0.33	647 900
Horse mackerel	54.5	0.183	-0.65	0.009	250	3	8	2	0.314	0.06	532 800
Kingklip	180	0.105	0.05	0.001	500	5	24	3	0.182	0.11	11 583
Lanternfish	7	1.66	0.06	0.008	646	0.5	2	1	0.226	0.0003	1 126 620
Lightfish	6	1.15	0.06	0.008	334	0.5	2	1	0.226	0.0003	1 126 620
Redeye	30.1	0.71	0.28	0.009	750	1	6	1	0.208	0.04	1 369 720
Sardine	26	0.26	-1.50	0.009	2400	2	10	1	0.365	0.16	460 000
Silver kob	116	0.12	-1.47	0.007	150	2	25	3	0.228	0.181	10 400
Snoek	115.3	0.294	-0.1	0.018	130	3	10	2	0.132	0.25	74 140

2.3.1.2.4   Simulations

Fishing scenarios

To test the sensitivity of indicators to fishing pressure, different fishing scenarios are simulated. Each scenario is run 25 times for 200 years, and only the 100 last years of each simulation are kept to calculate the average of the indicators. The first scenario simulated is an increase of the global fishing mortality, using a multiplier of fishing mortality (mF) for all species ranging from 1 to 2.5 with a step of 0.1. The second scenario corresponds to an increase of the fishing mortality of hakes (shallow and deep water). For the third scenario, an increase of the fishing mortality of small pelagic fish, i.e., for sardine, round herring and anchovy, is simulated. Finally, the fourth scenario corresponds to the development of a mesopelagic fishery targeting lanternfish and lightfish, by increasing the fishing mortality (F) of both species from 0 to $1y^{-1}$.

Reference state

To compare the relative variation of indicators in each fishing scenario, a reference state is defined. This reference state is chosen as the simulation of the fisheries state from 1990 to 1997 in the southern Benguela ecosystem. This reference state is also the one used for tuning the model. The Osmose model provides output species biomass, species abundance and species mean size, in the ecosystem and in the catches, globally and per size class. From all these outputs, the indicators previously described can be calculated and compared. To compare the sensitivity of the whole set of indicators on the same scale, indicators are graphically presented as the ratio between their value in each scenario and their value in the reference state (initial tuned 1990s model). For the case of the FiB index, the reference value equals 0 by definition. In order to compare it with the other indicators, FiB values were incremented by 1 to facilitate use of the ratio relative to the reference state for this indicator too.

2.3.1.3   Results

2.3.1.3.1   Simulations of the fishing scenarios

The variations of abundance of each species in each scenario are presented in Figure 2.8. It could be noted that the relative variations of biomass for each species in each scenarios are quite the same than the relative variations of abundances.

Increasing fishing mortality for all species leads to many species collapses (Figure 2.8a). By contrast, two species (horse mackerel and round herring) seem to take advantage of the ecosystem disturbance and display large increases in abundance.

As expected, hake abundance decreases in the second scenario (Figure 2.8b), but indirect effects of heavy hake fishing are also observed. The abundance of kingklip, which is a competitor of adult hakes, the abundance of horse mackerel, which is a competitor of juvenile hakes, and the abundance of sardine and round herring, which are prey of hakes, increase as hakes abundance decreases.

As expected, the scenario of small pelagic fish (Figure 2.8c) shows a decrease of the abundance of anchovy, sardine and round herring. Most species abundance decreases in parallel. However, despite a decrease in small pelagic fish, which are prey for hake, deep-water Cape hake shows a marked increase, possibly as a result of reduced competition between small deep-water Cape hake and other fish species. In addition, the small relative increase in mesopelagic abundance (+1.5%), which are the main prey for deep-water hake, corresponds actually to an absolute increase which is not negligible knowing that mesopelagic initial biomass is the highest among the species modelled (Table 2.5).

By targeting the lower trophic level, the mesopelagic fish scenario (Figure 2.8d) illustrates a possible bottom-up effect of fishing. It induces a decrease in mesopelagic fish abundance, which in turn causes a decrease of the abundance of the predators deep water hake and silver kob. On the contrary, the abundance of competing species like small pelagic fish (anchovy and sardine) increases greatly.
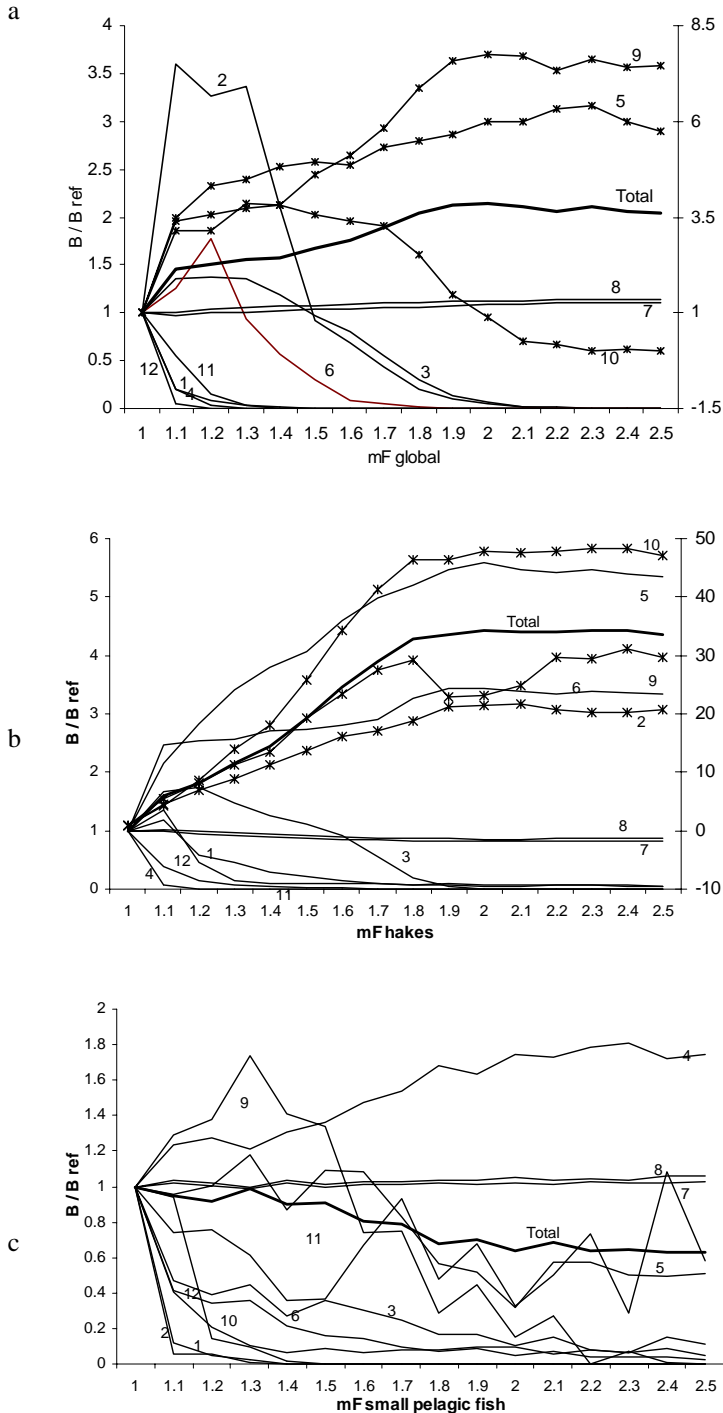
Figure 2.8: Relative changes in the abundance of the 12 modelled species in each fishing scenario: a) Global scenario, an increase of the fishing mortality for all species, b) Hake scenario, an increase of the fishing mortality for deep- and shallow-water hakes, c) Small pelagic scenario, an increase of the fishing mortality for anchovy, sardine and round herring, and d) Mesopelagic scenario, an increase of the fishing mortality of lanternfish and lightfish. The ratio between the simulated abundance N and the reference abundance Nref is represented for each species against a multiplier of fishing mortality (mF) or against the fishing mortality (F) in the case of the mesopelagic scenario (d). Species are indicated by numerals (1: anchovy, 2: chub mack-

erel, 3: shallow water hake, 4: deep water hake, 5: horse mackerel, 6: kingklip, 7: lantern-fish, 8: lightfish, 9: round herring, 10: sardine, 11: silver kob, 12: snoek) and the total abundance is represented by the thick line. In a) and b), curves marked with crosses should be read on the second y-axis on the right (species 5 and 9 in case (a) and species 2, 6 and 10 in case (b)).
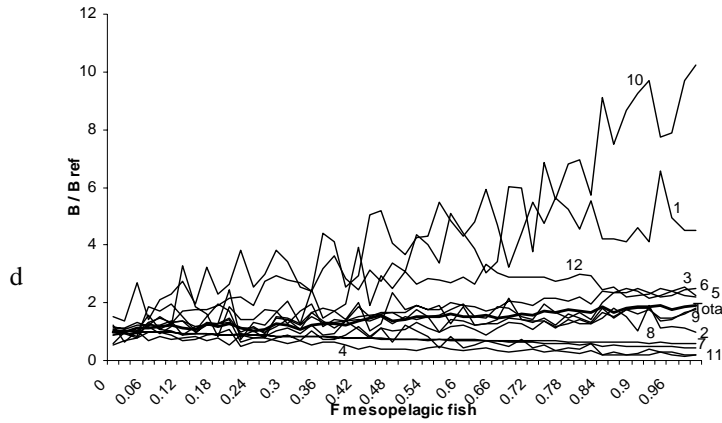
2.3.1.3.2 Species indicators

Species indicators help to explain the response of community indicators to an increasing fishing mortality. They are presented in Figure 2.9 for the global scenario. Contrary to theoretical predictions, the mean size of the fish community increases with global fishing mortality (Figure 2.9a). This trend can be explained by the increase in mean size of large fish such as kingklip and shallow-water Cape hake (Figures 2.9a) but above all with the strong decrease of the abundance of some small-sized fish species which were dominant in the initial reference state, such as anchovy (Figure 2.8a). The increase of the mean size of the community in catches whereas all the species mean sizes decrease (Figure 2.9b) confirm the cause of variations of this indictor to be mainly due to a decrease of abundance of small-sized fish species.

a)

b)
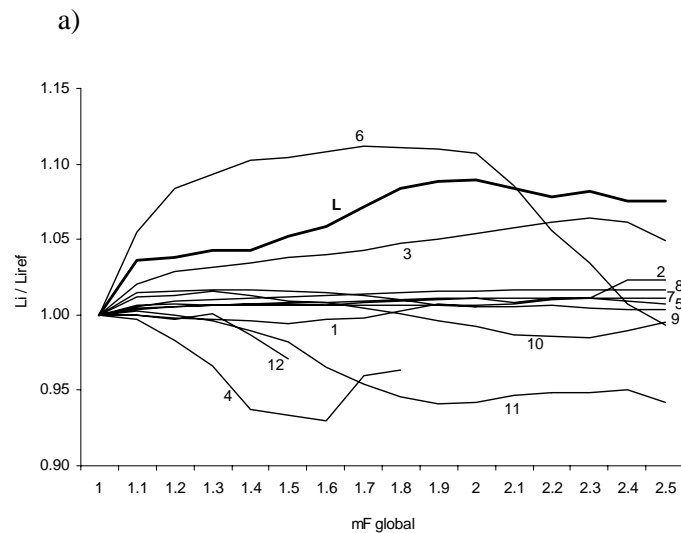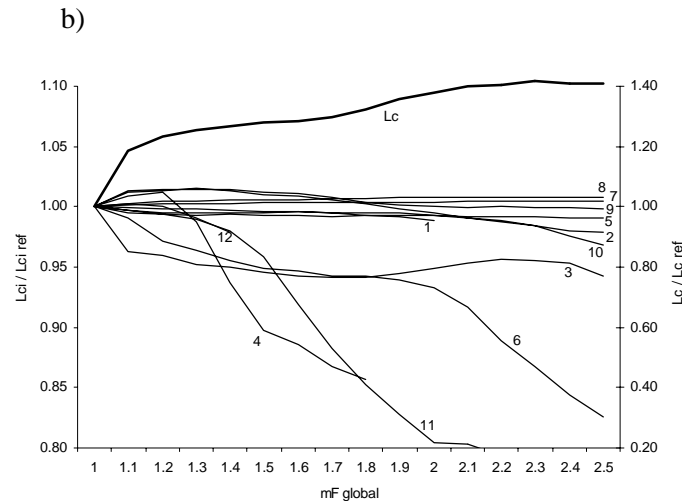
Figure 2.9: Mean size for the global scenario in the ecosystem (a) and in catches (b). The ratio between the simulated mean size and the reference mean size is represented for the twelve species (1: anchovy, 2: chub mackerel, 3: shallow water hake, 4: deep water hake, 5: horse mackerel, 6: kingklip, 7: lanternfish, 8: lightfish, 9: round herring, 10: sardine, 11: silver kob, 12: snoek) and for the community (L). The mean size of the community should be read on the second y-axis on the right. A stop in the line of a species mean size means the species collapses.

2.3.1.3.3   First-order community indicators

First-order indicators do not vary in the same direction and with the same amplitude in the four fishing scenarios (Figure 2.10). The mean size in the community decreases in the small pelagic fish scenario (Figure 2.10c) whereas it increases in the three other scenarios (Figures 2.10a-b-d). The mean maximum size of fish in the catch responds with high amplitude in the small pelagic fish scenario (Figure 2.10c) whereas it varies little in the hake scenario (Figures 2.10b). It can also be noted than first-order indicators in the global scenario vary in quite the same way than indicators in the hakes scenario (Figure 2.10a and 2.10b), whereas their variations are very different from the small pelagic fish and the mesopelagic fish scenarios. On the other hand, some indicators appear to vary concomitantly in all scenarios. For example, the mean size and the mean maximum size (in the ecosystem or in catches), have the same direction of change in each scenario (Figure 2.10). It can be noticed that, in all cases, mean maximum size is more sensitive to fishing mortality than mean size. The Shannon index H' also varies in a similar way to mean and mean maximum size in the ecosystem. By contrast, the mean trophic level does not seem to vary like other first-order indicators. Furthermore, its amplitude of variations is smaller than for the other indicators. Finally the FiB index displays various responses to fishing: it increases in the small pelagic and mesopelagic scenarios and it shows two direction of variation according to fishing intensity for the global and the hake scenarios.

2.3.1.3.4   Second-order community indicators

Second-order indicators, or indicators calculated from distributions, are presented for the four scenarios in Figure 2.11. They show large amplitudes of variation, particularly in the curvature of the quadratic size spectrum. In each scenario the slope of the size spectrum varies quasi linearly with fishing mortality, as it has already been shown by other studies (Gislason and Rice, 1998; Shin and Cury, 2004). The curvature of the quadratic fit of the size spectrum varies with a higher amplitude than the slope for each scenario. However, its response curve to fishing is more difficult to analyse (particularly in the global scenario, Figure 2.11a). The indicator w-statistic displays the same direction of variation in all four scenarios: it decreases in relative value (increases in real values as w-statistic takes negative values in this study) when the fishing mortality increases.

Diversity size spectra are fitted by a straight line, with a high determination coefficient ($0.88 < R^2 < 0.97$ for the four scenarios), compared to the regressions made in some empirical studies (e.g., Gislason and Rice, 1996; Bianchi *et al*., 2000). However, the slope of the diversity size spectrum does not vary significantly with fishing mortality (the slope of this indicator against fishing mortality is not found to be significantly different from 0 for all scenarios). Therefore, this indicator is considered not to be sensitive to fishing.
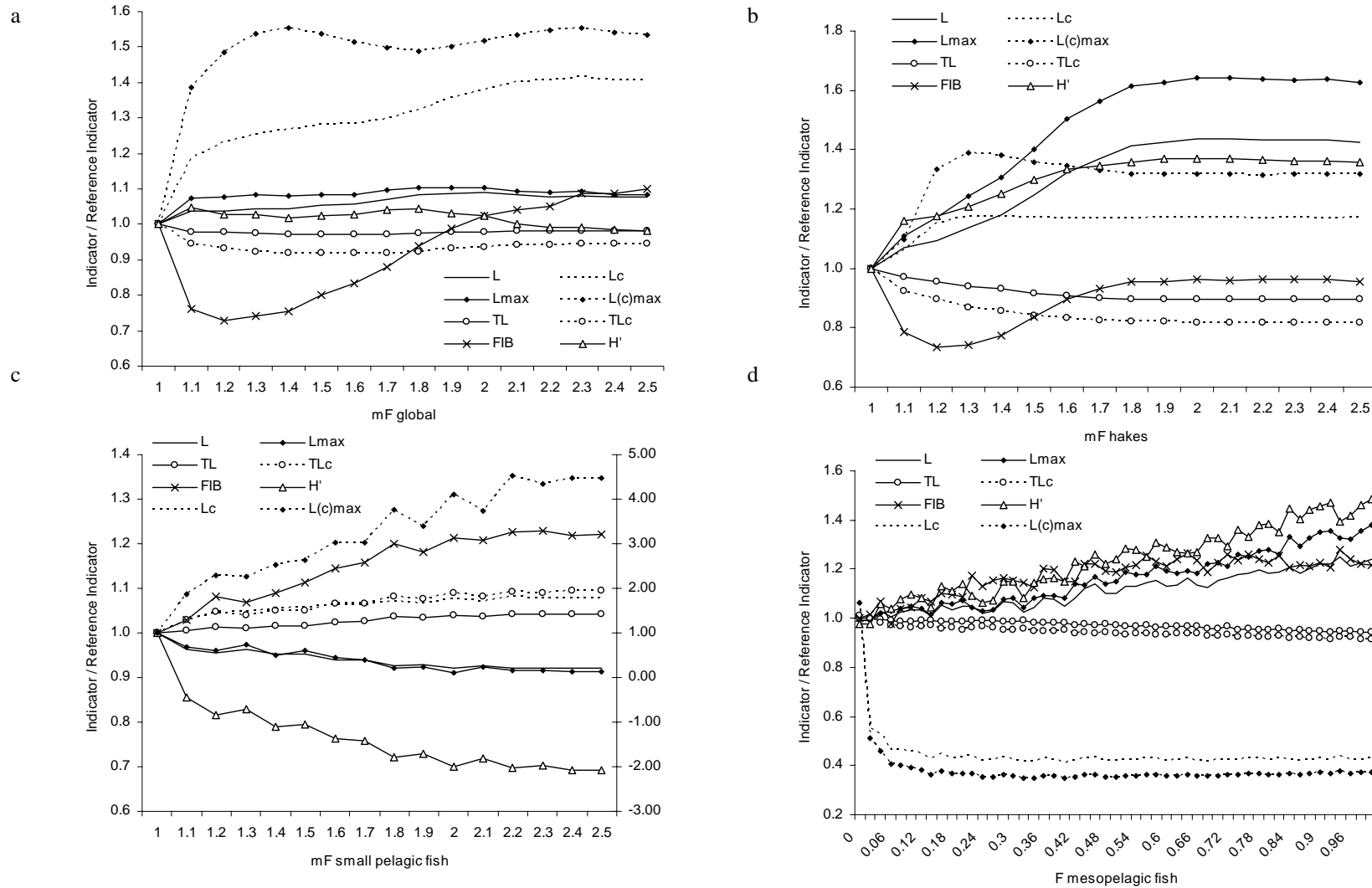
Figure 2.10: First-order indicators for each scenario: a) Global scenario, b) hake scenario, c) small pelagic fish scenario, d) mesopelagic fish scenario. Indicators are represented as a ratio between the simulated indicator value and its reference value. L is the mean size in the ecosystem, Lc is the mean size in catches, Lmax is the mean maximum size in ecosystem, Lmaxc is the mean maximum size in catches, TL is the mean trophic level in ecosystem, TLc is the mean trophic level in catches, FiB the fishing-in-balance index incremented of 1 (see text) and H' is the Shannon diversity index. In the small pelagic fish scenario (c), Lc and Lmaxc should be read on the second y-axis on the right.
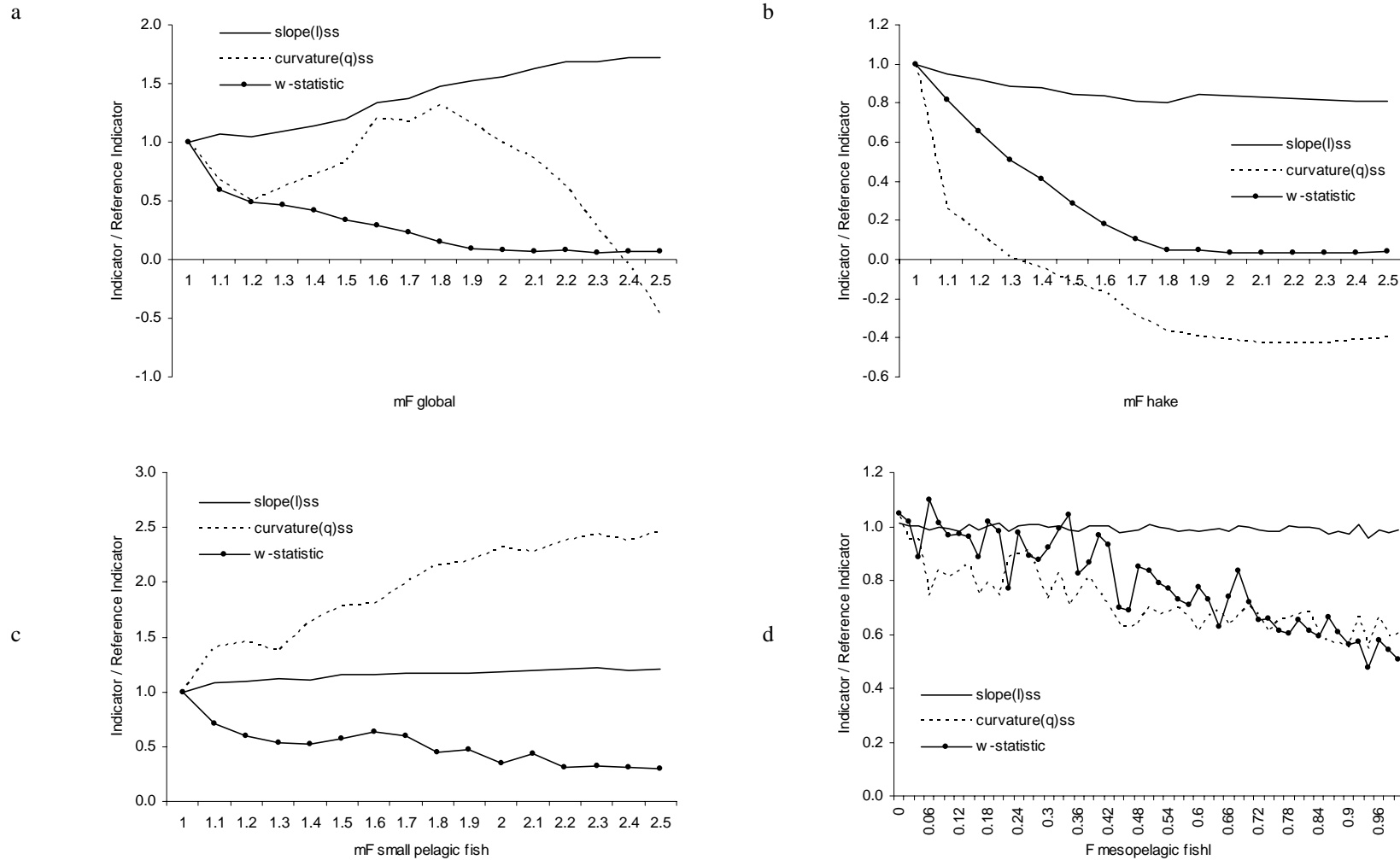
Figure 2.11: Second-order indicators for each scenario: a) Global scenario, b) hake scenario, c) small pelagic fish scenario, d) mesopelagic fish scenario. The slope is estimated by fitting a linear curve to the size spectra, curvature is estimated by fitting a quadratic curve to the size spectra. Indicators are represented by the ratio between the simulated value and the reference value.

2.3.1.4   Discussion

2.3.1.4.1   Community indicators *vs.* species indicators

In the frame of a global increase of the fishing mortality of all species, community indicators and species indicators do not vary in the same direction and with the same amplitude (Figure 2.9). These indicators do not reflect the same processes, but they are often correlated, notably by the way that community indicators are calculated. In our simulations, the mean length of fish in the southern Benguela fish community displays a direction of change that was opposite to the theoretically predicted direction or to the observed direction in specific communities, e.g., demersal (Rochet and Trenkel, 2003; Shin *et al.*, 2005). This result highlights the fact that community indicators reflect non-trivial properties, which emerge from complex and hardly predictable species interactions. Therefore, they must be analysed in parallel with population indicators for understanding underlying processes. Because the mean size of the community is calculated by the weighted average of the mean size of the species, the increase of this community indicator (Figure 2.9) can be either explained by variations of species indicators, or by variations of the relative abundances of small and large species. In the ecosystem, with an increase of the global fishing mortality, the increase of the mean length of the community seems to be caused by the collapse of a dominant species with a small mean size (e.g., anchovy which was initially abundant), counter-balanced by a relative large increase of abundance of species a little larger (such as round herring and horse mackerel). Moreover, the increase of mean length of the community is also due to the increase in the mean length of large species such as kingklip and shallow-water Cape hake (Figure 2.9).

2.3.1.4.2   Indicators in ecosystem and in catches

Catches are sometimes used as a sample of the ecosystem, but indicators in catches and in ecosystem do not have the same sensitivity or even change in the same directions. Indicators in catches concern only recruited stages, whereas indicators in the ecosystem would reflect the whole population, including pre-recruit stages. By comparing variations of indicators in the ecosystem and in catches, the respective role of recruits and of pre-recruits (and beyond this simulation framework, targeted versus non-targeted fish species or size classes) can be better understood. As an example, in Figure 2.9, the decreasing mean size of kingklip in catches allows us to understand that the increase in the mean size of kingklip in the ecosystem is either due to decrease of the abundance of pre-recruited stages, in other words a recruitment overfishing (indirect effect of fishing) or to a higher size at age of pre-recruits. Therefore indicators in catch are complementary to indicators in the ecosystem because they highlight which part of the life cycle of a species is responsible for changes in the indicators. Furthermore, the interplay between targeted and non-targeted species would be captured by the comparison between indicators in ecosystem and in catch, but it is not simulated here because all species included in the model are subject to fishing.

2.3.1.4.3   First-order indicators and weightings with abundance or biomass

For each scenario, many of the first-order indicators vary in the same way (Figure 2.10). For example, trophic level in the community and in catches present quite the same sensitivity, or mean size and mean maximum size have very similar variations, with just a little difference of amplitude. Conversely the mean trophic level does not present the same direction and amplitude of relative variation as the mean maximum size. The latter indicator represents the relative proportion of large and small species, which empirically has been shown to correspond globally to the proportion of species with high and low trophic level (Jennings *et al.*, 2001; Pauly *et al.*, 2001). Contrary to expectations, the differences or similarities of behaviour of some indicators can be explained by the weighting used for their calculation, rather than by their ecological meaning. Indeed, mean size, mean maximum size and the Shannon diversity index are weighted by abundances. Conversely, the mean trophic level is weighted by bio-

masses. It can be noticed that the FiB index does not vary in the same way as the trophic level in the catch. It is the only first-order indicator that still continues to change after the global mF exceeds 1.8, maybe because it accounts for the global biomass and is not calculated as a simple weighted average as the other first-order indicators.

Thus, depending on the way of weighting, community indicators show similar trends. Therefore, when evaluating fishing effects, it is important to track changes in at least two types of indicators weighted differently to detect different aspects of ecosystem changes. It must be noticed that the weighting by abundance can hide some processes by overweighting some species, like mesopelagic fish, that are very abundant in the Benguela ecosystem but are not commercially important.

The comparison of the indicators weighted by abundance can improve our understanding of the response of the ecosystem to fishing pressure. Indeed, as these indicators vary in the same way, it suggests that variation of the mean size in the community is mostly due to variation of the relative abundances of species (rather than to variations in the mean size of the species), just like the mean maximum size of the community, which is calculated from fixed maximum size of the species. For the four scenarios, fishing effects at the level of the community appear to be more inter-specific than intra-specific. Because diversity (H') and size-based indicators (L and Lmax) vary in the same way within a given scenario, they could be considered as being redundant in the frame of these simulations, reflecting changes in the relative abundances of the species. However, in some other configurations, mean length and mean maximum length of the community should help discriminate variations in length due to inter- or intra-specific changes.

2.3.1.4.4   Second-order indicators

As predicted by theory and reported in empirical studies (see review in Shin *et al.*, 2005), the slope of size spectra increases in absolute value with increasing fishing mortality in the global scenario (Figure 2.11a). Like in other modelling studies (Gislason and Rice, 1998; Shin and Cury, 2004), it varies linearly but with smaller amplitude for the three other scenarios than for the global one. Conversely, the curvature of size spectra when fitting a quadratic curve varies with high amplitude. Compared to the slope estimated by the linear regression, the curvature may be more sensitive to variations in abundance of very small fish in the size spectrum, because of the wider range of size used in the distribution, including small fish. However, the curvature does not always respond linearly to fishing mortality, which can make the interpretation of this indicator difficult (especially in Figure 2.11a). For example, in the global scenario, the initial decrease of the absolute value of the curvature (which is negative) until mF=1.2 is attributed to a high abundance of kingklip (the largest species after hakes) which then begins to decrease (where the relative value of the curvature starts to increase) and collapses around mF=1.7 (Figure 2.8a). The final relative decrease of this indicator can be explained by the delayed decrease in the biomass of sardine (Figure 2.8a). So the curvature of size spectra is very informative about the size structure but also the species composition of the fish community. Then, as slope and curvature do not inform us about the size structure of the same part of the community (respectively not including and including small size classes), in order to catch the maximal information, both characteristics of size spectra may be worth being studied. Furthermore, conversely to first-order indicators, second-order indicators do not provide the same variations in the global scenario and in the hakes scenario (Figures 2.10–2.11). These indicators may be more adapted than first-order indicators to track changes caused by different fishing pressure.

The response of the w-statistic to fishing is ambiguous. On the one hand, it presents a consistency in its direction of change and a quite high sensitivity. For all scenarios, the w-statistic

varies only in one direction, it decreases in absolute value as the fishing mortality increases. But on the other hand, this direction of change is not consistent with the underlying theory. Warwick and Clarke (1994) proposed the w-statistic as an indicator of the degree of global disturbance of the ecosystem. In the southern Benguela ecosystem, Yemane *et al.* (2005) reported a decrease in the w-statistic calculated for the demersal fish community only and related it to an increasing fishing pressure. Thus it appears that the use of the w-statistic can be biased in communities dominated by few species like in our case study where mesopelagic fish and small pelagic fish clearly dominate the system in abundance and in biomass. Furthermore, by exhibiting the same direction of change under different fishing scenarios, the w-statistic does not allow to discriminate which species or species groups are affected by fishing.

2.3.1.4.5   Range of sensitivity

The simulations allowed highlighting the fact that the sensitivity of indicators is not constant over a wide range of fishing mortality values. A range of sensitivity must be defined within which indicators may be useful for detecting fishing effects. For example, in the global scenario, the mean maximum size in catch does not vary after the threshold mF=1.3. Conversely, other indicators have the interesting property to be continuously sensitive to fishing, like the slope of the size spectrum, which varies linearly with increasing fishing mortality in the global and in the hake scenario. Simulations allow exploratory analyses of the range of sensitivity of indicators and to precise the response curves of indicators to fishing (e.g., linear, existence of a plateau or a dome), which is something difficult to test empirically. They can help selecting the relevant indicators in lightly or heavily fished situations.

2.3.1.5   Conclusion

The comparison of indicators in different fishing scenarios highlights the fact that the same indicators can increase or decrease from the same initial state according to which species are highly exploited. When considering all possible fishing scenarios, there are no consistent directions of reference for the indicators studied because depending on the functional group or size classes targeted, the response of the ecosystem to fishing will differ. This study shows that indicators do not always vary as predicted by theory, because indirect effects of fishing on the different components of the ecosystem are hardly predictable. Therefore a simulation approach can be very useful for better understanding the emergent properties of a particular ecosystem under fishing pressure. It highlights also the fact that when proposing reference directions of change for indicators in an EAF perspective, the fishing configuration (species targeted, fishing intensity) and the community studied should be carefully specified. For the southern Benguela upwelling ecosystem, the second-order indicators appear to be relatively sensitive to fishing pressure, in a wide range of fishing mortality and with an expected direction of change. The use of different kinds of indicators is needed to obtain a good picture of the ecosystem because the way indicators are calculated is informative on ecosystem structure: second-order indicators, first-order indicators weighted by abundance and biomass, species indicators. In addition, the ecosystem dynamics reflect a diversity of processes which can be analysed by the use of diverse types of indicators either size-based, species-based or trophodynamic indicators.

2.3.2 Community indicator sensitivity to fishing from a simulated Georges Bank fish community

2.3.2.1 Introduction

Quantitative ecosystem indicators are needed to fulfil the mandate for ecosystem-based fisheries management. A suite of community indices could potentially be used, but reference levels for these indices have not been determined and their sensitivity to fishing and other disturbances is unknown.

A size-structured model was developed for a multispecies fish community with a realistic distribution of life-history attributes. To maintain a level of realism and help ensure internal consistency, we have chosen to develop a model that is loosely based on a single real fish community. We have selected the Georges Bank fish community for this purpose, owing to the large body of information on the fish species and their dynamics in this region. We emphasise that we are not seeking to describe the dynamics of Georges Bank at the expense of generality. Rather, we are seeking to structure and parameterize a general model in a realistic manner, using the Georges Bank community as a template. This approach is preferable to constructing an entirely abstract system from scratch. In this paper we describe the model and use it to undertake some preliminary explorations of the utility and sensitivity of alternative multi-species metrics for monitoring the effects of fishing.

2.3.2.2 The model

We seek to model a subset of the Georges Bank fish community for which life history data and time series of abundance are available. Building on previous work by (Murawski and Idoine 1992, Duplisea and Bravington 1999), the model follows fifteen 10 cm wide length cohorts of individuals from each species throughout their life.

We represent growth in the model as a deterministic discredited von Bertalanffy process, parameterised from field data, which is not affected by the abundance of prey resources. Thus, there is no feedback between variations in predation mortality (and hence predator food intake), resulting from time-varying prey abundances, and predator growth.

Recruitment was described by the Ricker stock-recruitment equation parameterised by examining the relationship between α and the parameters of the von Bertalanffy growth equation from metadata (Myers *et al.* 1999). A suitable relationship was found between L_∞ and α which was used to estimate α values from empirical estimates of L_∞ for all species in the model. To parameterise the von Bertalanffy β it was deduced that there should be a linear relationship between the stock size (S) at which recruitment is maximized (S_{r_max}) and β when plotted on logarithmic scales. Using the same database as before (Myers *et al.* 1999) we found an appropriate relationship to model β.

Fishing and natural mortality (M) were entered as instantaneous rates. Mortality was divided into two components, predation mortality (M2) and residual natural mortality (M1). M1 was described as a "U" shaped function with high mortality at small and large sizes to reflect low survival as young and old fish. M2 was modelled very much like MSVPA (Magnusson 1995) where a constant ration, preferences and relative predator/prey abundance being used to calculate suitabilities. Predator-Prey preferences were calculated assuming a log-normally distributed predator/prey size ratio (Hahm and Langton 1984).

To examine the response of alternative descriptors of community structure to fishing pressure we undertook five model runs. All direct input parameters were held constant with the exception of fishing mortality, which was either 0, 0.4 or 0.8. The community indicators we examined were abundance and biomass spectra, species diversity size spectra and *k*-dominance curves.

2.3.2.3   Results

The community behaved as one would expect, with decreasing SSB after 25 years of fishing with constant F for almost all species. This explicable result provides confidence that the model is able to capture a trend that has been empirically and theoretically confirmed many times.

Biomass and abundance size-spectra in this model show a steady decrease over body size (i.e., steepening slope) after 25 years fishing with different levels of constant F (Figure 2.12a,b). In contrast to the size spectra, k-dominance curves show relatively little between communities fished with different constant F regimes (Figure 2.12c,d) except that there is a difference be-tween fished and unfished where dominance of both biomass and abundance by fewer species increases when fished. Though consistent with hypotheses on the effects of fishing, this is not an overly useful pattern as there it does not vary with level of F.

Species diversity size spectra show a clear bi-modality with a trough at about 75cm. This should not be over-interpreted a general ecological pattern as it reflects the parameterisation of L_∞ in our model from the Georges Bank fish community. This pattern would not occur for a community parameterised with a different distribution of L_∞. The diversity size spectrum shows, as might be expected, that diversity of individuals in larger size classes is much more affected by fishing than in smaller size classes, though this two fold response with body size and the separation between fishing regimes is not as marked as with biomass size spectra.

2.3.2.4   Discussion

The present model is essentially a length-based version of MSVPA but parameterised using life history characteristics of individual species. Thus, it is expected that the model is more generic than MSVPA and therefore can be a useful proving ground for assessing the response of various community metrics to changes in F generally.

Evaluating community based metrics in simulation environments is useful to test the robust-ness and ranges of sensitivity of these indicators. Additionally, by using models of different structure as simulation environments for testing the same strategies, one is essentially test the robustness of an indicator to various ecosystem structural mechanisms – or at least our con-ception of them. Thus the simulation environment for Georges Bank can be used as further test of the robustness and sensitivity of indicators examined for the Benguela upwelling fish community (Section 2.3.1). Developing simulation environments and testing the response of various indicators in these different environments should be a focus of future work to aid in the development of operational fish community indicators useful for fisheries and ecosystem-based management.
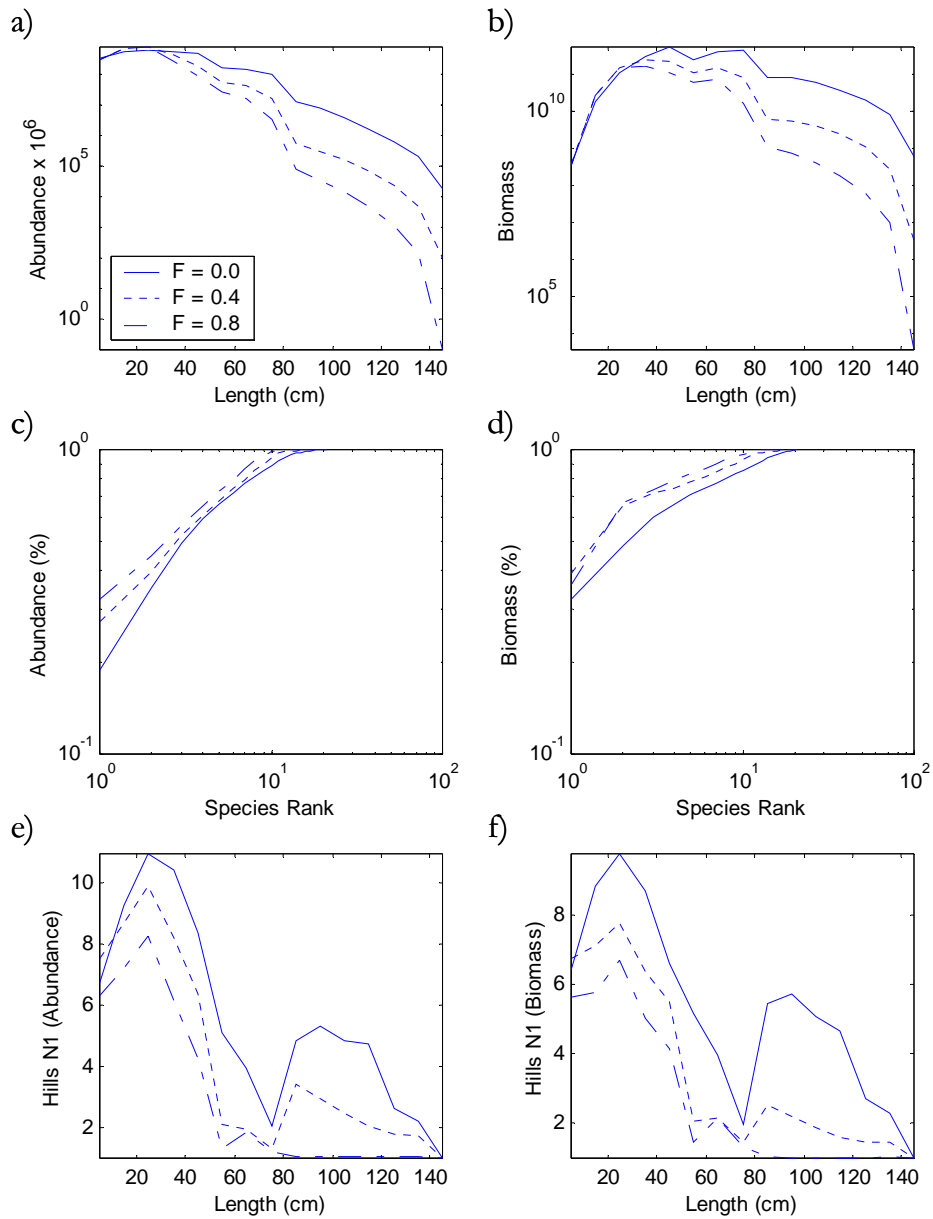
Figure 2.12: The response of various community level attributes to differing fishing mortalities imposed for 25yrs. a & b) abundance and biomass-size spectra: total abundance or biomass of individuals in each size class, c & d) k-dominance curves: log species rank versus cumulative proportional abundance or biomass, e & f) diversity-size spectrum: species diversity, calculated as Hill's N1 (exp Shannon's H), of individuals in each size class.

2.4   Recommendations for TORs in 2006 related to EcoQOs

- **Establishing a protocol/checklist for:**

i.   filtering data: points related to selecting data in size ranges and for species such that we have a consistently catchability and that what is represented by the data can be considered a "system" with internal feedbacks, spatial scales and transient species.

ii. representation of indicators and statistical criteria: standardised plots with relative variation or coefficients of variation that are comparable across systems.

iii. analysis: using the same statistical tests for detecting trends, differences between indicators, methods for detecting correlations between ecological and fishing indicators, and statistical power to detect changes.

This checklist may serve as a rational framework for future investigations and comparative studies. Ideally, the case studies addressed in the WGFE should be revisited using this protocol in order to make comparative approaches possible between different ecosystems but also between different indicators.

- Collecting from other ICES Working Groups temporal series of fishing mortality rates for the main species of the assemblages under consideration in order to provide estimates of multispecies F.

- **What is a large fish** and the implications for evaluating the proportion and the number of large fish. Explore the implications of choosing different size thresholds based on statistical, biological and fishing criteria

- **Towards selecting indicators:**

i.   For systems in which we have sufficient information, develop community F estimates and evaluated changes in indicators in relation to F trends. This will aid in objective empirical evaluation of a suite of indicators to enable us to determine their utility.

ii. **Use simulation tools** to evaluate various ECO QO indicators for sensitivity to F and understand what these indicators are reflecting and their underlying processes and robustness of results across models. These models will also allow us to test methods developed for suggesting pristine ecosystem state and look for convergence of methods over model to suggest what might be the pristine state for a system.

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3   Essential fish habitat

3.1   Introduction

The term essential fish habitat (EFH) has been defined by the US Magnuson-Stevens Fishery Conservation and Management Act (Magnuson-Stevens Act) as "those waters and substrate necessary to fish for spawning, breeding, feeding, or growth to maturity" and this has become the most widely accepted definition of the term. In the Magnusson-Stevens act, "necessary" is taken to mean the habitat required to support a sustainable fishery and the managed species' contribution to a healthy ecosystem, however, in practice, it is often difficult to determine which habitats are strictly necessary (as opposed to preferred) and in many cases, any area presently or historically utilised by a species during a particular life-history stage could be considered to be EFH.

In earlier reports, the concept of Essential Fish Habitat was reviewed (ICES, 2003) and subsequently descriptions of the known fish habitat requirements for a number of commercially important shelf species were provided (ICES 2004a). The descriptions below extend this work by (a) the inclusion of known habitat requirements for a number of deepwater fish species, (b) a description of the little-studied Le Danois Bank in the Cantabrian Sea, (c) a study of habitat preference with respect to environmental variables for selected fish species in the Barents Sea, and (d) mapping biodiversity on the Grand Banks and northeast Newfoundland and Labrador Shelf.

3.2   Deepwater species

Many deepwater species utilise specific depth strata during specific life-history stages and some form aggregations in specific locations for feeding or spawning. Some species have spawning grounds on the continental shelf (e.g., hake *Merluccius merluccius* and tusk *Brosme brosme*). For many species, knowledge of habitat requirements is lacking and for some, whole life history stages are unknown, for example pregnant female leafscale gulper shark *Centrophorus squamosus* are rarely caught, and the location of spawning and nursery grounds for the Rockall Trough population of black scabbard fish *Aphanopus carbo* remain uncertain.

Koslow *et al.* (2000) characterised deep-water fishes as belonging to two distinct functional guilds, "bank and seamount aggregating species" and "slope and open seafloor-associated species". According to Koslow *et al.* (2000) bank and seamount aggregating species form a distinct guild based on common features of their morphology, composition, physiology and metabolism, ecology, and life history. They tend to be robust and deep-bodied, have flesh with low water content, extremely long life-spans and typically do not migrate vertically. Commercial species known to occur in association with seamounts in the Northeast Atlantic include tusk, blue ling *Molva dypterygia*, mora *Mora moro,* cardinal fish *Epigonus telescopus*, orange roughy *Hoplostethus atlanticus*, alfonsinos *Beryx* spp., black scabbard fish *Aphanopus carbo* and roundnose grenadier *Coryphaenoides rupestris*. Of these, only orange roughy and the alfonsinos conform to the stereotype of seamount aggregating species described by Koslow *et al.* (2000) and none of these species is exclusively confined to seamounts, either exhibiting a more general distribution which includes seamounts as well as other habitats or having a dispersed phase alternating with seasonal aggregations.

The 2004 report of the ICES Working Group on the Biology and Assessment of Deep-Sea Fisheries Resources (WGDEEP) summarised existing information on locations of spawning aggregations (ICES 2004b) however much of the existing data was derived form commercial fisheries and consequently could not be included for reasons of commercial sensitivity. Two trans-national projects are currently underway which will contribute to our knowledge and understanding of seamounts in the North East Atlantic: the OASIS project, working on the

Sedlo and Seine Seamounts and the MAR-ECO project on the Mid-Atlantic Ridge. The ecology of seamounts has also been reviewed by Rogers (1994).

Shelf-edge fish assemblages have been studied in the Rockall Trough and Porcupine Bank with regard to pelagic and bentho-pelagic fishes (Gordon and Duncan, 1985; Merrett *et al.*, 1986; 1991; Gordon and Bergstad, 1992; Hulley, 1992) and interactions between them (Mauchline and Gordon, 1983; 1984; 1991).

The role of deepwater coral as essential fish habitat has been addressed by the Study Group on Cold Water Corals, 2004 (ICES 2004c). They concluded that, while several studies have demonstrated increased fish diversity and biomass associated with *Lophelia* reefs, none have so far identified an obligate association of any fish species with cold water corals and that high densities of fishes in particular coral habitats versus adjacent non-coral areas does not necessarily indicate that corals are "important" habitats for fishes in a demographic sense. Nevertheless the role of corals and other complex habitats for fish requires more comprehensive investigations.

The following brief species descriptions summarise the available information known aggregation locations, which may represent important habitat for spawning or feeding.

Orange roughy *Hoplostethus atlanticus* is a benthopelagic species found over steep continental slopes, oceanic ridges and seamounts. It has a widespread distribution, but forms localised aggregations for spawning or feeding. Aggregations of orange roughy have been described on the Olympus Seamount (Subarea X) (ICES 2004b) at several discrete locations on the continental slope west of Ireland (Division VIa and Subarea VII) (ICES 2004b) and some seamounts in the Azores archipelago (Melo and Menezes, 2002; Barceloss *et al*, 2002). French fisheries in the 1990s exploited aggregating orange roughy on the Hebrides Terrace Seamount (Division VIa) but this aggregation is now considered to be depleted. Unpublished French survey data have identified aggregations on the slopes of the Bay of Biscay in Subarea VIII (ICES 2004b). It is very likely that other aggregation sites are known to fishermen but commercial sensitivity means that this information is not readily available to scientists.

Lorance *et al.* (2002) used a manned submersible to observe aggregations around an underwater canyon in the Bay of Biscay. Orange roughy were observed to form dense aggregation of more than 4,000 individuals per ha. close to the bottom of the canyon where they appeared to adopt physiological and behavioural strategies to conserve energy. It was observed that aggregations were probably associated with areas of high water mass movements and mixing and speculated that this may be an adaptation to exploit areas with specific hydrological conditions which offer high prey encounter rates and shelter during metabolic relaxation phases between foraging trips

Roundnose grenadier *Coryphaenoides rupestris* has a general distribution on slopes and flat bottoms. Russian research on the Mid-Atlantic Ridge has found aggregations of this species located at seamount ridges and at near-by slopes at the depths of 1000–1400 m, more often at 1100–1300 m (Gerber *et al.*, 2004). Aggregations were usually in contact with the seabed and varied in height between 30 and 280 m.

Some Russian researchers have concluded that the main spawning area is the Mid-Atlantic Ridge, with pelagic early life stages being transported by oceanic currents to other areas (Vinnichenko and Khlivnoy 2004; Vinnichenko *et al.*, 2004). However, spawning has now been recorded in the Rockall Trough (Kelly *et al.*, 1997), Mid-Atlantic Ridge, Hatton Bank and the Canadian continental slope. Several authors (e.g., Gordon and Bergstad 1992, Kelly *et al,* 1997) have observed an atypical depth distribution of juveniles, with smaller individuals (less than 6 cm pre-anal length) found in the deepest part of the depth range (greater than 1000 m) while larger fish are found at all depths. Gerber *et al*. (2004) reported catches consisting

entirely of juvenile grenadier (12–30 cm total length) at depths of 880–1140 m on seamounts between 51° and 55° N on the Mid-Atlantic Ridge.

Black scabbard fish *Aphanopus carbo* Are known to aggregate on the Ampere, Lion, Seine, and Susan Seamounts near Madeira (Martins and Ferreira, 1995) as well as seamounts around the Azores, the Hatton Bank and Reykjanes ridge. They also have a dispersed distribution in the Rockall Trough and off mainland Portugal. Very few juvenile black scabbard fish have been caught and the eggs and larvae are unknown. Mature fish have been observed at Madeira but not in the Rockall Trough or the Portuguese continental slope and it has been suggested that the seamounts around Madeira may be the sole spawning location for a single northeast Atlantic stock (Swan *et al*., 2003)

Blue ling *Molva dypterygia* is widespread on continental slopes but forms spawning aggregations on seamounts and ridges where they are exploited by commercial fisheries. Magnússon and Magnússon, (1995) reported spawning on a small seamount near the Vestmanna Islands and in a southerly area of the Reykjanes Ridge. Other spawning aggregations have been discovered and exploited by commercial fisheries but their positions are commercially sensitive. A number of known spawning sites on the on the Reykjanes Ridge, at the southern border of the Icelandic EEZ, south of the Vestmanna Isles, in the Storegga area on the continental slopes of the Norwegian Sea and in the northern part of the Rockall Trough were described by ICES (2004b).

Deepwater sharks: Trawl fisheries in the Rockall Trough catch a disproportionately high percentage of pregnant female Portuguese dogfish *Centroscymnus coelolepis* suggesting that this area represents an important habitat for this life-history stage. Conversely, no pregnant females of leafscale gulper shark *C. squamosus* have been observed and the location of maturation and parturition grounds for this species, as well as habitat utilised by pups of both species, remain unknown.

3.3 Le Danois Bank

The Le Danois Bank, a distinct part of the Asturian marginal shelf, covers an area of 350 km^2, and is at a depth of 450–600 m. The north end of the bank has one of the steepest slopes in the world, extending from 400 m to 5000 m depth (i.e., to the abyssal plain) over a distance of about 8 km. The bank is 25 km from the continental shelf, and separated by it from a small basin of 850 m depth. Mediterranean Overflow Water (MOW) passes this bank and so it may be an important feature that structures the marine communities in the area. The Le Danois Bank is also affected by the shelf break current, which is an important source of organic material from the western Cantabrian Sea, where upwelling events are well known.

A research team composed of scientists from IEO, CSIC and CNRS are carrying out a multidisciplinary project on this little-known bank which is now subject to important fishing activities. Two surveys (autumn 2003 and spring 2004) were carried out on board of the RV *Vizconde de Eza* (SGPM). The main goal of the ECOMARG project is the multidisciplinary study of the Le Danois Bank, focusing on the ecosystem and its resources. The presence of biogenic habitats of cold-water corals (*Lophelia*) was described many years ago (Le Danois, 1948) but the current status of these reefs is unknown. The objectives of the project are:

- Morpho-sedimentary and bathymetric study of the bank and surrounding area
- Characteristics and dynamics of water masses
- Benthic and demersal communities description
- Fish and crustaceans trophic ecology
- Fisheries impacts study
- Create a trophodynamic model

- Proposal of sustainable uses of the bank

Sampling using different devices (Figure 3.1) was conducted in the same stations to combine the data from infaunal, epibenthic, suprabenthic invertebrates and demersal fishes (Figure 3.2), based on the systematic sampling and identification of all the taxa, and the combined analysis of quantitative biotic data (abundance and biomass) in relation to abiotic factors of the water column and sediment. This methodological approach can improve the results of multivariate analysis, inter-specific links and biodiversity, including many ecosystem compartments.
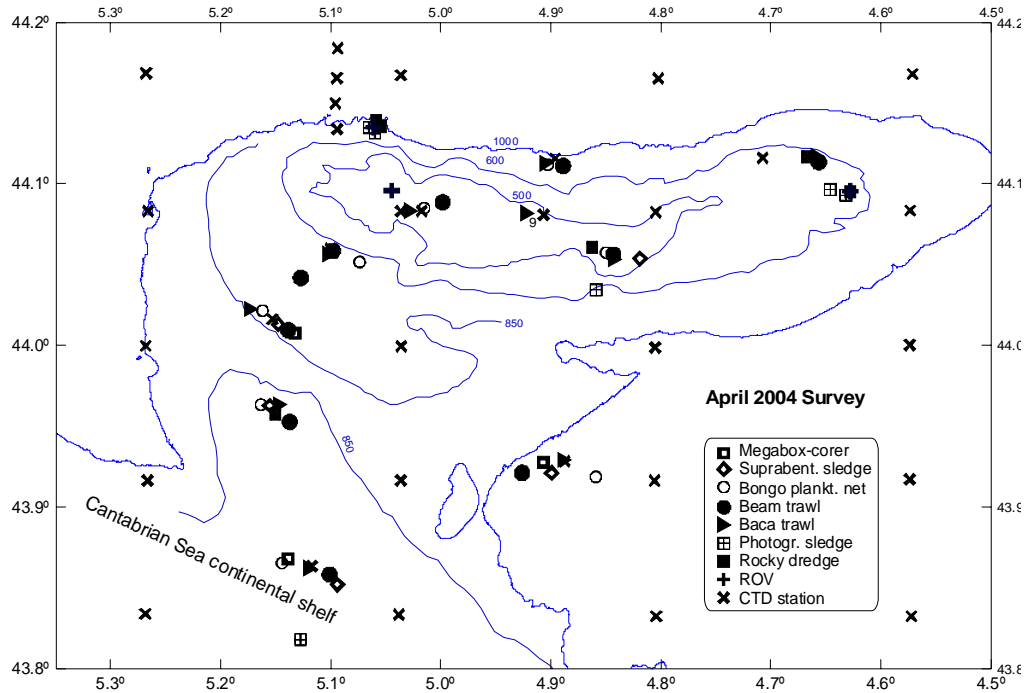
Figure 3.1: Location of different sampling stations during the spring 2004 survey of Le Danois bank.

The trophic ecology of the dominant species of fish and crustaceans will be used to estimate energy flow, and the consumption and the niche overlap among high-level trophic groups. All this information, together with the study of the impact of the fisheries working in the area, will be integrated in a trophodynamic mass-balance model (Ecopath) that allows us to explain and to synthesize the characteristics of the ecosystem and to try to predict the consequences of the possible management measures that can be adopted in this unique habitat. This area is subject to increasing interest to commercial trawlers using rockhopper ground gears.

Preliminary results show that there is a large biomass of suprabenthic fauna, which feed on suspended material, and preliminary field studies have found 23 species new to science and nine species previously considered endemic to the Mediterranean. The overlying sediments of the Le Danois Bank are very thin, with dead coral often recorded, though no live coral has yet been observed in this study. There are many interesting communities on the bank, including other structural fauna, including gorgonians and other soft corals, sponges and hydroids.

In terms of the fish community on the bank, many commercially valuable species occur (Table 3.1) and the adults of several commercial species, including blue whiting, blue-mouth redfish *Helicolenus dactylopterus*, and forkbeard *Phycis blennoides* are often recorded. The adults of these species are uncommon on the Cantabrian shelf, and the increased abundance in this area may be that the Le Danois Bank is an important habitat for these species, as either a spawning or feeding ground. The abundance of adults may also be due to the historically low levels of

fishing effort. These studies will provide a comprehensive account of this unique ecosystem, and help inform any spatial management.
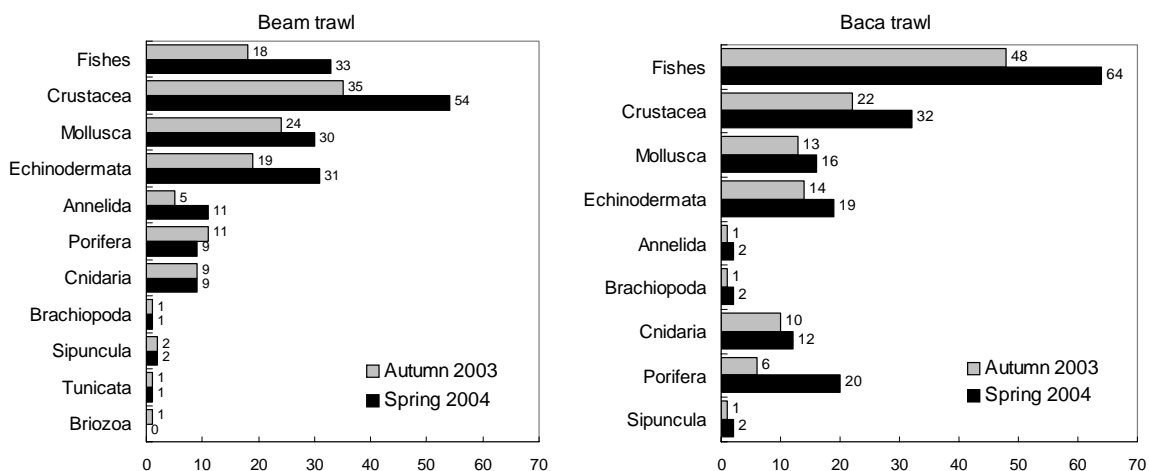
Figure 3.2: Epibenthic invertebrate species (from beam trawl samples) and demersal species (from baca trawl samples) richness (number of species) during the autumn 2003 and spring 2004 surveys on Le Danois bank.

Table 3.1: Biomass (kg/haul) and abundance (number/haul) indices of fish species from bottom trawl survey of autumn 2003 and spring 2004.

AUTUMN 2003

Species	Kg/haul	N./haul	Species	Kg/haul	N./haul
Micromesistius poutassou	129.148	1586.67	*Helicolenus dactylopterus*	0.569	1.50
Chimaera monstrosa	122.271	208.00	*Conger conger*	0.270	0.67
Galeus melastomus	121.643	261.83	*Malacocephalus laevis*	0.265	2.17
Trachyrhynchus scabrus	102.777	284.42	*Centrolophus niger*	0.169	0.25
Etmopterus spinax	22.114	104.92	*Molva dipterygia*	0.097	0.17
Deania calceus	20.950	9.50	*Caelorhynchus caelorhynchus*	0.076	1.33
Trachyscorpia cristulata echi.	12.108	18.92	*Beryx decadactylus*	0.076	0.25
Phycis blennoides	10.860	31.00	*Epigonus telescopus*	0.073	0.50
Lophius piscatorius	7.412	1.58	*Centroscymnus crepidater*	0.062	0.25
Lepidion eques	6.664	85.83	*Dipturus batis*	0.055	0.42
Hydrolagus mirabilis	4.775	8.00	*Halargyreus johnsonii*	0.050	1.75
Coryphaenoides rupestris	4.656	11.00	*Serrivomer beani*	0.044	0.25
Dalatias licha	3.015	1.83	*Rajella fyllae*	0.041	0.33
Aphanopus carbo	2.909	4.25	*Stomias boa*	0.024	0.75
Alepocephalus rostratus	2.689	26.25	*Notacanthus bonapartei*	0.016	0.75
Hoplostethus mediterraneus	2.599	23.58	*Gadiculus argenteus*	0.013	1.17
Mora moro	2.077	24.83	Myctophidae	0.010	0.75
Etmopterus pusillus	1.561	5.00	*Synaphobranchus kaupi*	0.009	2.00
Nezumia sclerorhynchus	1.492	22.75	*Lampanyctus crocodilus*	0.009	0.42
Xenodermichthys copei	1.000	62.00	*Argyropelecus olfersii*	0.008	1.33
Scymnodom ringens	0.872	2.25	*Melanonus zugmayeri*	0.007	0.25
Lepidorhombus boscii	0.787	17.17	*Paralepis coregonoides*	0.003	0.42
Bathysolea profundicola	0.663	11.33	*Paraliparis membranaceus*	0.002	0.50
Leucoraja circularis	0.621	1.00	*Argyropelecus hemigymnus*	0.001	0.92

SPRING 2004	**Kg/haul**	**N./haul**	**Species**	**Kg/haul**	**N./haul**
Galeus melastomus	47.550	98.83	*Centroscymnus crepidater*	0.124	0.13
Trachyrhynchus scabrus	46.833	149.00	*Halargyreus johnsonii*	0.122	1.63
Chimaera monstrosa	44.459	62.29	*Beryx splendens*	0.118	0.25
Deania calceus	20.345	11.50	*Halosaurus ovenii*	0.103	0.50
Alepocephalus rostratus	12.432	25.17	*Merluccius merluccius*	0.102	0.17
Micromesistius poutassou	10.825	97.83	*Molva dipterygia*	0.099	0.25
Alepocephalus bairdii	7.970	48.04	*Hoplostethus atlanticus*	0.093	0.13
Trachyscorpia cristulata ech.	7.608	11.13	*Conger conger*	0.092	0.08
Etmopterus spinax	6.289	76.13	*Melanonus zugmayeri*	0.079	1.38
Phycis blennoides	5.281	13.75	*Coelorhynchus coelorhynchus*	0.073	0.67
Lepidion eques	4.054	49.67	*Nessorhamphus ingolfianus*	0.055	0.50
Lophius piscatorius	3.907	0.92	*Synaphobranchus kaupi*	0.043	2.79
Xenodermichthys copei	3.643	242.71	*Nettastoma melanurum*	0.037	0.08
Coryphaenoides rupestris	3.548	7.50	*Malacocephalus laevis*	0.028	0.38
Scymnodom ringens	3.304	2.04	*Raja clavata*	0.016	0.25
Centroscymnus coelolepis	2.663	0.25	*Argyropelecus olfersii*	0.013	1.92
Mora moro	2.373	7.17	*Lampanyctus* sp	0.013	0.88
Chlorophthalmus agassizii	1.609	22.25	*Gadiculus argenteus*	0.010	1.00
Hoplostethus mediterraneus	1.489	8.71	*Bathypterois dubius*	0.009	0.63
Hydrolagus mirabilis	1.392	3.58	*Leptoderma macrops*	0.007	1.63
Nezumia sclerorhynchus	1.245	20.00	*Trigla lyra*	0.006	0.17
Aphanopus carbo	1.069	1.46	*Neoscopelus macrolepidotus*	0.006	0.13
Leucoraja circularis	1.046	0.67	*Bathytroctes microlepis*	0.005	0.25
Lepidorhombus boscii	0.700	15.00	*Melagnostigma atlanticum*	0.003	1.63
Dipturus batis	0.619	0.25	*Chauliodus sloani*	0.003	0.25
Bathysolea profundicola	0.355	7.33	*Diaphus* sp.	0.003	0.21
Dalatias licha	0.340	0.13	*Hymenocephalus italicus*	0.001	0.25
Helicolenus dactylopterus	0.249	0.75	*Borostomias antarcticus*	0.001	0.13
Beryx decadactylus	0.220	0.83	*Lampanyctus crocodilus*	0.001	0.08
Notacanthus bonapartei	0.144	1.88	*Argyropelecus hemigymnus*	0.000	0.21
Etmopterus pusillus	0.143	0.50	*Stomias boa*	0.000	0.13
Epigonus telescopus	0.131	0.25	*Ceratoscopelus maderensis*	0.000	0.08

3.4 Barents Sea

The essential fish habitat conception assumes a detailed knowledge about the habitat conditions for each species and their life-history stages. For the evaluation of possible distribution range for each fish species the data on habitat preferences are necessary. The habitat preferences (in terms of depth, temperature and salinity) of selected fish species (Table 3.2) in the Barents Sea were analysed (Dolgov and Karsakov, In prep.). Data from the research surveys conducted by PINRO in the Barents Sea in 1997–2004 were used. When conducting the trawl station, bottom temperature and water salinity were measured by CTD-unit. Later on, using temperature and salinity, as well as geographical position data, the trawl stations were related to these or those water masses applying the methods proposed by Ozhigin and Ivshin (1999).

Table 3.2: Number of stations which data on depth, temperature and salinity were available for the various fish species of the Barents Sea.

Fish species	Depth	Salinity	Temperature
Anarhichas denticulatus	695	459	482
Anarhichas lupus	971	664	673
Anarhichas minor	757	520	535
Arctozenus rissoi	161	127	127
Argentina silus	73	57	57
Artediellus atlanticus	594	444	444
Bathyraja spinicauda	34	22	22
Bentosema glacialis	16	16	16
Boreogadus saida	1048	720	743
Brosme brosme	70	57	58
Careproctus reinhardti	593	441	442
Chimaera monstrosa	3	3	3
Clupea harengus	598	430	436
Clupea harengus marisalbi	7	6	6
Clupea pallasi suworowi	34	5	13
Cottunculus microps	378	220	223
Cottunculus sadko	73	55	55
Cyclopterus lumpus	1384	912	961
Enchelyopus cimbrius	14	12	12
Eumicrotremus derjugini	9	9	9
Eumicrotremus spinosus	216	150	151
Gadiculus argenteus thori	18	17	17
Gadus morhua	2405	1586	1654
Gaidropsarus argentatus	60	22	24
Gasterosteus aculeatus	32	17	17
Glyptocephalus cynoglossus	43	30	30
Gymnacantus tricuspis	38	11	11
Hippoglossoides platessoides	2424	1595	1661
Hippoglossus hippoglossus	10	10	10
Icelus bicornis	36	27	27
Icelus spatula	18	13	13
Lepidorhombus whiffiagonis	3	3	3
Leptagonus decagonus	381	256	257
Leptoclinus maclatus	136	106	106
Limanda limanda	46	27	29
Liparis fabricii	56	47	47
Liparis gibbus	145	96	96
Liparis tunicatus	13	4	4
Lumpenus fabricii	69	29	29
Lumpenus lampretaeformis	49	37	37
Lycodes esmarki	345	190	192
Lycodes eudipleurostictus	74	46	46
Lycodes pallidus	58	30	30
Lycodes polaris	24	7	7
Lycodes reticulates	134	98	98
Lycodes rossi	83	43	43
Lycodes seminudis	148	106	106
Lycodes vahli gracilis	417	328	328
Macrourus berglax	236	73	73
Mallotus villosus	871	648	653
Maurolicus muelleri	5	4	4

Fish species	Depth	Salinity	Temperature
Melanogrammus aeglefinus	1699	1168	1216
Merlangius merlangus	4	4	4
Micromesistius poutassu	1091	734	745
Microstomus kitt	10	8	8
Myoxocephalus scorpius	82	40	41
Pleuronectes platessa	307	177	191
Pollachius virens	210	159	166
Raja (Dipturus) batis	46	27	28
Raja (Rajella) fyllae	130	92	92
Raja (Amblyraja) hyperborean	170	81	88
Raja (Dipturus) lintea	9	6	6
Raja (Amblyraja) radiata	1907	1265	1305
Reinhardtius hippoglossoides	1312	853	865
Sebastes marinus	1077	747	777
Sebastes mentella	1099	723	745
Sebastes viviparous	159	128	130
Somniosus microcephalus	7	6	6
Squalus acanthias	3	2	3
Triglops murrayi	428	334	334
Triglops nybelini	120	83	83
Triglops pingelii	48	34	34
Trisopterus esmarki	447	345	352

It is possible to separate three groups of species by habitat depth (Figure 3.3). Many species (e.g., cod *Gadus morhua*, long-rough dab *Hippoglossoides platessoides*, thorny skate *Amblyraja radiata*) occurred over a wide range of depths. Shallow-water species (e.g., shorthorn sculpin *Myoxocephalus scorpius*, Arctic staghorn sculpin *Gymnacanthus tricuspis*, dab *Limanda limanda*, lumpsucker *Eumicrotremus derjugini*) occurred mainly in shallower waters of up to 80–100 m. Deep-water species (e.g., Roughhead grenadier *Macrourus berglax*, Arctic rockling *Gaidropsarus argentatus*, doubleline eelpout *Lycodes eudipleurostictus*, and blue skate *Dipturus batis*) usually occurred in depths of up to 700–800 m.

While analyzing the mean temperature, and its range, of the different fish species in their habitat it was observed that a wide temperature range was peculiar to the majority of fishes, ranging from $1-2^0$C for witch *Glyptocephalus cynoglossus* and Atlantic herring *Clupea harengus,* to 10^0C for cod, long rough dab and thorny skate (Figure 3.4). At the same time, the difference in mean habitat temperature weighted by catch was more pronounced. The species most associated with cold-water included leatherfin lumpsucker *Eumicrotremus spinosus* and gelatinous snailfish *Liparis fabricii*, which occurred generally at temperatures < 0ºC. Species occurring at the highest temperatures in this area were lemon sole *Microstomus kitt,* greater argentine *Argentina silus*, Norway redfish *Sebastes vivaprous* and Norway pout *Trisopterus esmarki.*

The same was observed when analyzing salinity, under which different fish species occurred (Figure 3.5). Dab, plaice *Pleuronectes platessa* and shorthorn sculpin occurred in the waters with a lower salinity (< 34.5%). Rabbitfish *Chimaera monstrosa*, Norway haddock, greater argentine, lemon sole occurred in waters with high salinity (> 35 $^0/_{00}$).
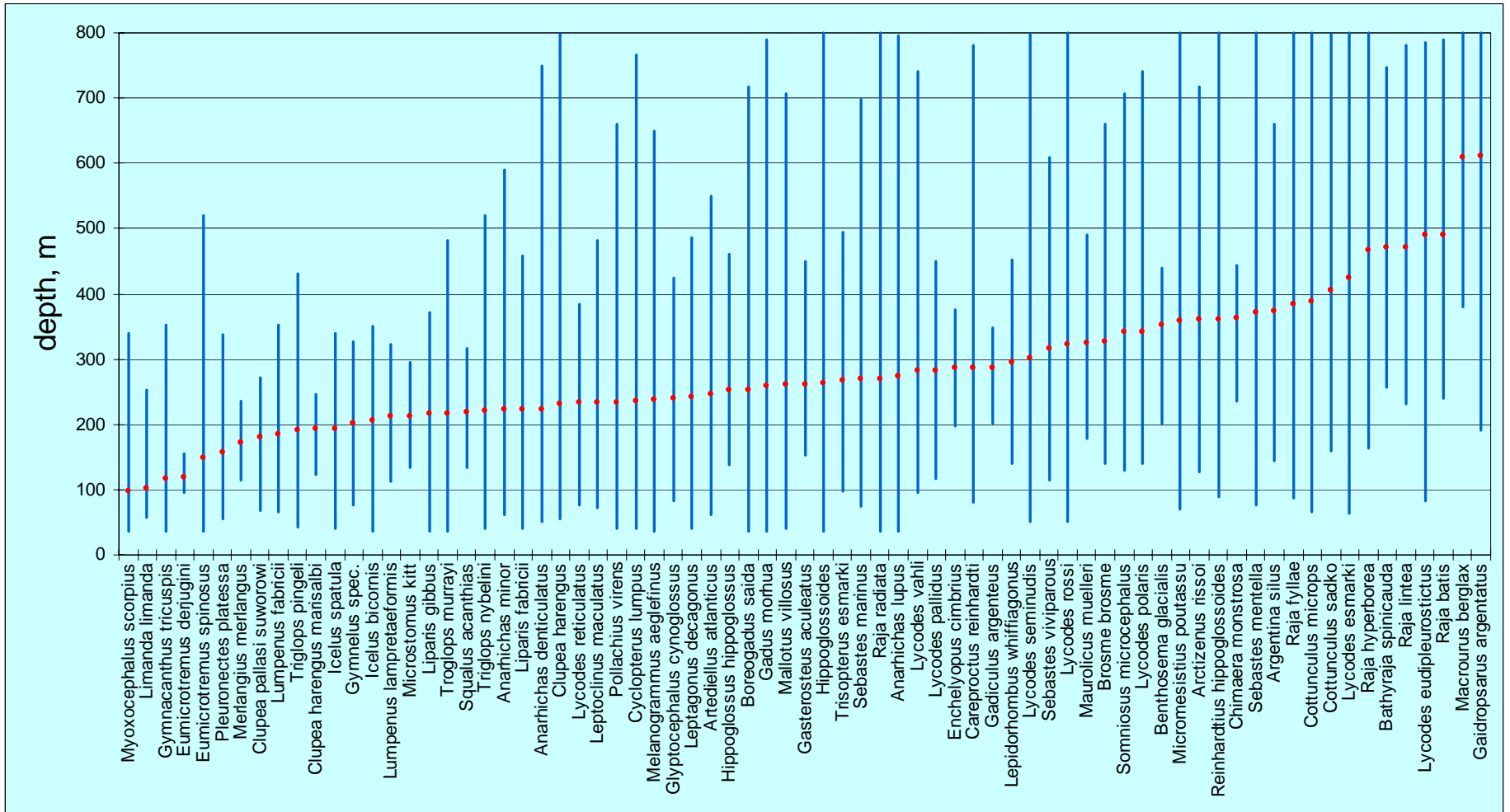
Figure 3.3: Mean depth and depth range of some fish species from the Barents Sea.
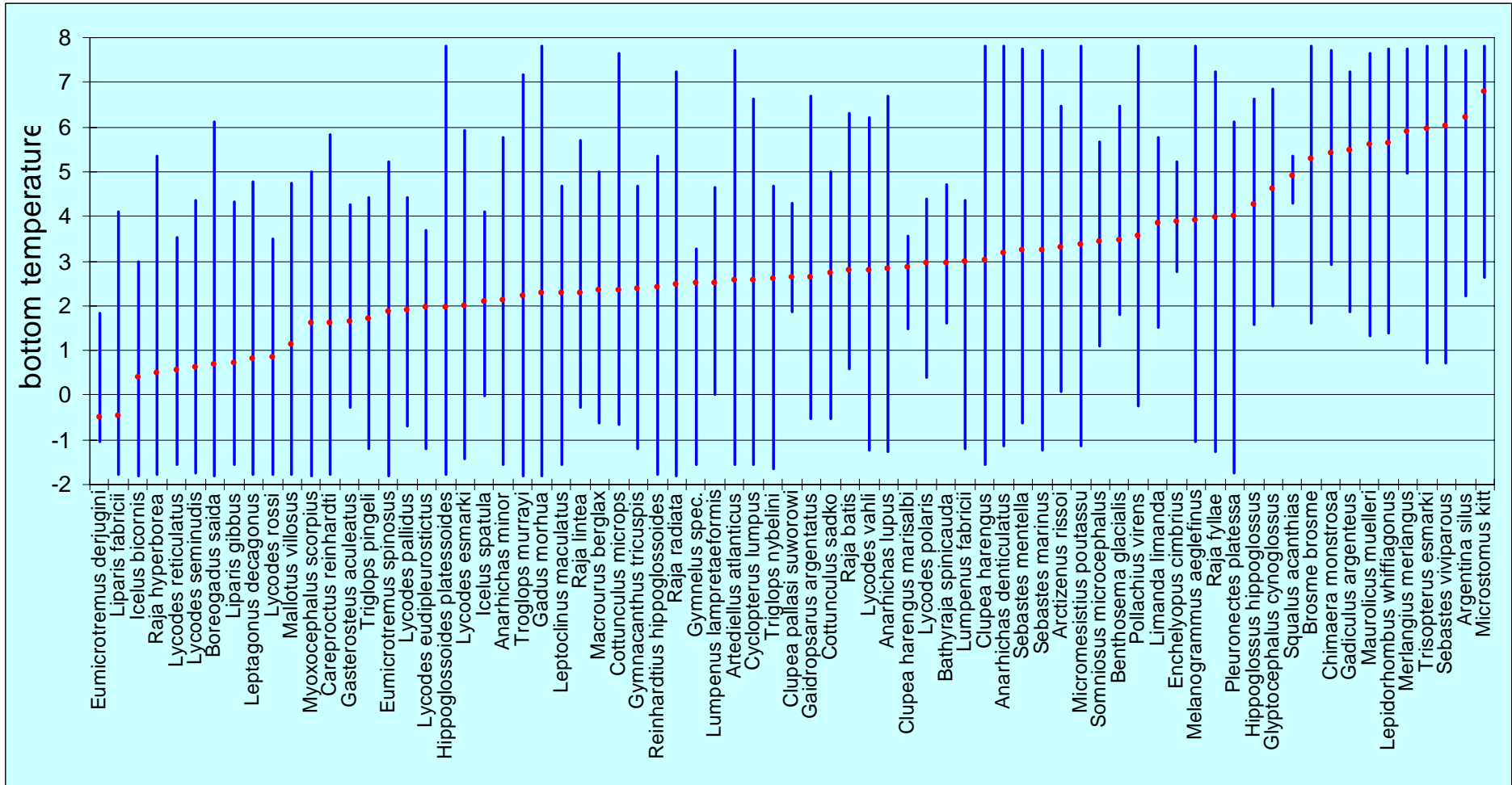
Figure 3.4: Mean bottom temperature and temperature range of some fish species from the Barents Sea.
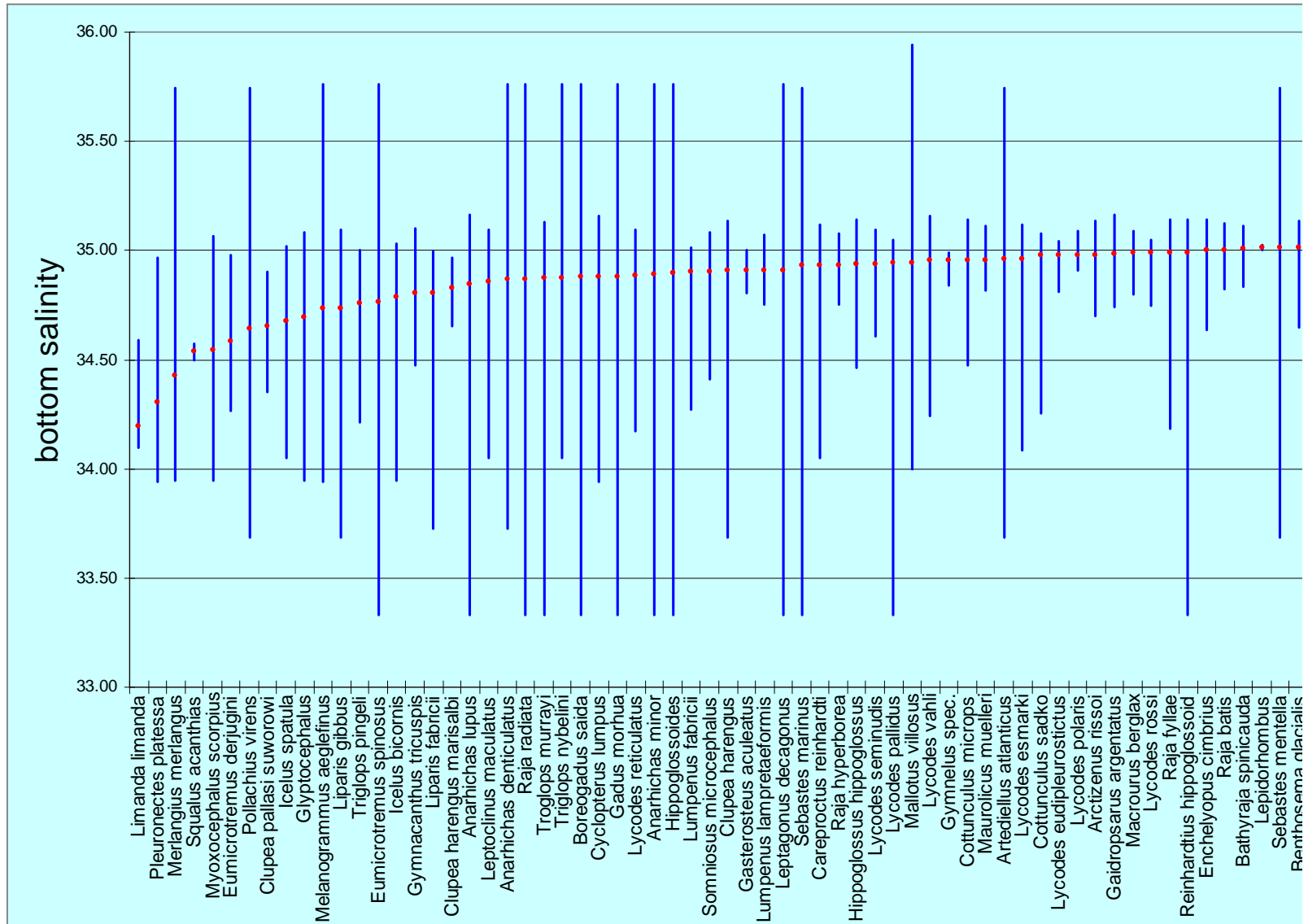
Figure 3.5: Mean bottom salinity and salinity range of some fish species from the Barents Sea.

The range of habitat characteristics utilized by the fish species were quite variable and the range of habitats that may be utilised obviously affects the overall distribution. Two examples of different distribution patterns are shown in Figure 3.6 for leatherfin lumpsucker (shallow cold waters with medium salinity) and thorny skate (wide range of depth, temperature and salinity). The widest distributions were observed for those species with the widest range of depth, temperature and salinity (cod, long rough dab and thorny skate), reiterating that habitat preferences can be an important element in explaining abundance-occupancy relationships.
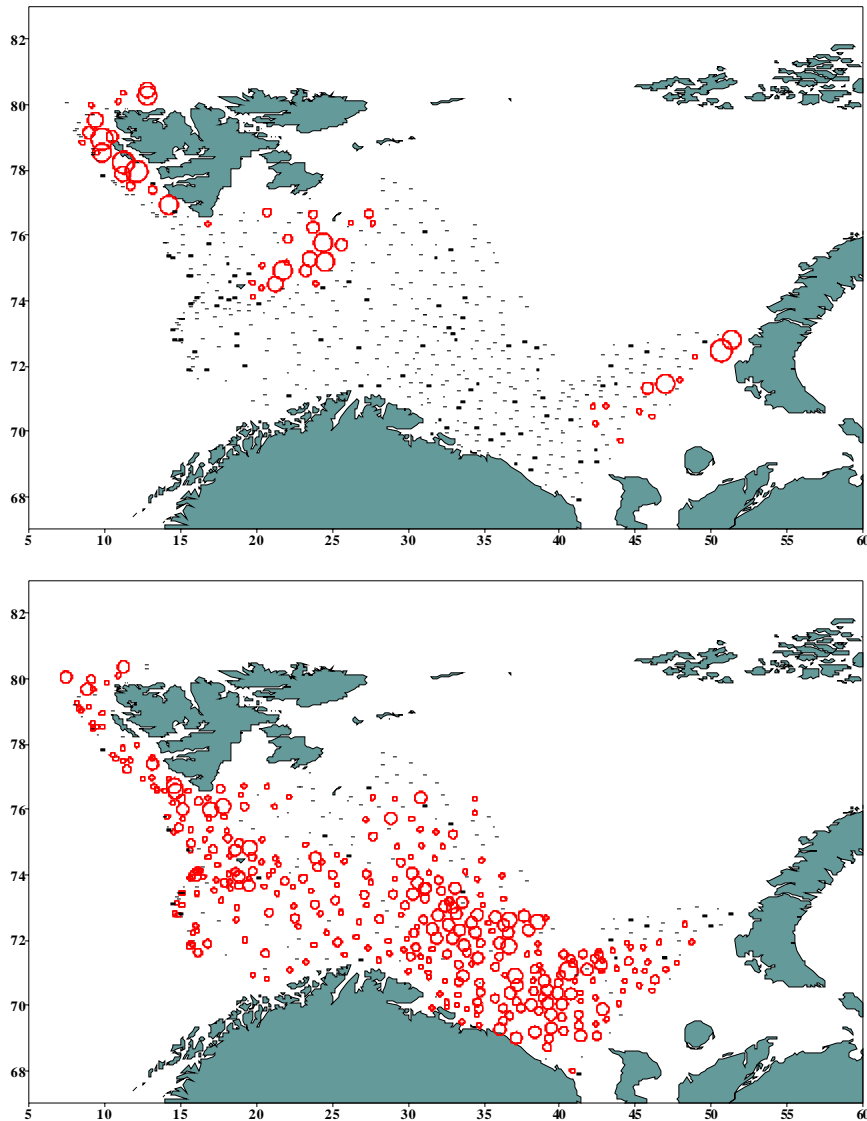
Figure 3.6: Distribution and relative abundance of leatherfin lumpsucker *Eumicrotremus spinosus* (top) and thorny skate *Amblyraja radiata* (bottom), spec. per 1 hour haul.

Preliminary studies to consider the occurrence of different fish species in certain water masses of the Barents Sea (Figure 3.7) using data collected in 2002 was made. Major differences in the ichthyofaunal species composition in different water masses were noticed (Figure 3.8). The number of species was maximal in the Atlantic waters (64 species), with 24–33 fish species occurring in the other water masses. Boreal species predominated in all the water masses. Southern boreal and widely distributed species only occurred in the Atlantic and coastal Norwegian and Murman waters.
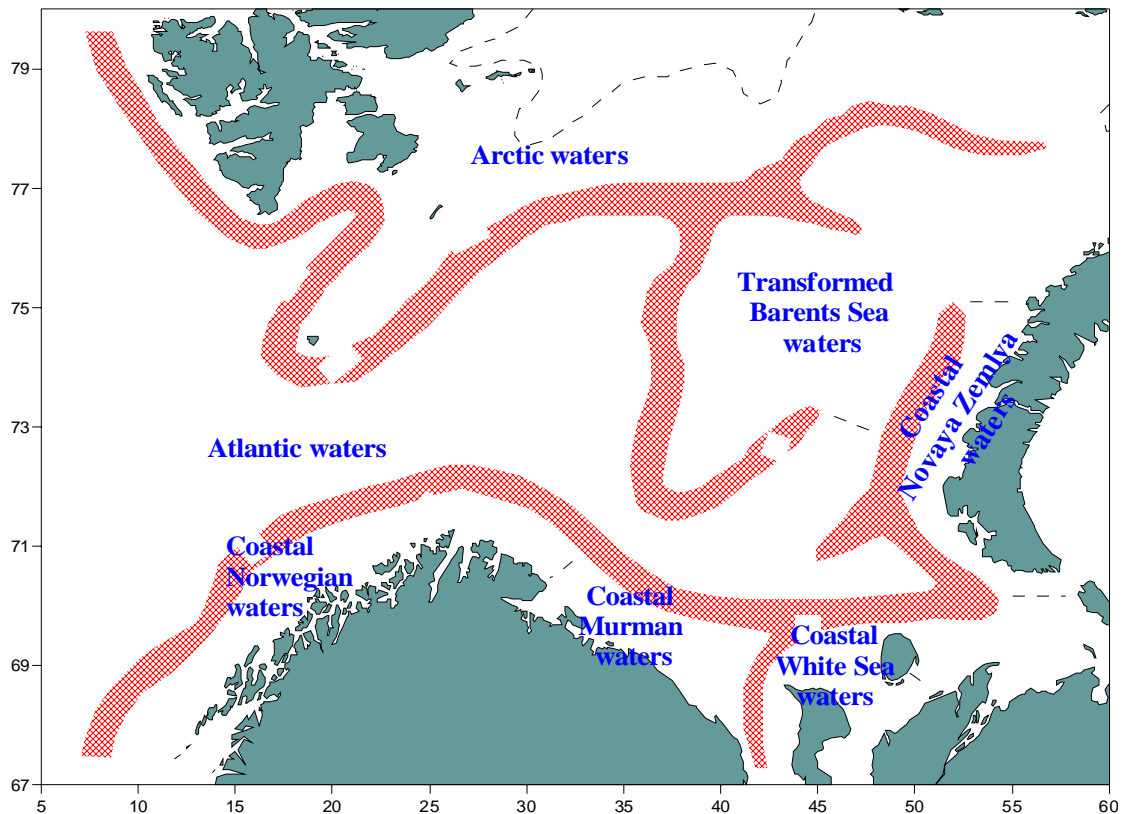
Figure 3.7: Water masses in the Barents Sea (according to Ozhigin and Ivshin, 1999).

The species with the widest range of temperature and salinity occurred in all water masses. Simultaneously, some fish species dwelled in certain water masses. So, blue catfish *Anarhichas denticulatus* and Atlantic spiny lumpsucker *Eumicrotremus spinosus* were typical of Arctic waters; Arctic skate *Amblyraja hyperbnorea*, Longear eelpout *Lycodes seminudus*, sea tadpole *Careproctus reinhardti* and Atlantic poacher *Leptagonus decagonus* rarely occurred outside of the transformed Barents Sea waters. Plaice and dab were related to coastal White Sea and Murman waters, and Norway redfish, greater argentine, Norway pout to the Atlantic and coastal Norwegian and Murman waters.
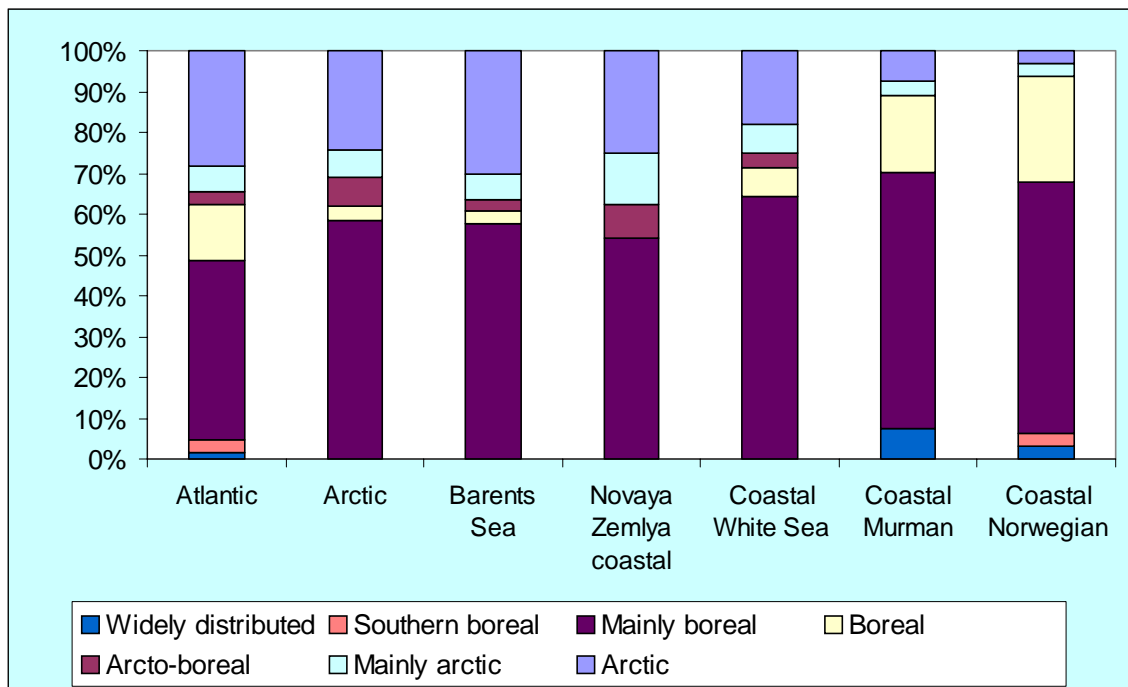
Figure3.8: The proportions of the fish species from the different zoogeographic groups in the different water masses of the Barents Sea.

Variations in water temperature lead to increased distribution areas for some fish species and narrower distributions for other species. It can cause a change in the spatial structure of fish communities and lead to changes in the proportion of the different zoogeographical groupings in catches. It is pronounced best in the boundaries of the Barents Sea - the area between Spits-bergen and the Franz Joseph Land, and the southern Barents Sea. So, between 2000–2003, in the northern Barents Sea, there has been a quite abrupt reduction in the proportion of Arctic species, and the percentage of mainly boreal and boreal species has increased (Figure 3.9). In the same, in the southern Barents Sea, a similar decrease in the percentage of Arctic and arcto-boreal species and an increase in the proportion of mainly boreal species (Figure 3.10) was also observed. Besides, the reduction and disappearance of southern boreal species, the change of the proportion of fish from different zoogeographical groupings was closely connected with water temperature in different areas of the Barents Sea. Temperature rises in Sections 29 and 31 (the northern branch of the North Cape Current and the eastern branch of the Norwegian Current) conditioned the increase in the proportion of boreal and mainly boreal and widely distributed species and, correspondingly, a reduction in the proportion of mainly Arctic and arcto-boreal species (Tables 3.3 and 3.4).
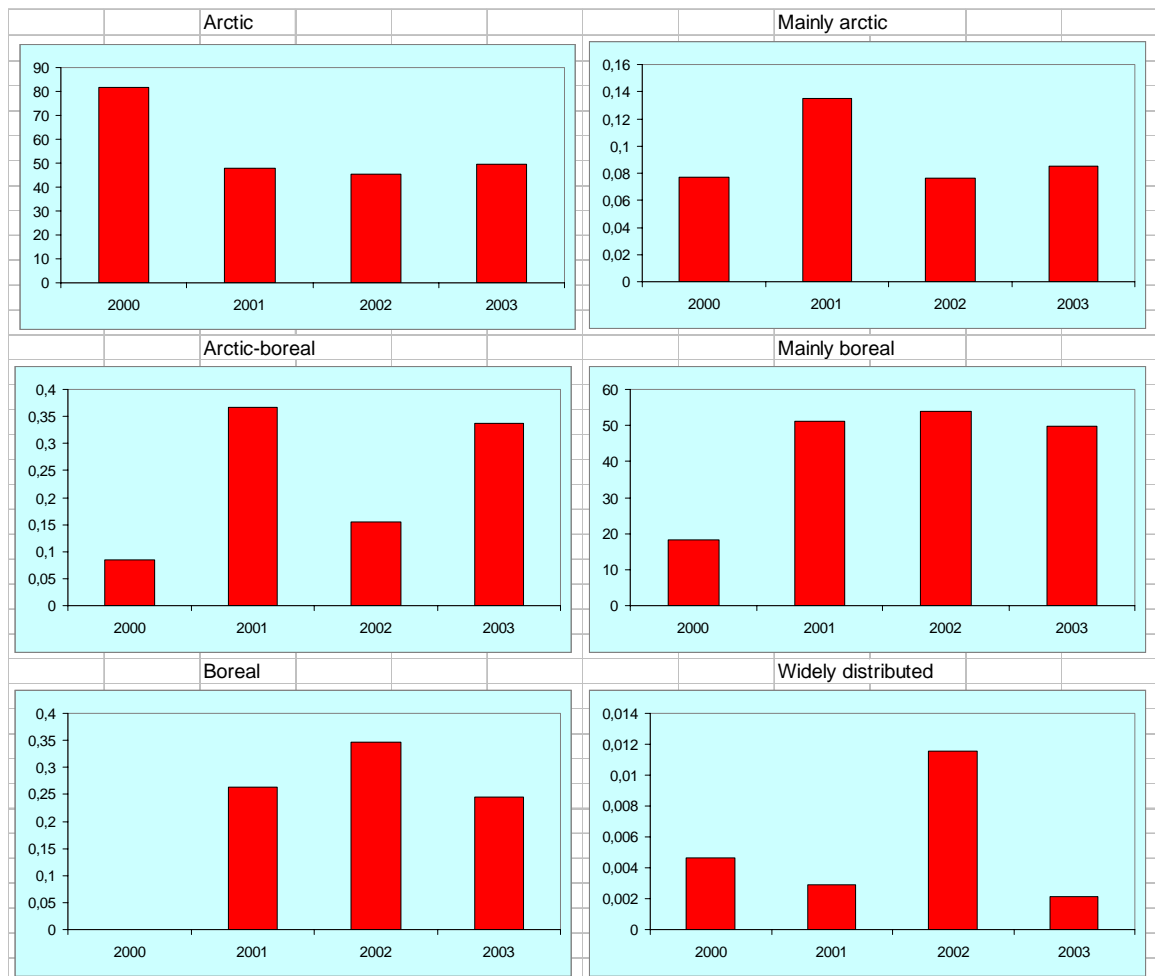
Figure 3.9: Changes in the proportion of fish species from different zoogeographic groups in the northern part of the Barents Sea (Spitsbergen and Franz Josef Land archipelagos).
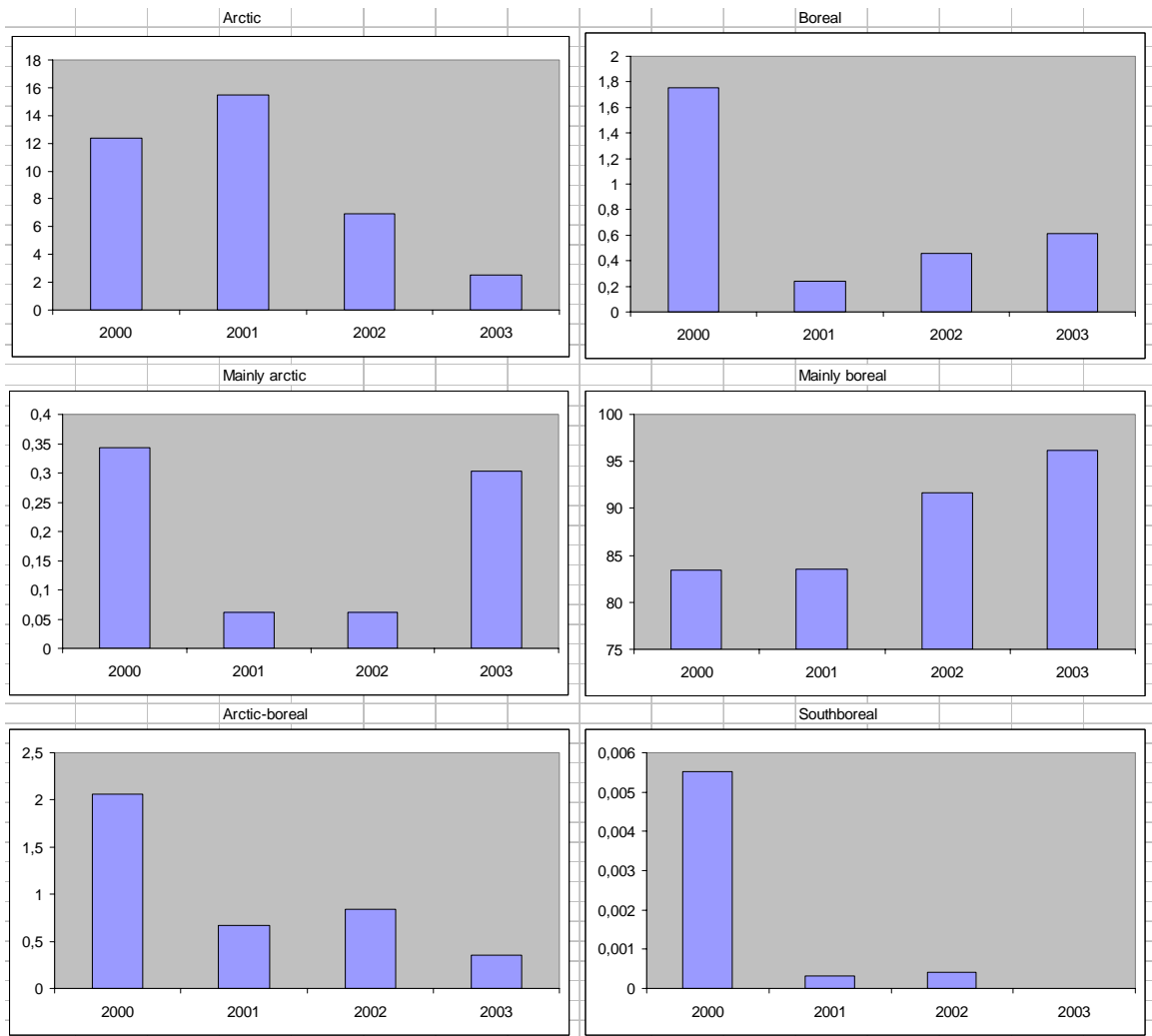
Figure 3.10: Changes in the proportion of fish species from different zoogeographic groups in the southern part of the Barents Sea.

Table 3.3: Pearson r values of relationships between the proportion of the catches of fish species from the different zoogeographic groups in Spitsbergen – Franz Josef Land area and water temperature in oceanographic sections

ZOOGEOGRAPHIC GROUPS	KOLA SECTION (STATION 3–7) (THE MAIN BRANCH OF THE MURMAN CURRENT)	KOLA SECTION (STATION 8–10) (THE CENTRAL BRANCH OF THE NORTH CAPE CURRENT)	SECTION 29 (THE NORTHERN BRANCH OF THE NORTH CAPE CURRENT)	SECTION 31 (THE EASTERN BRANCH OF THE NORWEGIAN CURRENT)	SECTION FUGLØYA-BJORNØYA (THE MAIN BRANCH OF THE NORTH CAPE CURRENT)
Arctic	**0.75**	-0.01	0.08	0.15	**-0.99**
Mainly arctic	**-0.89**	**0.72**	**-0.71**	**-0.78**	0.41
Arctic-boreal	**-0.83**	0.06	**-0.79**	**-0.82**	**0.74**
Boreal	**-0.66**	0.02	0.12	0.05	**0.95**
Mainly boreal	**-0.75**	0.01	-0.08	-0.14	**0.99**
Widely distributed	0.20	0.08	**0.94**	**0.90**	0.09

Table 3.4: Pearson r values of relationships between the proportions of the catches of fish species from the different zoogeographic groups in the southern Barents Sea and water temperature in oceanographic sections

ZOOGEOGRAPHIC GROUPS	KOLA SECTION (STATION 3–7) (THE MAIN BRANCH OF THE MURMAN CURRENT)	KOLA SECTION (STATION 8–10) (THE CENTRAL BRANCH OF THE NORTH CAPE CURRENT)	SECTION 29 (THE NORTHERN BRANCH OF THE NORTH CAPE CURRENT)	SECTION 31 (THE EASTERN BRANCH OF THE NORWEGIAN CURRENT)	SECTION FUGLØYA-BJORNØYA (THE MAIN BRANCH OF THE NORTH CAPE CURRENT)
Arctic	-0.30	**0.93**	-0.22	-0.28	-0.34
Mainly arctic	**0.75**	**-0.63**	-0.18	-0.06	**-0.67**
Arctic-boreal	**0.70**	0.20	0.34	0.37	**-0.96**
Boreal	**0.88**	-0.19	0.21	0.29	**-0.98**
Mainly boreal	0.07	**-0.84**	0.14	0.18	0.54
Southern boreal	**0.75**	0.08	0.20	0.26	-1.00
Widely distributed	**-0.89**	0.11	**-0.66**	**-0.71**	**0.85**

3.5 Biodiversity mapping

Preliminary information on broad species distributions and spatial aspects of biodiversity for the Grand Banks and northeast Newfoundland and Labrador Shelf (Canadian Atlantic) was used to describe spatial changes, area occupied and habitat associations. The biodiversity mapping was done to examine spatial diversity in relation to habitat and the primary purpose of mapping the individual species is to explore spatial changes in species that may be at risk of extinction. The eventual product will be an atlas similar to that produced for northern Spanish waters (Sanchez et. al 2002). The spatial and temporal patterns in fishing activity in this area were mapped in previous studies (Kulka and Pitcher, 2001).
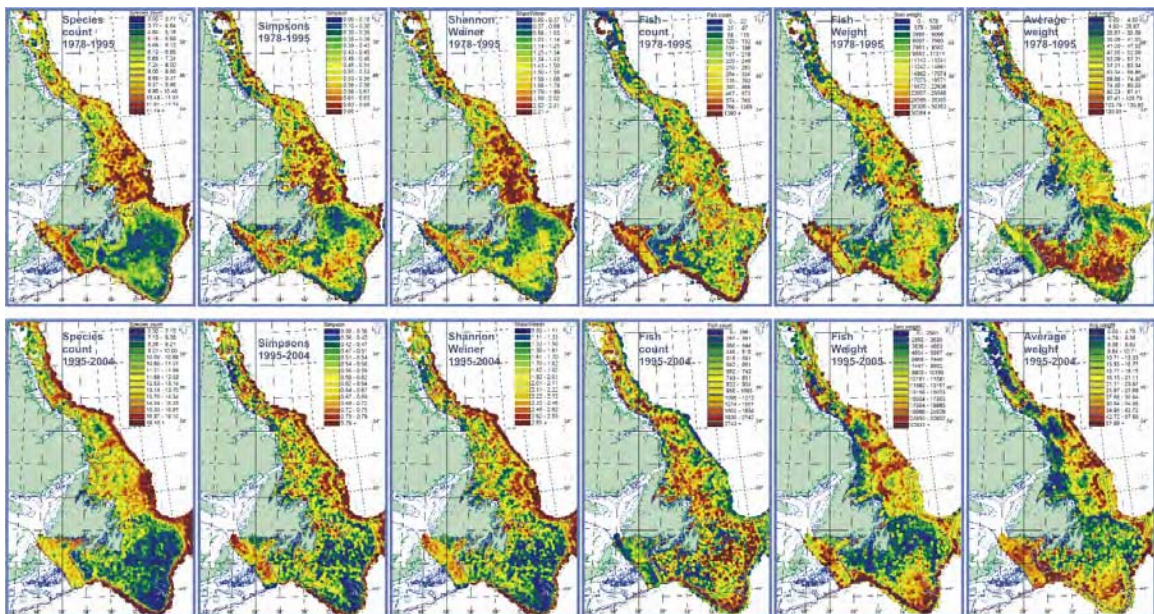
Figure 3.11: Different measures of biodiversity for marine fish taken in research trawl surveys (1978–1995 and 1995–2004) off Newfoundland and Labrador from left to right: Species count, Simpson, Shannon-Weiner, total counts of individuals (showing distribution of total abundance), total weight (showing distribution of total weight) and average weight (total weight/total count of individuals).

The data used for these studies comprise spring and fall Fisheries & Oceans, Newfoundland Region demersal research trawl surveys from 1978 to 2004 covering the Grand Banks, Northeast Newfoundland and Labrador Shelf. About 400 demersal fish species during the surveys, of which about 80% were relatively rare. The mapping methodology was described in Kulka (1998). Mapped species richness (species count) was compared to two equitability (evenness)-richness indices, Simpson and Shannon-Weiner for the entire time period, showing a spatially high degree of similarity between the equitability indices with moderate similarity to species count (Figure 3.11). Also mapped were total fish count, total weight of fish and average weight of fish, all of which showed very different spatial patterns from the diversity indices. For example, average size of fish was greatest on the southern Grand Banks. Relationships among these spatial patterns will be explored over time in relation to a possible ecological shift.

The three biodiversity indices: species count; Simpson and Shannon-Weiner were, as expected, highly correlated with bottom temperature – higher temperature-higher biodiversity (Figure 3.12). Future research will examine changes in the distribution of biodiversity (richness and equitability) over time in relation to bottom temperature and depth.
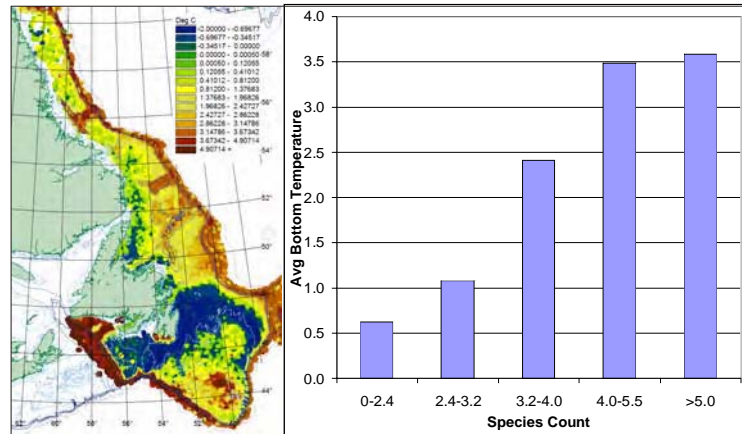
Figure 3.12: Bottom temperature (left panel) in relation to species count (right panel).

Examples of distribution of individual species were also presented (Figures 3.13–3.14) to illustrate different associations among species with respect to depth and bottom temperature (as an available measures of habitat). Bottom temperatures within the study area generally range between -2ºC and 6ºC. Examples of species associated with 'warm-water' (average associated bottom temperature >3ºC) *vs.* cold-water (< ~ 1ºC) conditions and deep (average depth > 700m) *vs.* shallow (< 200m) were shown.

Based on 30 species examined to date, several have been found to have highly overlapping distributions such as saithe *Pollachius virens* and haddock *Melanogrammus aeglefinus* (warm), spatulate sculpin *Icelus spatula*, northern alligatorfish *Leptagonus decagonus* and moutsache sculpin *Triglops murrayi* (cold), Longhorn sculpin *Myoxocephalus octodecemspinosus* and yellowtail flounder *Limanda ferruginea* (shallow) and roundnose grenadier *Coryphaenoides rupestris* and black dogfish *Centroscyllium fabricii* (deep).
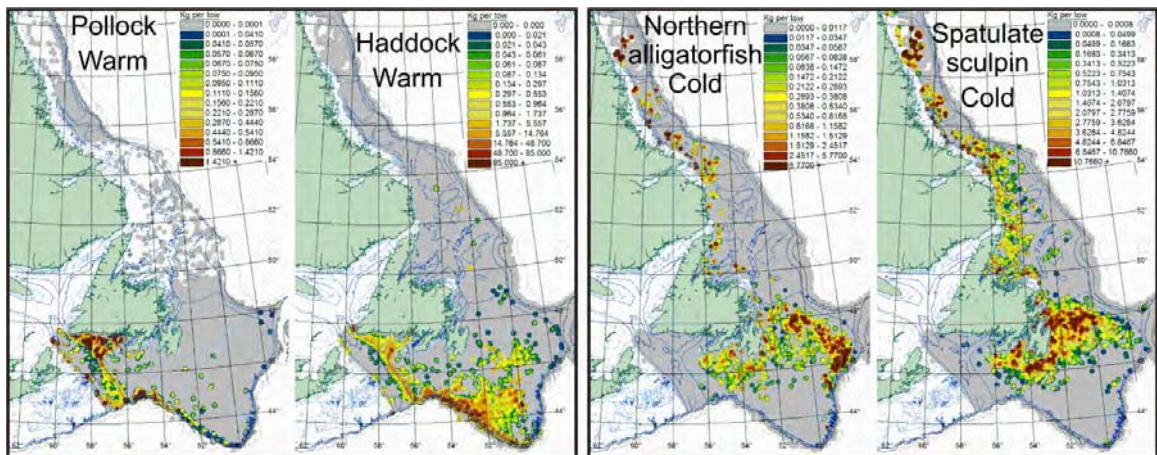
Figure 3.13: Examples of closely associated species in relation to bottom temperature.
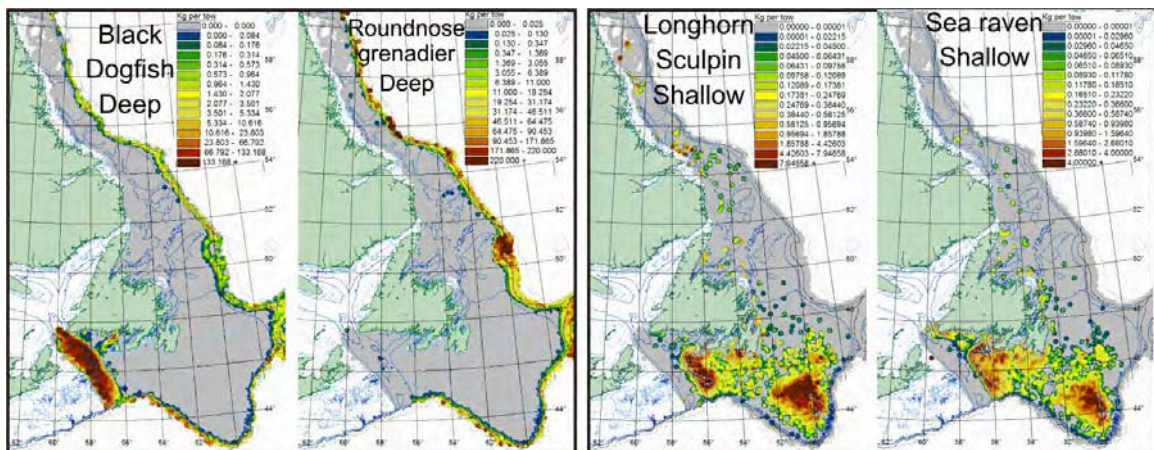
Figure 3.14: Examples of closely associated species in relation to depth.

Other species were observed to have very high spatial association (e.g., smooth skate *Malacoraja senta* and four-bearded rockling *Enchelyopus cimbrius*) comprising several well separated aggregations that appeared to bear little relationship to either depth or bottom temperature. In some cases, the species were taxonomically related. The changes in abundance over time will be examined for a sample of 'warm-water' *vs.* cold-water species. Preliminary analyses suggest very different trajectories during an apparent "ecological shift" that occurred during the 1980's and early 1990's. Also apparent from these preliminary analyses was a fairly consistent pattern of smaller average size in the cold-water species. A time series of matrices of temperature and depth association by size will illustrate these relationships across all species examined.

A more detailed analysis of wolffish (*Anarhichas lupus, A. minor* and *A. denticulatus*) was used to illustrate habitat associations. Habitat associations and critical habitat for wolffish and marine fish in general has not been well defined. Determining what is critical to the survival of a species in an enclosed or coastal environment is far less complicated than species that inhabit vast expanses of the open ocean because of spatial scale and less detailed knowledge of the offshore environment. Three factors impede the definition of critical habitat in the open ocean in general and for wolffish in particular. Firstly, insufficient knowledge of wolffish life history; secondly, limited information on the influence of multi-scale processes upon wolffish population dynamics; and thirdly the lack of information on acceptable targets for wolffish population abundance and range. Consequently it is difficult to define critical habitats for wolffish, particularly since each developmental stage may have different requirements which are at present unknown.

Temporal trends in distribution indicated that all three wolffish species occurring in the northwest Atlantic underwent significant changes in their distribution (Figure 3.15). These changes were concurrent within changes in abundance (Kulka *et al.*, 2004). Formerly occurring in high abundance across the entire Northeast Newfoundland and Labrador Shelf, the remaining populations are mainly concentrated on the outer shelf. Figure 3.16 illustrates the changes in area occupied and in the degree of concentration. Only a small portion of the shelf now contains high densities of these species.
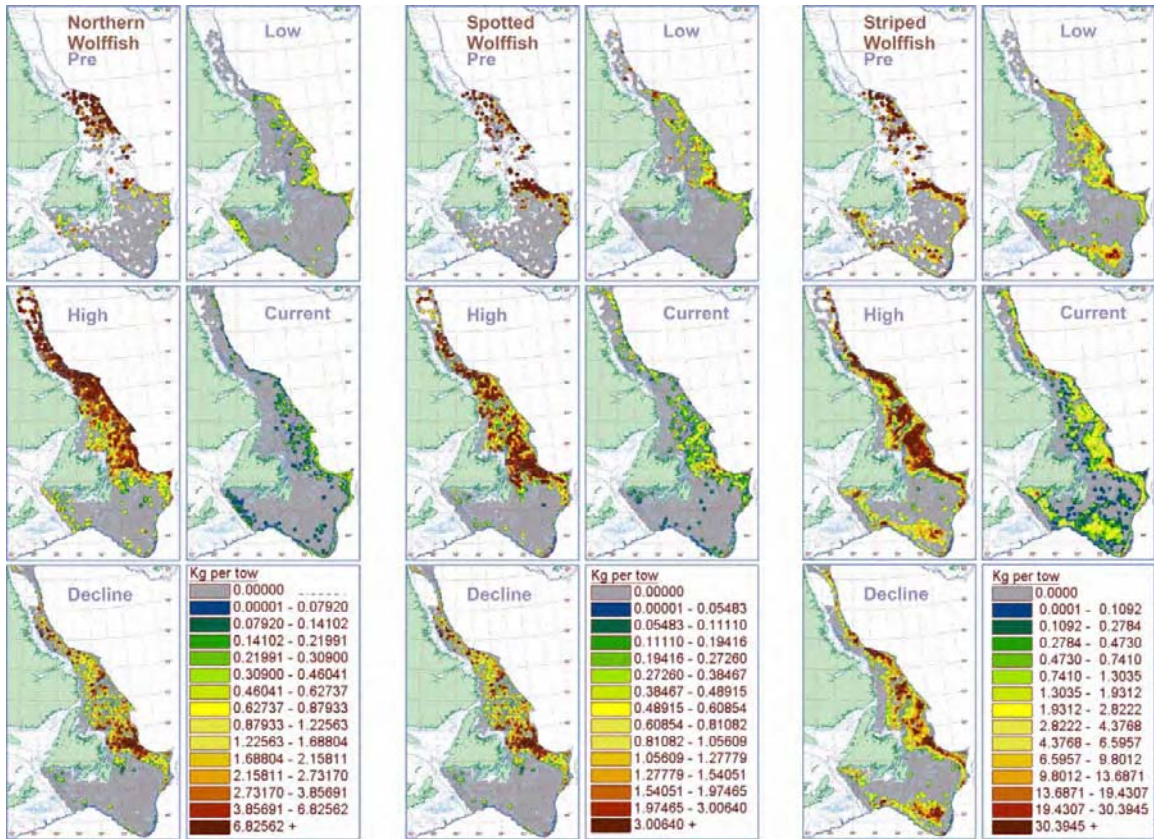
Figure 3.15: Changes in distribution of three wolffish species in the northwest Atlantic: Pre – 1971–1977, High – 1978–1984, Decline – 1985–1989, Low – 19980–1995 (to August), Current – 1995–2004. Grey areas indicate where the survey caught no wolffish (after Kulka *et al.* 2004).
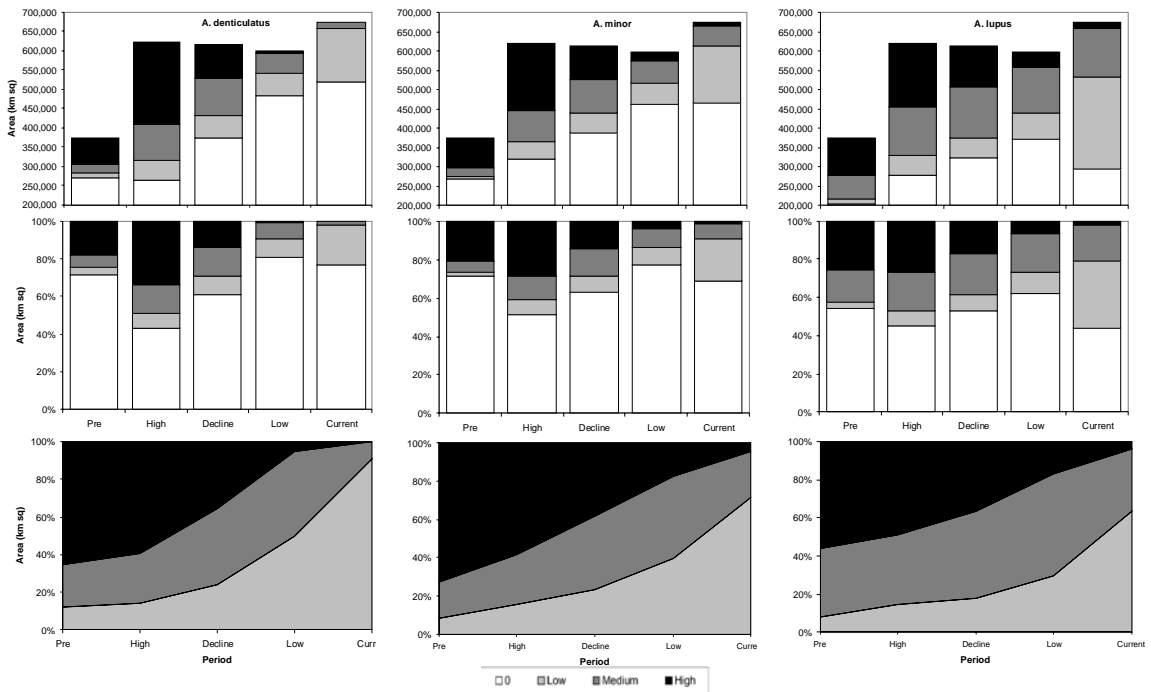
Figure 3.16: Changes in area occupied (upper panels) and degree of concentration (lower panels): *A. denticulatus* – High >2.2 kg per tow; Med 1.5–2.2; Low 0.1–1.5, *A. minor* – High >1.3 kg per tow; Med 0.4–1.3; Low 0.1–1.3. *A. lupus* – High >9.8 kg per tow; Med 1.3–9.8; Low 0.1–1.3. White portions of the bars represent the are surveyed that contain no wolffish (after Kulka *et al.* 2004). Time periods as given in Figure 3.15.
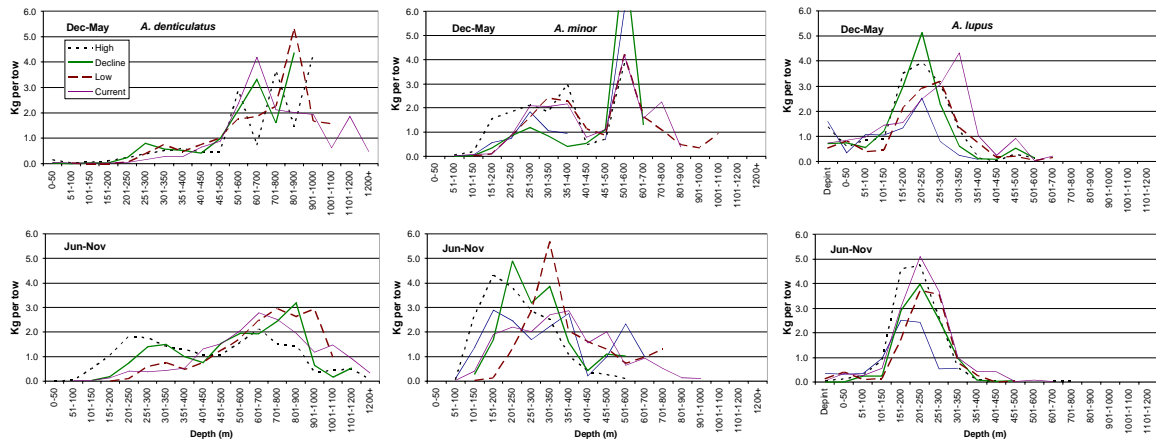
Figure 3.17: Distribution of wolffish in relation to depth. Periods: Pre–1971–1977, High-1978–1984; Decline-1985–1989; Low-1990–1995; Current-1995–2003.
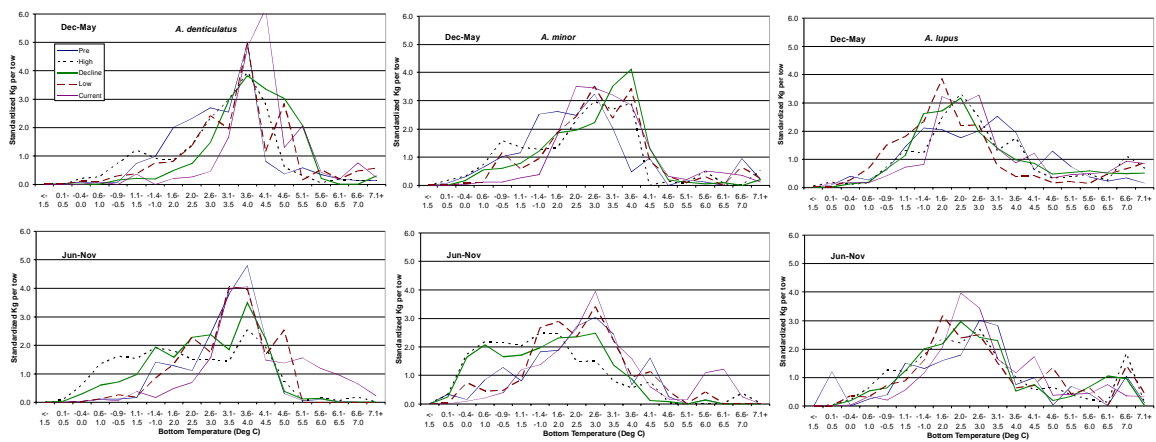
Figure 3.18: Distribution of wolffish in relation to bottom temperature. Periods: Pre – 1971–1977, High, 1978–1984; Decline, 1985–1989; Low, 1990–1995; 1995–2003.

Habitat characteristics of much of the area occupied by wolffish are poorly described. However the current analysis, based on the occurrence of wolffish in research trawls, and the associated depth and temperature collected during the survey sets, indicates that temperature is an important feature of wolffish habitat. On average, *A. denticulatus* are found in deeper waters than either *A. minor* or *A. lupus*. All three species occur in a relatively narrow thermal habitat with above average bottom temperatures, i.e., 1.5–4.5ºC over an area where bottom temperatures < 1.5ºC occupy a large part the range (Figures 3.17–3.18).

Based on the results of this study, wolffish can be classified as "temperature keepers" (sensu Perry and Smith 1994) – they maintain a similar temperature range by changing their depth distribution, as opposed to a depth-keeper such as yellowtail flounder (Perry and Smith 1994) who tolerate a wide range of temperature variation while maintaining their depth distribution. As well, based on focal animal observations of *A. lupus* near shore, it has been observed that they always occur deeper than major haloclines in estuarine locations and thus are not tolerant of low salinity.

Unlike the other two species, *A. lupus* is concentrated on the south central part of the Grand Banks at shallow depths. Bottom temperatures there are relatively warm, but within the observed range of the other two species, so temperature may not explain the absence of *A. denticulatus* and *A. minor* from this area. Bottom type at that location comprises gravel fining to sand with some boulder fields (Kulka 1991), which may be less suitable for the two species.

In addition to associations with temperature, depth and salinity, the distribution of wolffish is also related to sediment type. Based on direct observations of Atlantic wolffish in near-shore areas, they appear to not occur in areas where sediments can be stirred up such as on muddy substrates. As well, boulder areas where eggs can be deposited are also required (Keats *et al.*, 1985). In contrast, analysis of bottom type data using ROXANN data indicated that *A. minor* and *A. lupus* were widely distributed on various sediment types. *A. denticulatus*, however, did occupy sand/shell hash, gravely sand and rock sediments more frequently than the occurrence of those sediment types in the environment would suggest. Conflicting results from individual and large scale methods emphasizes the difficulty in resolving critical habitat features in oceanic areas outside the shallow inshore zone. In addition, observed associations of wolffish with particular sediments, temperatures or depths may be related to additional factors with which these habitat variables are associated such as prey distribution or other environmental factors. Untangling these associations to determine habitat elements that are critical to wolffish requires presently unavailable data collected experimental protocols.

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4   REGNS

4.1   Introduction

WGFE received the following TOR from REGNS:

With reference to the request from REGNS for data on fish individual abundance at length, weight at length, age at length and maturity at length, for all species (both commercial and non-commercial), discards data for all gear types and all fleets, effort data for all gear types and all fleets based on logbook data at the scale of ICES rectangle across the North Sea for the period 1984–2004. Further to this, extract and compile all available stomach content data for diet analysis.

   i.   *determine what parts of the data request can be met within the time frame specified by REGNS;*
   ii.   *identify data gaps and impediments to data access;*
   iii. *interact with the Bureau Working Group on Data and IT issues with a view of contributing to developing a longer term strategy to address issues on data gaps and impediments to data access;*

The Planning Group on Commercial Catch, Discards and Biological Sampling (PGCCDBS) met in 2004, with terms of reference including to:

   • *Further regional coordination and co-operation in collecting biological data of landings of fish and shellfish*
   • *Consider data delivered by fisheries inspectors and how these can be compiled in a consistent way to be used by Assessment Working Groups;*
   • *Compare and standardise protocols for raising national catch and discard data to the international level.*

The resulting report (ICES, 2004) provides information that partly addresses the request from REGNS, and future meetings of this group could provide a useful forum for the further development of a long-term strategy for collating international effort and discards data.

The ICES secretariat will develop a database (interCatch) during 2005 that will store commercial landings, effort and discard data. These data will be provided by the national institutes and will include fields such as country, year and species. Data may be further broken down by fleet, area and season, if specified by the ICES assessment working groups. The assessment workings will by 2006 use the database when raising national catch and discard data to international level. This will ensure that the data manipulation is documented and raising data from national to international are standardised from year to year.

4.2   Survey data (DATRAS)

Within the North Sea, eight countries undertake annual, internationally-coordinated groundfish surveys using the GOV trawl. Data from these surveys are held in the DATRAS database in ICES. Most of the existing biological information of potential interest to REGNS is held in this database.

For the eight standard species, herring, sprat, mackerel, cod, haddock, whiting, saithe and Norway pout data on sex, age, maturity are collected and data are available as a sub-sample of the total catch. For all other species only length are measured.

Data are stored at three levels in the DATRAS database; haul based CPUE, mean CPUE by ICES statistical rectangle and, for the standard species, as mean CPUE by index area. The data are collected by quarter and rectangle and can be analysed at this level for the more abundant

commercial species (Figure 4.1). For many non-target species and less frequent species, data may better be examined at a more regional scale (e.g., ICES division or subarea) and for some of the more unusual species, data may be better examined as presence/absence (Figure 4.2). Furthermore, it must be stressed that not all species are sampled effectively by GOV (e.g., some large-bodied species, small benthic species, epipelagic species), and some habitats (e.g., rocky grounds, estuarine and inshore areas) are not sampled extensively by the survey.

Within the DATRAS database there are several taxa that have been recorded at different taxonomic levels (e.g., species, genus or family) and so analyses of these taxa should only be undertaken for aggregated data. To illustrate the species that are sampled in the IBTS survey, the numbers of each species recorded in Q1 surveys are summarised in Table 4.1.
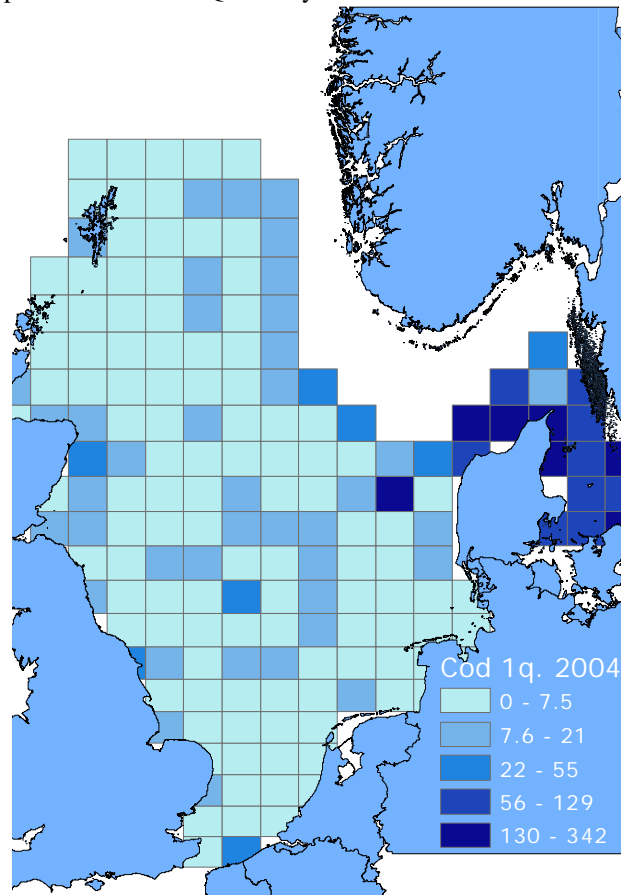
Figure 4.1: Distribution and relative abundance of cod in the North Sea (Q1, 2004).
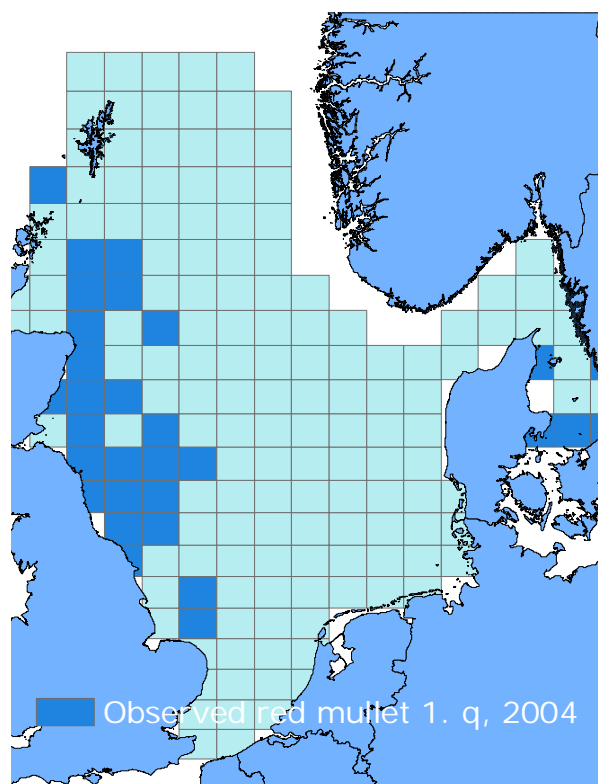
Figure 4.2: Distribution of red mullet in the North Sea (Q1, 2004).

4.2.1 Abundance at length

These data are available for all commercial and non-commercial species, though some data for non-commercial fish taxa may be collected at the level of species, genus or family. These data are to the centimetre below.

4.2.2 Weight at length

Weight at length are collected by the national institutes for most commercial and non-commercial species, though some data for non-commercial fish taxa may be collected at the level of species, genus or family. Length data are to the centimetre below, though the resolution of individual weight will vary between 1 and 5g. Historically these data have not been delivered to ICES and access to these data can only be obtained by requesting each institute. By 2005 it will be possible for national institutes to submit weight at length data to the DATRAS database and it is expected that these data will be available in the future at least on the raw sub-sampled level.

4.2.3 Age at length

These data are available for the following commercial species; herring, sprat, mackerel, cod, whiting, haddock, saithe and Norway pout. The data are available as CPUE data from the DATRAS database and can be used directly into analysis.

4.2.4 Maturity at length

These data are available for following commercial species; herring, sprat, mackerel, cod, whiting, haddock, saithe and Norway pout, with some increase in maturity data for skates and rays

(Rajidae) in recent years. These data may be available at the scale of ICES rectangle for the most abundant commercial teleost species, but in most instances data should only be examined at a more regional scale (e.g., roundfish area, ICES division or subarea). The data are only available from the DATRAS database on the sub sample level and before data can be used in analysis they have to be raised to CPUE.

4.3 Other survey data

Many nations bordering the North Sea will have nationally-held data sets for those surveys that are not part of internationally-coordinated surveys. Such surveys may be part of long-term monitoring programmes, or more specific projects with a more restricted temporal and spatial resolution. Examples of the kinds of data that are available are given below.

NATION:	UK (ENGLAND AND WALES)
Type of data:	**Young fish survey**
Period covered:	1983 to 2004
Area covered:	Eastern English Channel and North Sea (50–55ºN)
Description of data:	Data prior to 2001 held in SAS data sets as numbers per 1000 m sq, with length data on paper records. Since 2002 data held in FSS database and including estimates of benthic by-catch. Temperature and salinity data, and grab samples recorded at most locations. Data for push net and 2-metre beam until 1999, since then only 2-m beam trawl has been used.
Data owner:	CEFAS/Defra
Data availability:	-
Recent papers describing or using data:	Rogers *et al.* (1998)

NATION:	UK (ENGLAND AND WALES)
Type of data:	4m-beam trawl survey
Period covered:	1989 to 2004
Area covered:	Covers parts of ICES area IVc, though most of the survey grid is in the eastern English Channel.
Description of data:	Distribution and relative abundance of fish and shellfish, catch per unit effort (numbers and biomass), size frequency, biological data for plaice, sole, whiting, cod, skates and rays
Data owner:	CEFAS/Defra
Data availability:	These data will eventually be included in the DATRAS database, but are currently held nationally
Recent papers describing or using data:	Kaiser *et al.* (1999)

4.4 Discards data

Improved coordination of discard sampling has occurred in recent years (e.g., Study Group on Discards and Bycatch information, SGDBI). An example of the kind of discards data that are available is given below:

NATION:	UK (ENGLAND AND WALES)
Type of data:	Discards data
Period covered:	1994 to present
Area covered:	NE English trawl fisheries in IVb mainly until 2001. Subsequently all English & Welsh >10m vessels sampled under EC Data Collection Regulation 1639/2001.
Description of data:	Numbers at length and age for cod, haddock, and whiting, maybe some other species, up until 2001. Subsequently, all commercial spp were sampled, and non-commercial fish as time on deck permitted. Vessel and trip details are also held.

Data owner:	Fishing vessel owners and skippers + CEFAS
Data availability:	Fully available if aggregated so that individual vessels cannot be identified.
Recent papers describing or using relevant data:	Allen *et al.* (2001, 2002), Cotter *et al.* (2002), Course *et al.* (1999), Tamsett *et al.* (1999)
NATION:	**UK (SCOTLAND)**
Type of data:	Discards data
Period covered:	Demersal: 1975 – present (Area IVa, IVb) 1978 – present (Area VIa) Pelagic: 1997 – present *Nephrops*: 1990 - present
Area covered:	Demersal: Pelagic: *Nephrops*: Firth of Forth, Moray Firth, Fladden (2000 onwards), South Minch and Clyde (as defined by WGNEPH)
Description of data:	Demersal: Age compositions and length frequency of discards, as well as mean lengths, and weights at age for cod, whiting, haddock and saithe. For species that are not sampled for age (multispecies discards), discarded quantities are sampled. The fleets sampled are motor trawl, light trawl, demersal pair trawl, seine net, *Nephrops* trawl Pelagic: Age, length, weight and maturity for herring. Age, length, weight and maturity (2004 onwards) for mackerel. *Nephrops*: Length, weight and maturity.
Data owner:	Fishing vessel owners and skippers and FRS
Data availability:	Fully available if aggregated so that individual vessels cannot be identified. Multispecies discards are not publicly available.
Recent papers describing or using relevant data:	-

NATION	**GERMANY**
Type of data:	Discards data
Period covered:	Some data are available prior to 1998, but not all electronic.
Area covered:	North Sea
Description of data:	Discards data by gear type and fleet, though fleet definitions have changed over time. Biological data (age, weight, maturity and abundance at length) are available for commercial species, though fewer data are available for non-commercial species
Data owner:	
Data availability:	Available to ICES Working Groups
Recent papers describing or using relevant data:	

4.5   Effort data

Most nations will collect some form of effort data for the major fleets, though raw data are not generally available. Aggregated data are usually available for major gear types by ICES areas and month. Other data that may be available include:

NATION:	**UK**
Type of data:	Overflight data
Period covered:	1985 to present
Area covered:	England and Wales waters
Description of data:	Collected routinely by fishery protection aircraft and vessels, and the Royal Navy. Information recorded includes the location, type and identification number of all vessels sighted in the territorial waters of England and Wales, and whether or not they are fishing. Overflight surveillance has been scaled down considerably since the introduction of satellite monitoring in 2000.
Data owner:	Crown
Data availability:	Anonymised data on request from the Sea Fisheries Inspectorate
Recent papers describing or using data:	Rogers *et al.* (2001) Jennings *et al.* (1999)

NATION:	UK
Type of data:	Satellite Vessel Monitoring System (VMS)
Period covered:	2000 to present
Area covered:	European seas
Description of data:	From 1 January 2000, all EC fishing vessels over 24m were required to report their location, via satellite, to monitoring centres in their flag states, at 2-h intervals. The only exception is made for vessels that undertake trips of <24 h or fish exclusively within territorial waters. VMS 'black box' recorders emit regular signals transmitting vessel position, speed, bearing and vessel registration number. The signal is first received by satellites and then by national receiving stations on the ground. From 2004, VMS coverage was extended to all vessels over 15m.
Data owner:	Vessel owners / Crown
Data availability:	Anonymised data on request from the Sea Fisheries Inspectorate
Recent papers describing or using data:	Dinmore *et al.* (2003), Mills *et al.* (2004)

NATION	UK
Type of data:	Landings and effort data (for UK vessels landing in UK and elsewhere and non-UK vessels landing into UK)
Period covered:	1984 – present
Area covered:	ICES area IV
Description of data:	Landings and effort data for UK vessels aggregated across all fishing trips within a particular area/month/gear sampling cell. As the data are aggregated across trips, the effort data within a gear stratum are not standardised according to fishing power other than by the fact some gear stratification is based on vessel size. Effort data are nominally available by days absent, days fished and hours fished. Hours fished is based on a non-mandatory data field in vessel logbooks and changing practices regarding the recording of these data have lead to a severe distortion of the recorded effort data in recent years.
Data owner:	Defra
Data availability:	FRS, for example, has access to the Scottish Executive FIN database of individual trip records. In FIN data are recorded by trip rather than haul. In FIN, landings and effort data are recorded by trip rather than haul, this may require values to be split in more-or less arbitrary ways if more than one statistical rectangle is fished within a single trip but means that these data are available by statistical rectangle. The use of individual trip data means that effort data can, in principle, be derived on the basis of selecting trips from appropriate métiers and, through the use of appropriate models, power-corrected CPUE values can be derived. Analysis of métiers has been undertaken for three years (2000–2002), but power-corrected effort values have yet to be calculated for any year.
Recent papers describing or using data:	

4.6   Stomach contents data

The diets of many North Sea fish species have been quantified (see Section 10). Most studies have been specific projects targeting either certain regions or taxa, and many of these studies date from before 1984. More comprehensive data for the stomach contents of North Sea fishes are available from the 1981 Stomach Sampling Project (Daan, 1989) and 1991 Stomach Sampling Project (ICES, 1997), and these studies provide good seasonal and spatial coverage.

4.6.1   Stomach Sampling Project (1981)

The first internationally coordinated study examined the diets of five species (cod, haddock, whiting, saithe and mackerel) by quarter, size class and ICES rectangle/roundfish area (Table 4.2; Daan, 1989), with nine countries (Belgium, Denmark, England, Germany, Netherlands, Norway, Russia and Scotland) participating. The results from this study have been discussed for cod (Daan, 1983; Cramer and Dean, 1986; Bromley and Kell, 1993), haddock (Villermarque and de la Villemarque, 1985; Cramer and Dean, 1986; Bromley and Kell, 1993; Alder-

stein *et al*., 2002), whiting (Hislop *et al*., 1983), saithe (Gislason, 1983) and mackerel (Mehl and Westgaard, 1983).

4.6.2 Year of the stomach (1991)

The second internationally coordinated study examined the diets of those species examined in 1981, but also examined other species, including grey gurnard, rays and scad (Table 4.3). Once again, data were collected by quarter, size class and ICES rectangle/roundfish area (ICES, 1997). The results from this study have been discussed for cod (Dorman *et al*., 1992; Kikkert, 1993), haddock (Alderstein *et al*., 2002), whiting (Robb *et al*., 1994), grey gurnard (Gee and Kikkert, 1993) and rays (Daan *et al*., 1993).

4.7 Data gaps

Within the DATRAS database there is a need for expanding the facilities with the possibility to allocate maturity ogives and weight at length to the CPUE values. However, there have not been the required resources in the ICES secretariat to include these facilities in 2005.

4.8 Long-term strategy

When the new interCatch database is developed (expected by the end of 2005), commercial data from 2005 will be available from one common source. With regard to other future data and database development the Bureau Working Group on Data and IT issues will provide the longer-term strategy as well as a new data access policy for ICES data in general.

4.9 References

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Table 4.1: Total CPUE per species by time period from the IBTS survey.

TSN CODE	GENUS	SPECIES	1983–1990	1991–1995	1996–2000	2001–2004
159719	Lampetra	fluviatilis		2		
159722	Petromyzon	marinus		2	14	24
159772	Myxine	glutinosa		1001	5352	1791
160065	Scyliorhinus	canicula	947	724	1498	3016
160067	Scyliorhinus	stellaris	10			2
160181	Galeorhinus	galeus	10	4	10	4
160226	Mustelus			218		
160240	Mustelus	asterias		12	427	475
160242	Mustelus	mustelus	28	231	51	118
160604	Squalidae		142			
160611	Somniosus	microcephalus			2	2
160617	Squalus	acanthias	1620	549	639	407
160670	Etmopterus	spinax	9	4	2	2
160845	Rajidae		23	6	6	
160846	Raja		70	12		
160880	Raja	brachyura	18	39	9	6
160883	Raja	montagui	377	646	330	401
160900	Raja	undulata	3			
160901	Raja	clavata	2074	6505	901	835
161022	Chimaera	monstrosa	55	26	68	8
161128	Anguilla	anguilla	37	24	22	26
161341	Conger	conger	6		3	
161701	Alosa		2			
161708	Alosa	alosa	10	10	154	134
161716	Alosa	fallax	11	55	3431	1662
161722	Clupea	harengus	15898617	6371955	7645374	8278191
161789	Sprattus	sprattus	6569382	4875329	5026637	4104372
161813	Sardina	pilchardus	78	54	1050	3898
161831	Engraulis	encrasicolus	11	1174	5112	8827
161996	Salmo	salar	2			
161997	Salmo	trutta	8	12		
162039	Osmerus	eperlanus	1550	41	188	12
162057	Argentinidae		1184			
162061	Argentina		210		4068	
162064	Argentina	silus	632	1241	2876	2772
162071	Argentina	sphyraena	16205	39881	29198	24928
162187	Maurolicus	muelleri	24135	2294	4656	731
164475	Lepadogaster			2		
164482	Diplecogaster	bimaculata	1	2		
164501	Lophius	piscatorius	1343	1458	1486	964
164502	Lophius	budegassa		4		
164712	Gadus	morhua	178759	74357	96174	39281
164727	Pollachius	virens	67541	14252	16631	36462
164728	Pollachius	pollachius	2143	1669	326	319
164740	Brosme	brosme	204	158	148	127
164744	Melanogrammus	aeglefinus	2188423	1574094	1998066	1574220
164748	Enchelyopus	cimbrius	6167	7007	9471	5261
164751	Phycis	blennoides	5	9	2	
164754	Trisopterus	minutus	76260	45162	30431	51039

TSN CODE	GENUS	SPECIES	1983–1990	1991–1995	1996–2000	2001–2004
164755	Trisopterus	luscus	10802	6800	3912	2196
164756	Trisopterus	esmarkii	5246431	4808370	7377723	2358300
164758	Merlangius	merlangus	4354678	3275947	2104947	2180186
164760	Molva	molva	2397	873	658	768
164761	Molva	dypterygia	4	2		
164764	Gaidropsarus		8			
164765	Gaidropsarus	vulgaris	114	122	76	39
164766	Gaidropsarus	mediterraneus	16	4		
164771	Gadiculus		47			
164772	Gadiculus	argenteus	2549	4184	8733	8032
164774	Micromesistius	poutassou	2595	269	13472	9266
164777	Raniceps	raninus	3	20	14	4
164779	Ciliata	mustella	188	260	226	433
164780	Ciliata	septentrionalis	24		1	28
164789	Merlucciidae		48			
164795	Merluccius	merluccius	4568	2393	3217	1409
165116	Echiodon	drumondi	10	1	14	2
165215	Zoarcidae				2	
165243	Lycenchelys	sarsi		95	236	124
165284	Lycodes	vahlii	1803	4691	5050	1694
165324	Zoarces	viviparus	1611	314	210	48
165350	Coryphaenoides	rupestris	1		4	2
165419	Trachyrincus	murrayi		1		
165594	Belone	belone	12		9	
165612	Scomberesox	saurus	2			
166025	Atherina	presbyter			16	4
166287	Zeus	faber	12	28	53	80
166309	Caproidae			6		
166320	Capros	aper	3	13	18	8
166363	Gasterosteidae		4			
166365	Gasterosteus	aculeatus	4738	1284	945	349
166401	Spinachia	spinachia		6	953	6
166443	Syngnathidae		76	224	189	780
166444	Syngnathus				2	
166463	Syngnathus	rostellatus	28	90	200	790
166464	Syngnathus	acus	36	26	194	101
166467	Syngnathus	typhle			12	16
166591	Entelurus	aequerius	15	4	24	49
166595	Nerophis	ophidion		2		7
166613	Acentronura					2
166745	Sebastes	marinus	174			2
166779	Sebastes	viviparus	2045	895	310	488
166787	Helicolenus	dactylopterus	8	3382	324	52
166839	Scorpaena	scrofa		6		
166972	Triglidae		2741	14		
167039	Trigla	lucerna	170	69	80	79
167044	Eutrigla	gurnardus	173192	184534	376915	326157
167046	Trigloporus	lastoviza				4
167049	Aspitrigla	cuculus	474	881	1844	1519
167196	Cottidae		14			
167209	Artediellus	europaeus			2	

TSN CODE	GENUS	SPECIES	1983–1990	1991–1995	1996–2000	2001–2004
167316	Myoxocephalus	quadricornis				70
167317	Myoxocephalus	scorpioides				200
167318	Myoxocephalus	scorpius	6364	3182	1846	3793
167375	Triglops	murrayi	22	12	13	22
167390	Taurulus	bubalis	305	394	1888	820
167391	Taurulus	lilljeborgi			16	
167454	Agonus	cataphractus	7134	5188	4638	7278
167478	Leptagonus	decagonus			2	
167550	Liparis		269	15		4
167578	Liparis	liparis	1174	512	723	717
167581	Liparis	montagui	6	26	44	72
167612	Cyclopterus	lumpus	2247	1220	1021	546
168588	Trachurus	trachurus	9208	2453	35046	97919
169180	Sparidae			4		
169215	Pagellus	erythrinus		2		
169229	Spondyliosoma	cantharus	15	11		6
169418	Mullus	surmuletus	26	38	805	943
170317	Dicentrarchus	labrax		15	65	18
170333	Mugilidae				2	
170733	Ctenolabrus	rupestris			20	6
170737	Labrus	bergylta		5		4
170991	Trachinus	vipera	4338	4616	11366	21684
170992	Trachinus	draco	13524	156	1252	56
171341	Anarhichas	lupus	993	380	163	98
171342	Anarhichas	minor	2			
171554	Stichaeidae			4		2
171588	Lumpenus	lumpretaeformis	413	853	7302	2367
171603	Leptoclinus	maculatus	2			
171645	Pholis	gunnellus	117	1598	39	354
171670	Ammodytidae		19271	6493	27450	30820
171671	Ammodytes		28015	26		29442
171676	Ammodytes	tobianus	4787	14	872	31767
171677	Ammodytes	marinus	12078	876	4557	356
171680	Gymnammodytes	semisquamatus		18	8	
171681	Hyperoplus				124	
171682	Hyperoplus	lanceolatus	580	2648	1686	3023
171683	Hyperoplus	immaculatus		86	10922	295
171691	Callionymidae		325	158	110	
171692	Callionymus				256	27
171698	Callionymus	lyra	6813	7845	4186	6594
171699	Callionymus	maculatus	2390	10543	14754	5680
171712	Callionymus	reticulatus	14	245	70	278
171746	Gobiidae		2308	3883	2404	1359
171833	Gobius		18	108		56
171841	Gobius	cobitis				2
171850	Gobius	niger	2	2	4	8
171971	Crystallogobius	linearis	8		8	28
171977	Pomatoschistus		1910	5912	7654	1598
171978	Pomatoschistus	minutus	752	1752	1780	1291
171982	Pomatoschistus	microps		34		
172033	Aphia	minuta				57

TSN CODE	GENUS	SPECIES	1983–1990	1991–1995	1996–2000	2001–2004
172034	Lesueurigobius			6	8	
172036	Lesueurigobius	friesii			37	20
172414	Scomber	scombrus	48825	12578	306215	70742
172421	Thunnus	thynnus			2	
172714	Bothidae			16	16	
172749	Scophthalmus	rhombus	264	255	250	377
172803	Arnoglossus			6	11	6
172805	Arnoglossus	laterna	576	847	1843	2260
172806	Arnoglossus	imperialis		2		4
172828	Zeugopterus			4		
172829	Zeugopterus	punctatus	21	37	70	88
172835	Lepidorhombus	whiffiagonis	2575	1480	1450	1361
172873	Glyptocephalus	cynoglossus	4080	4712	6000	2645
172877	Hippoglossoides	platessoides	248101	293069	447722	227720
172881	Limanda	limanda	2013278	1231772	916613	942912
172888	Microstomus	kitt	22747	13163	19713	19726
172894	Platichthys	flesus	32966	24946	15096	9632
172902	Pleuronectes	platessa	185476	53768	63927	69428
172933	Hippoglossus	hippoglossus	77	62	51	76
172980	Soleidae		92	523		
173001	Solea	vulgaris	2221	3633	2329	1407
173020	Buglossidium		2	72	4	
173021	Buglossidium	luteum	1874	4640	9213	21117
173022	Microchirus			10		
173026	Microchirus	variegatus	11	2	139	93
173051	Pegusa	lascaris				10
564126	Dipturus	batis	69	17	18	2
564128	Leucoraja	circularis	4		10	
564134	Leucoraja	fullonica	4	11	12	4
564140	Leucoraja	lentiginosa			2	
564141	Dipturus	linteus			2	
564143	Leucoraja	naevus	1170	991	610	429
564149	Amblyraja	radiata	13382	12969	11056	5656
615903	Lamprididae			2		
616195	Psetta	maxima	502	446	204	259
616605	Zeugopterus	regius	5	2		
616613	Zeugopterus	norvegicus	90	85	105	72

Table 4.2: Number of fish examined in the 1981 stomach sampling project (Adapted from Daan (1989)). Cod, haddock and whiting were sampled in 1981, saithe were sampled in 1980–1982, and mackerel from 1980–1983.

Cod	Area	Q1	Q2	Q3	Q4	Total
	1	740	470	460	275	1945
	2	718	205	325	444	1692
	3	255	166	237	112	770
	4	267	314	226	119	926
	5	21	62	49	96	228
	6	1789	1170	884	1256	5099
	7	356	43	148	211	758
	Total	4146	2430	2329	2513	**11418**
Haddock	Area	1	2	3	4	Total
	1	645	2131	3093	2512	8381
	2	407	395	930	805	2537
	3	1119	909	1320	172	4520
	4	169	193	356	194	912
	5	6	2	3	6	17
	6	212	165	123	93	593
	7	252	0	0	184	436
	Total	2810	3795	5825	4966	**17396**
Whiting	Area	1	2	3	4	Total
	1	1615	1334	954	539	4442
	2	940	214	393	555	2102
	3	1867	1104	608	432	4011
	4	713	277	363	243	1596
	5	108	184	258	38	588
	6	2035	1098	1066	1213	5412
	7	554	0	85	427	1066
	Total	7832	4211	3727	3447	**19217**
Mackerel	Area	1	2	3	4	Total
	Northwestern	48	39	344	204	635
	Northeastern	59	324	637	38	1058
	Central	122	411	936	255	1724
	Southern	19	503	820	186	1528
	Total	248	1277	2737	683	**4945**
Saithe	All areas	547	185	899	559	**2190**

Table 4.3a: Number of cod, haddock, whiting, saithe and mackerel examined in the 1991 stomach sampling project (Adapted from ICES (1997)).

COD	AREA	Q1	Q2	Q3	Q4	TOTAL
	1	526	806	614	657	2603
	2	232	769	627	217	1845
	3	142	173	298	39	652
	4	121	104	164	17	406
	5	44	61	47	22	174
	6	910	927	276	808	2921
	7	207	334	347	239	1127
	8	6	0	0	610	616
	Total	2188	3174	2373	2609	**10344**
Haddock	Area	1	2	3	4	Total
	1	1065	1143	2397	1631	6236
	2	669	784	1003	644	3100
	3	530	441	1021	475	2467
	4	28	155	274	85	542
	5	0	3	0	0	3
	6	2	16	1	11	30
	7	38	210	246	33	527
	8	0	0	0	205	205
	Total	2332	2752	4942	3084	**13110**
Whiting	Area	1	2	3	4	Total
	1	1546	2618	3472	2594	10230
	2	1234	2197	1870	1284	6585
	3	1242	1134	2117	1131	5624
	4	228	844	991	409	2472
	5	208	580	482	566	1836
	6	1114	3115	1564	2818	8611
	7	579	842	1046	568	3035
	8	0	0	0	1047	1047
	Total	6151	11330	11542	10417	**39440**
Mackerel	Area	1	2	3	4	Total
	1	289	240	1244	235	2008
	2	1	520	396	24	941
	3	1	103	259	75	438
	4	0	102	46	0	148
	5	0	137	1	8	146
	6	0	993	624	363	1980
	7	0	235	227	0	462
	8	0	0	0	0	0
	Total	291	2330	2797	705	**6123**
Saithe	Area	1	2	3	4	Total
	1	719	1105	364	824	3012
	2	6	15	28	4	53
	3	0	12	0	0	12
	4	0	0	1	0	1
	5	0	0	0	0	0
	6	0	3	1	1	5
	7	59	43	0	25	127
	8	0	0	0	0	0
	Total	784	1178	394	854	**3210**

Table 4.3b: Number of other fish species examined in the 1991 stomach sampling project (Adapted from ICES (1997)).

SPECIES	Q1	Q2	Q3	Q4	TOTAL
Tope	1	3	11	17	**32**
Starry smooth hound	0	3	3	1	**7**
Lesser-spotted dogfish	35	43	93	4	**175**
Spurdog	49	366	58	75	**518**
Starry ray	637	651	1475	438	**3201**
Thornback ray	109	18	72	7	**206**
Cuckoo ray	83	54	51	4	**192**
Spotted ray	45	13	29	46	**133**
Pollack	0	58	15	8	**81**
Ling	52	71	33	48	**204**
Hake	54	176	226	93	**549**
Torsk	3	14	5	5	**27**
Bib	56	17	0	29	**102**
Blue whiting	0	0	7	0	**7**
Anglerfish	29	54	111	66	**260**
Norway haddock	0	0	7	0	**7**
Grey gurnard	1916	4128	3955	1701	**11700**
Scad	31	941	1555	986	**3513**
Red gurnard	132	19	17	2	**170**
Tub gurnard	0	106	148	137	**391**
Red mullet	0	26	0	0	**26**
Wolf-fish	1	2	0	0	**3**
Greater sandeel	3	779	8	4	**794**
Sandeel (indet.)	0	65	0	0	**65**
Turbot	13	25	17	33	**88**
Brill	14	10	1	2	**27**
Megrim	60	40	128	16	**244**
Long-rough dab	574	414	1172	353	**2513**
Halibut	2	7	0	2	**11**

5   Gear specific catchability of elasmobranchs in the North Sea

5.1   Species present

Although 36 species of elasmobranch have been reported from the North Sea (ICES, 2004a; Table 5.1), many of these are either occasional vagrants, or are deep-water species that occur only in the northern North Sea and Norwegian Deep. The species that are most widespread and common in the North Sea are spurdog *Squalus acanthias*, lesser-spotted dogfish *Scyliorhinus canicula*, tope *Galeorhinus galeus*, starry smooth hound *Mustelus asterias*, common smooth hound *M. mustelus*, starry ray *Amblyraja radiata*, cuckoo ray *Leucoraja naevus*, blonde ray *Raja brachyura*, thornback ray *R. clavata* and spotted ray *R. montagui*. Species of conservation interest that were formerly common in the North Sea include common skate *Dipturus batis* and angel shark *Squatina squatina*. The distribution patterns for many of these species in the North Sea are illustrated in Knijn *et al*. (1993) and Ellis *et al*. (2005). In terms of large pelagic sharks, the occurrence of porbeagle *Lamna nasus* and thresher shark *Alopias vulpinus* have been discussed by Gauld (1989) and Ellis (2004) respectively.

Table 5.1: Elasmobranch fishes known from the North Sea.

Hexanchidae	*Hexanchus griseus*	Six-gilled shark
Alopiidae	*Alopias vulpinus*	Thresher shark
Cetorhinidae	*Cetorhinus maximus*	Basking shark
Lamnidae	*Isurus oxyrinchus*	Mako shark
	Lamna nasus	Porbeagle shark
Scyliorhinidae	*Galeus melastomus*	Blackmouthed dogfish
	Scyliorhinus canicula	Lesser-spotted dogfish
	Scyliorhinus stellaris	Nurse hound
Triakidae	*Galeorhinus galeus*	Tope shark
	Mustelus asterias	Starry smooth hound
	Mustelus mustelus	Smooth hound
Carcharhinidae	*Prionace glauca*	Blue shark
Sphyrnidae	*Sphyrna zygaena*	Common hammerhead
Dalatiidae	*Dalatias licha*	Darkie charlie
	Etmopterus spinax	Velvet belly
	Somniosus microcephalus	Greenland shark
Squalidae	*Squalus acanthias*	Spurdog
Echinorhinidae	*Echinorhinus brucus*	Bramble shark
Squatinidae	*Squatina squatina*	Angel shark
Torpedinidae	*Torpedo marmorata*	Marbled electric ray
	Torpedo nobiliana	Common electric ray
Rajidae	*Amblyraja radiata*	Starry ray
	Dipturus batis	Common skate
	Dipturus nidarosiensis	Black skate
	Dipturus oxyrinchus	Long-nose skate
	Leucoraja circularis	Sandy ray
	Leucoraja fullonica	Shagreen ray
	Leucoraja naevus	Cuckoo ray
	Raja brachyura	Blonde ray
	Raja clavata	Thornback ray
	Raja microocellata	Painted ray
	Raja montagui	Spotted ray

	Raja undulata	Undulate ray
	Rajella fyllae	Round skate
Myliobatidae	*Myliobatis aquila*	Eagle ray
Dasyatidae	*Dasyatis pastinaca*	Sting ray

5.2   Landings and fisheries

Annual landings data, as submitted to ICES, have been collated and summarised by WGEF and the commercial fisheries operating in the North Sea described (ICES, 2003, 2004b). Landings from the UK (Table 5.2) highlight the major types of gear used in North Sea fisheries. Spurdog and mixed skates and rays dominate the reported landings, and are taken in both mixed and targeted fisheries, with other species (e.g., *Scyliorhinus, Galeorhinus, Mustelus*) generally taken as by-catch. Spurdog are landed primarily by long line, otter trawl and seine, with smaller quantities taken by gill net and beam trawl. Skates and rays are landed primarily by otter trawl, long line, gill net and beam trawl.

Table 5.2: Total landings (Tonnes) of elasmobranchs by UK vessels (and non-UK vessels landing into UK ports) for the period 1984–2004 (Defra landings data).

	OTTER TRAWL	GILL NET	LINES	BEAM TRAWL	SEINE (DANISH, FLY, PURSE)	DRIFT NET	OTHER	TOTAL
Dogfish indet.	120.1	130.8	14.4	5.9	7.0	5.5	1.9	285.7
Greater spotted dogfish	20.0	12.6	0.5	0.0	0.6	0.0	0.0	33.8
Lesser spotted dogfish	106.0	360.3	8.6	8.2	0.0	0.1	2.7	485.9
Porbeagle	0.6	0.1	0.1	0.0	0.0	0.0	0.0	0.8
Spurdog	7891.8	574.6	12320.1	285.5	4487.1	3.6	7.0	25569.8
Tope	64.5	63.5	320.2	1.4	5.7	5.9	1.2	462.3
Scyliorhinidae	0.0	0.1	0.0	0.5	0.0	0.0	0.0	0.6
Sharks	34.3	305.5	2.4	7.2	2.7	0.3	0.1	352.5
Skates and rays	9586.0	2363.7	6582.3	1720.4	163.8	63.4	34.0	20513.7
Smoothhound	6.9	22.6	12.2	0.4	0.0	1.9	0.0	44.0

Seasonal patterns in landings for spurdog and mixed skates and rays (from UK landings data) are illustrated in Figures 5.1–5.8 for the major gear types (otter and beam trawl, gill net and long line). Spurdog were taken by gill net in the southern North Sea, especially in the Outer Thames Estuary, and west of the Shetland Islands (Qtr 2). Beam trawl landings were also predominantly from the southern and central North Sea. Landings from otter trawl more wide-

spread, with UK landings highest off North-East England. Skates and rays were taken by gill net in the southern North Sea and west of the Shetland Islands, with beam trawl and long line landings also predominantly from the southern and central North Sea, and landings from otter trawl more widespread. UK landings were highest in the Outer Thames Estuary and off East Anglia.

Seasonal and spatial patterns in reported landings of elasmobranchs should, however, be viewed in relation to the distribution of the species in question and in the overall spatial and temporal distribution of fishing effort. Such data are not readily available for the entire North Sea, as such data are typically held by national institutes. Fishing effort of UK vessels from satellite monitoring is illustrated in Figures 5.9–5.11, and although the areas of reportedly high elasmobranch landings from trawlers generally correspond with the distribution of fishing effort for beam and otter trawl, it is apparent that the reported landings of elasmobranchs in fixed gears (long lines, gill nets) around the coast of East Anglia and in the Outer Thames Estuary are not reflected in satellite monitoring data. Hence, other sources of effort data will be required for the inshore metiers targeting elasmobranchs.

5.3 Discards

Data from the CEFAS discards database was examined in order to determine the sizes of various species that were retained or discarded during observer trips on fishing vessels in the North Sea. These data are illustrated for spurdog, scyliorhinids (aggregated), tope and smooth hounds (combined) and the major skate species (Figures 5.12–5.14). For many species, these data are too limited to enable the capture and levels of discarding to be assessed for the various metiers. Nevertheless, they do provide some information on the length at first capture (Table 5.3), which for several species equates with size at birth or hatching. Data for trawl fisheries are more comprehensive, though data for the targeted fixed gear fisheries are extremely limited (Figure 5.15).

Table 5.3: Size at first capture and minimum/maximum sizes for discarded and retained elasmobranchs (UK data, North Sea, all gears combined, 2002–2004)

SPECIES	SIZE AT FIRST CAPTURE	RETAINED		DISCARDED	
		Min LT	Max LT	Min LT	Max LT
Blonde ray	25*	40	106	25	55
Cuckoo ray	29*	35	60	29	62
Spotted ray	10	39	69	10	59
Starry ray	12	27	52	12	51
Thornback ray	10	41	95	10	59
Galeorhinus	53	82	120	53	91
Mustelus	26	31	120	26	115
Scyliorhinidae	20	25	103	20	79
Squalus	47	47	107	65	93
*Observed values could be higher than true values, with other species of ray caught at 10–12cm					

5.4 Sustainable fishing mortality rates of elasmobranchs

The major change differences in the richness of UK shallow water fish faunas can be attributed to the decline in abundance of elasmobranchs, this is particularly apparent in the southern North Sea (Jennings, Greenstreet and Reynolds, 1999; Jennings, Reynolds and Mills, 1998; Rogers, Clarke and Reynolds, 1999). Declines and local extinctions of some species have re-

sulted in concerns for their conservation. The common skate is now found only in the NW North Sea and the thornback ray is now most abundant in a few restricted locations, including the Thames estuary area and north coast of East Anglia (Walker, Howlett and Millner, 1997). Assessing vulnerability to exploitation requires estimates of the current fishing mortality rate *F* and some measure of the capacity of the population to withstand that mortality (Pope *et al.*, 2000).

The usual approach to estimating sustainable mortality rates used age-based Virtual Population Analysis (Pope *et al.*, 2000), or length-based assessment methods (Jones, 1981; Pauly, 1987). However such data are often not available for non-target species due to their low economic importance, which necessitates the use of less demanding life history and demographic approaches (Dulvy *et al.*, 2004). Age structured demographic models have been used to estimate maximum total mortality, (fishing plus natural mortality) that the population could withstand, provided that each female replaces herself in the population (Walker and Hislop, 1998). This is based on the Euler-Lotka equation:

$$1 = \sum_{a}^{d} e^{-rx} l_x m_x$$

Here *a* is the age at maturity and *d* is the maximum age attained, and l_x and m_x are survival and number of offspring produced at each age, *x*. Walker and Hislop (1998) calculated the replacement mortality, i.e., the total mortality where *r* = 0, and compared this hypothetical mortality rate for a stable population with estimates of actual mortality (last two columns of Table 2). This comparison suggests that only the smallest species of ray is being killed sustainably. This result is particularly sensitive to age at maturity, rather than fecundity (Brander, 1981). This approach requires that assumptions are made on the age-specific scheduling of survival and offspring production, and consequently it is difficult parameterise these models with any confidence due to the difficulty of ageing individuals and calculating age-specific vital rates (Crouse, Crowder and Caswell, 1993; Miller, Frisk and Fogarty, 2002; Mollet and Cailliet, 2002).

Table 5.4: Life history characteristics of selected species of skates and rays (Rajidae) that are caught by commercial trawlers in the North Sea. Estimates are for females (Walker and Hislop, 1998). *r* = rate of population increase.

SPECIES	LENGTH MAXIMUM (CM)	LENGTH MATURE (CM)	AGE MATURE (YR)	FECUNDITY (EGGS)	GROWTH RATE (*K*)	ESTIMATED MORTALITY	THEORETICAL REPLACEMENT MORTALITY
Common Skate *Dipturus batis*	237	160	11	40	0.06	na	0.38
Thornback Ray *Raja clavata*	118	86	10	140	0.14	0.60	0.52
Spotted Ray *R. montagui*	79	62	8	60	0.21	0.72	0.54
Cuckoo Ray *Leucoraja naevus*	75	56	8	90	0.23	0.69	0.58
Starry Ray *Amblyraja radiata*	71	39	5	38	0.22	0.79	0.73

A simple less data intensive method has been used to calculate the capacity of a population to withstand fishing mortality, based on length-based theory combined with life history invariants (Beverton, 1992; Dulvy *et al*., 2004; Pope *et al*., 2000). The method calculates the fishing mortality $F\varphi$, which will reduce the spawners produced per recruit (SSB/R) to a threshold or 'jeopardy' level, the proposed level is 5% of virgin SSB/R, but other thresholds could be considered, such as 50%, 30% or 10%, which are more consistent with IUCN decline criteria (Pope *et al*., 2000).

This method uses the ratio of length at maturity to maximum asymptotic length (L_α/L_∞, also known as the Beverton–Holt invariant h), the von Bertalanffy growth coefficient (k), natural mortality (m), the age at maturity (t_α) and the age at first capture (t_c) (Beverton, 1992; Beverton and Holt, 1957; Pope *et al*., 2000).

The parameter demand can be reduced to just the von Bertalanffy growth equation parameters (k, L_∞) and age at first capture t_c by using life history invariants to estimate the other parameters (Dulvy *et al*., 2004). Length at maturity L_α, natural mortality and age at maturity can be estimated from the von Bertalanffy growth equation parameters as follows:

$$h = L_\alpha/L_\infty = 0.73$$

$$m = 1.7k$$

$$t_\alpha = 1.7/m$$

(after Charnov, 1993; Frisk, Miller and Fogarty, 2001).

The basis of the method is as follows. The cumulative lifetime biomass, SSB of a year class of R recruits which recruit at age t_r, are captured at age t_c and mature at age t_α is:

$$SSB = R \cdot e\left(-M\left(t_c - t_r\right) - Z\left(t_\alpha - t_c\right)\right) \cdot a \cdot L_\infty^3 \cdot \sum_{i=0}^{3} U_i \left[\frac{\left(1-h\right)^i}{1-e\left(-Z-iK\right)}\right],$$

where M is natural mortality, $Z = F+M$, where the summation constant in the cubic expansion of the growth equation is $U_i = +1, -3, +3, -1$ for $n = 0, 1, 2, 3$, respectively; this is required to convert length into weight in order to calculate spawning stock biomass (SSB). This equation can be rearranged to find the fishing mortality, $F\varphi$, where φ represents the jeopardy threshold level of SSB/R, e.g., 5%, 10%, 30% or 50% is:

$$F(\varphi) = \frac{e\left(-F_\varphi\left(t_\alpha - t_c\right)\right)\sum_{i=0}^{3}\dfrac{\left(1-h\right)^i}{1-e\left(-Z-iK\right)}}{\sum_{i=0}^{3}\left[\dfrac{\left(1-h\right)^i}{1-e\left(-M-iK\right)}\right]}$$

At present the theory has only been developed for the situation where fish are caught prior to maturation $t_c > t_\alpha$ (Pope *et al*., 2000).

For non-target species or where data availability is limited, fishing mortality can be calculated using length cohort analysis (Jones, 1981) or swept area methods (Pope *et al.*, 2000).

A general exploration of the link between life histories, exploitation pattern and vulnerability ($F\varphi$) can be made by starting with a plausible range of growth rates and estimating the other parameters (h, M and t_α) using the life history invariant relationships described above. Species with slower body growth rates have lower natural mortality rates (Figure 5.16a) and correspondingly higher ages at maturity (Figure 5.16b). Species with faster growth rates have higher F jeopardy levels; i.e., they can withstand higher levels of fishing mortality before declining to the 5% level of SSB/R (Figure 5.16c). The later a fish is captured, with respect to its age of maturity, the greater the fishing mortality required to reduce the population to the jeopardy level of SSB/R. For fishes with fast growth rates, increasing the age of capture substantially increases the F required to drive it to the jeopardy level. Conversely, varying the age of capture has relatively little effect on F required to drive it to the jeopardy level for very slow growing fishes, such as elasmobranchs (Figure 5.16c).

This method was applied to the main elasmobranchs in the North Sea (Table 5.3). Von Bertalanffy growth parameters were compiled from the literature; female values were used except for the smoothhound, where the male growth parameters were more consistent with the maximum size observed in the North Sea (Table 5.5). Age at which 50% of the population were mature was derived from the literature, except for the blonde ray and lesser-spotted dogfish, where age at maturity was calculated from length at 50% maturity using the von Bertalanffy growth parameters. Age at capture was calculated from length at capture using the von Bertalanffy growth parameters (Table 5.5).

The jeopardy fishing mortality $F\varphi$, was calculated which would drive SSB/R to 10%, 30% and 50% of virgin levels. The sensitivity to uncertainty in size at first capture was explored by calculating $F\varphi$ for the observed age at first capture ± 20% in size.

There is a positive correlation between the life history, as measured by the growth k, and the jeopardy fishing mortality $F\varphi$, such that slow growing species can be depleted to 90% of virgin SSB/R at lower fishing mortalities, compared to fast growing species (Figure 5.17, Table 5.5). The largest, slowest growing species (tope, spurdog, smoothhound, thornback ray) can be driven to <90% of virgin SSB/R by fishing mortalities of <0.25, the smaller faster growing species (cuckoo ray, blonde ray, lesser-spotted dogfish, spotted ray, starry ray) can be driven to <90% of virgin SSB/R by fishing mortalities between 0.25–0.45 (Figure 5.17, Table 5.5). Error in the age at capture/age at maturity ratio has little effect on jeopardy fishing mortality $F\varphi$ of species caught well before maturation, and corresponding greater effect on species caught later in their juvenile life (spurdog, cuckoo ray and lesser spotted dogfish). The method is particularly sensitive to overestimation of age at first capture, and is insensitive to underestimation of age at first capture. Most species are caught long before maturation, (mean =13% of the age at maturity, range 0.8–34%) and delaying capture until the age at capture such that $t_c/t_\alpha = 0.5$ increases the $F\varphi$ 10% by approximately 72% (Table 5.5).

Higher fishing mortalities are needed to cause greater reductions in SSB/R levels (Figure 5.18). Fishing mortalities of $F<0.15$ are sufficient to reduce SSB/R to 50% of virgin levels across all of the shark and ray species considered here, $F<0.25$ is sufficient to reduce SSB/R to 30% of virgin levels and F<0.45 is required to reduce SSB/R to 10% of virgin levels (Figure 5.18).

Average demersal fishing mortality was as high as $F=0.7$ in the 1980s and early 1990s, but has declined over the last decade to approximately $F\sim0.5$ (Figure 5.19). Assuming that fishing mortality of non-target species is at least as high as that on target assessed demersal species, then such a fishing mortality regime would be sufficient to reduce the SBB/R of almost all sharks and rays to as low as 10% of virgin levels. It is usually assumed that non-target fishing

mortality is lower than target fishing mortality (Pope *et al.*, 2000). However, the converse assumption that fishing mortality of non-target species is at least as high as that of target assessed demersal species is not unreasonable for large slow growing species with relatively high catchabilities. Some skates and rays have been subject to fishing mortalities similar to the target species mortalities, particularly that of flatfishes, *F*=0.57–0.79 (Table 5.4) (Dulvy *et al.*, 2000; Walker and Hislop, 1998).

This relatively simple approach is theoretically grounded in well understood length-based and life history theory and requires relatively few assumptions and parameters (Beverton and Holt, 1957; Charnov, 1993; Dulvy *et al.*, 2004; Jones, 1981). The method is insensitive to most likely sampling bias, that of the underestimating the age at first maturity. These vulnerabilities (*Fφ*) appears to be consistent with the demersal fishing mortality rate, as evidenced by observed population declines and the general low abundance of sharks and rays, and is consistent with demographic models and life history theory (Walker and Heessen, 1996; Walker and Hislop, 1998).

Table 5.5: Length at first capture, growth parameters and corresponding age at first capture, and age at first capture / age at maturity (t_c/t_α) ratios. The left-hand ratio is 80% of the observed t_c/t_α ratio, the middle ratio is the observed t_c/t_α ratio and the right-hand ratio is 120% of the observed t_c/t_α ratio.

COMMON NAME	LENGTH AT 1ST CAPTURE	VON BERTALANFFY GROWTH PARAMETERS			AGE AT 1ST CAPTURE	80% T_c/T_α	T_c/T_α	120% T_c/T_α
		t_o	k	L_∞				
Blonde ray	25	-0.8	0.19	118.4	0.4	0.045	0.060	0.090
Cuckoo ray	29	-0.95	0.16	75.2	2.1	0.233	0.288	0.342
Spotted ray	10	-1.11	0.21	79.2	0.1	0.014	0.014	0.014
Starry ray	12	-1.05	0.22	67.4	0.2	0.019	0.038	0.075
Thornback ray	10	-0.88	0.14	118	0.1	0.011	0.011	0.011
Tope	53	-3	0.075	163	2.2	0.102	0.140	0.185
Smoothhound	26	-2.14	0.12	145	0.1	0.007	0.007	0.007
Lesser spotted dogfish	20	-0.01	0.20	88	1.3	0.224	0.265	0.286
Spurdog	47	-1.57	0.09	98.8	5.6	0.271	0.329	0.394

Table 5.6: Fishing mortality require to reduce SSB/R to a proportion of virgin level, Fφ for seven shark and rays species. Three SSB/R levels are used, 10%, 30% and 50% of virgin SSB/R.

SPECIES	VBGF GROWTH, K	Fφ 10%$_1$	Fφ 30%	Fφ 50%	Fφ 10%$_2$	PERCENT CHANGE Fφ 10%$_{1-2}$
		t_c/t_α = as observed			t_c/t_α = 0.5	
Tope	0.075	0.14	0.08	0.04	0.22	58
Spurdog	0.09	0.21	0.10	0.06	0.26	26
Smoothhound	0.12	0.20	0.10	0.06	0.36	81
Thornback ray	0.14	0.24	0.12	0.06	0.42	79
Cuckoo ray	0.16	0.36	0.18	0.10	0.48	33
Blonde ray	0.19	0.34	0.18	0.10	0.6	77
Lesser-spotted dogfish	0.20	0.45	0.22	0.12	0.62	39
Spotted ray	0.21	0.36	0.18	0.10	0.66	83
Starry ray	0.22	0.17	0.08	0.06	0.5	39

5.5   Future studies

Research vessel data can be used to determine the gear-specific catchability for some of these species for beam and 'otter trawl' gears, though studies to examine the different catchabilities of various trawl, long line and gill nets are lacking. The approaches used by Huse *et al.* (1999, 2000) for cod, haddock and Greenland halibut could be usefully employed for elasmobranchs.

Further work to determine what mortality rates that can be experienced by the populations of elasmobranchs is required, and it will also be important to evaluate which life-history stages (age or length classes) are the most important for population growth rates? Demographic models that explore the impacts of F on length/age classes may help address this. The simple method outlined should be compared with estimates from age- or stage-structured demographic models.

Furthermore, better estimates between the overlap between the spatial and temporal distribution of fishing effort (by gear) with fish distribution are required, particularly for inshore net and line fisheries that target elasmobranchs. For the various metiers operating, better estimates of which elasmobranchs species are caught (by sex and size), the proportion that are caught (i.e., what is the relative catchability and selectivity of the various gears), which are discarded/retained, and what is the survivorship of discards are required. In terms of catchability, earlier studies have suggested that the catchability of skates is similar to that of flatfish (0.044–0.10), and that spurdog are similar to demersal gadoids (0.36–0.56) (Harley *et al.*, 2001), though better data are required. Similarly, studies on the survivorship of trawl-caught elasmobranchs are limited (e.g., Kaiser and Spencer, 1995; Revill *et al.*, 2005) and survivorship from other gears lacking.

Further studies on this TOR will be conducted at WGEF in June 2005.

5.6   References

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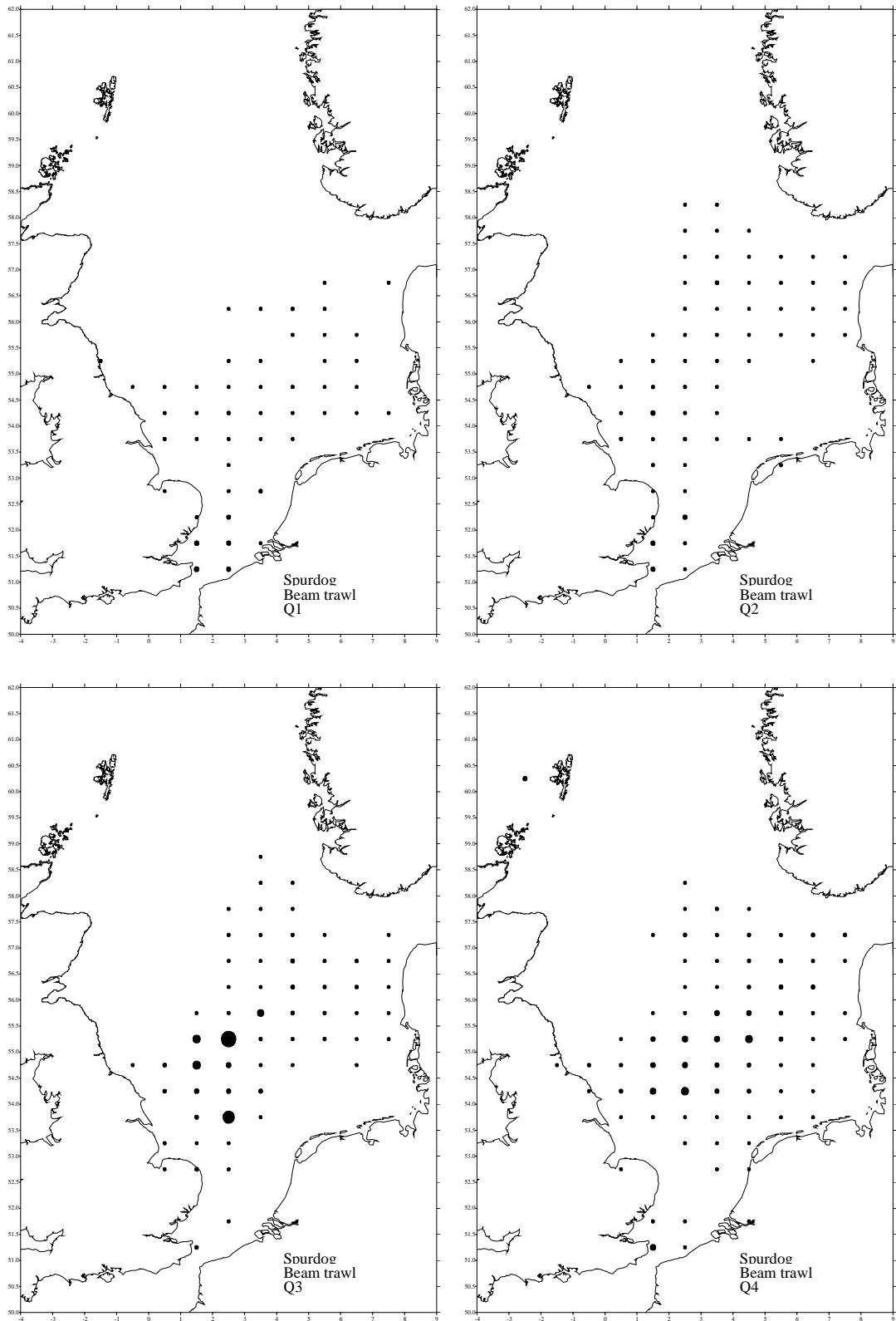
Figure 5.1: Total landings (UK and non-UK vessels landing into the UK, 1995–2004) of spurdog by beam trawl and quarter (largest symbol = 3.95 tonnes).
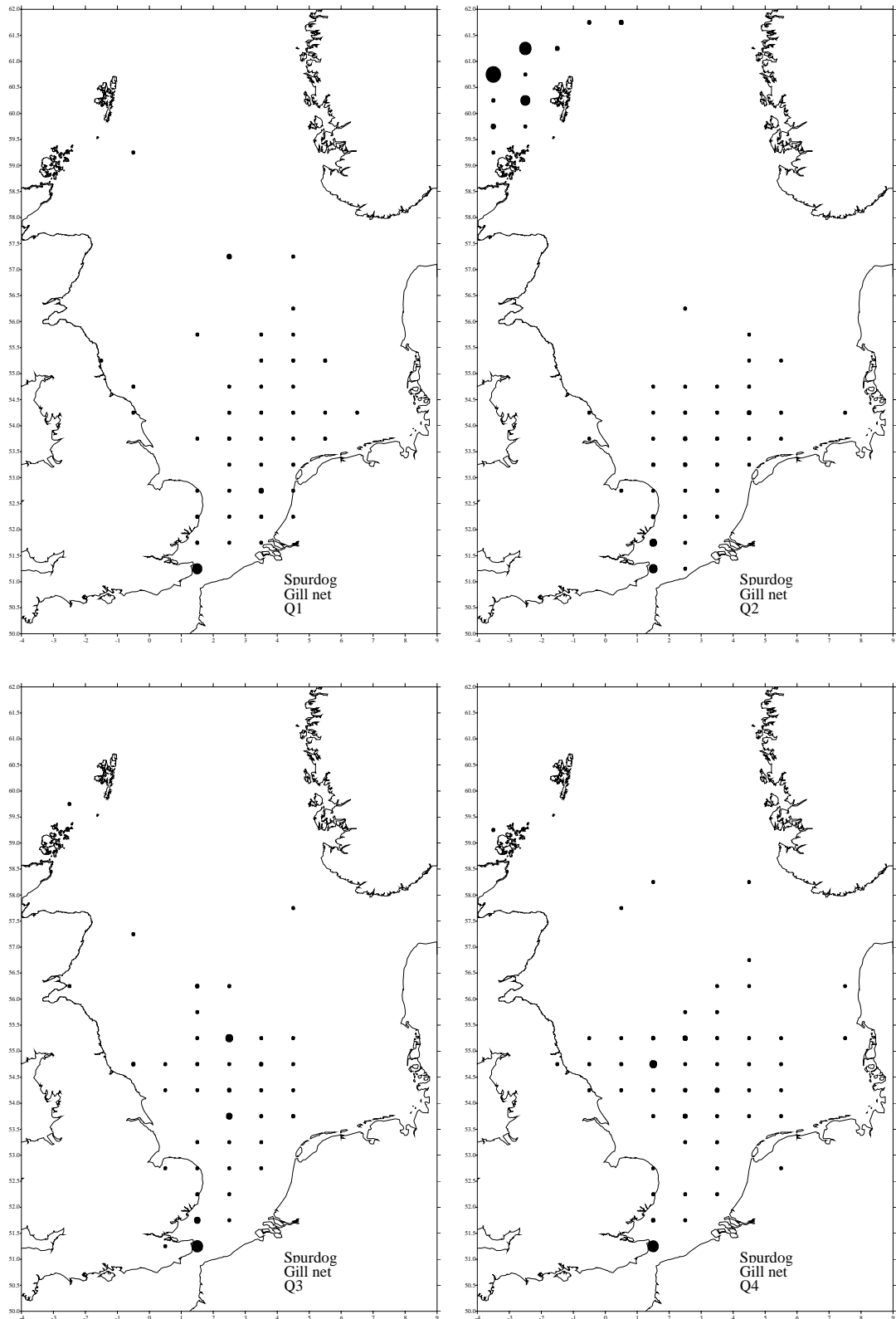
Figure 5.2: Total landings (UK and non-UK vessels landing into the UK, 1995–2004) of spurdog by gill net and quarter (largest symbol = 15.9 tonnes).
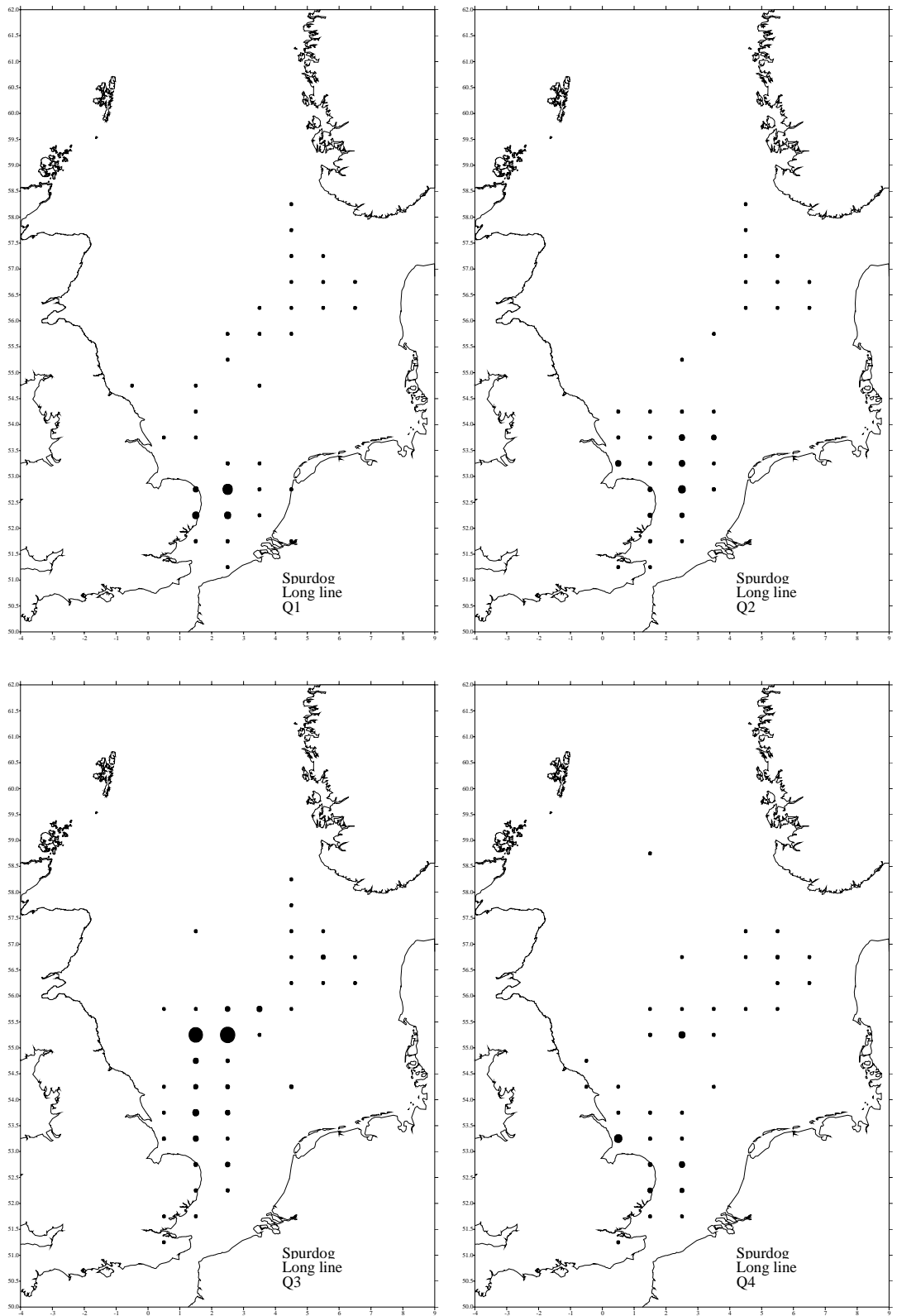
Figure 5.3: Total landings (UK and non-UK vessels landing into the UK, 1995–2004) of spurdog by long line and quarter (largest symbol = 225.5 tonnes).
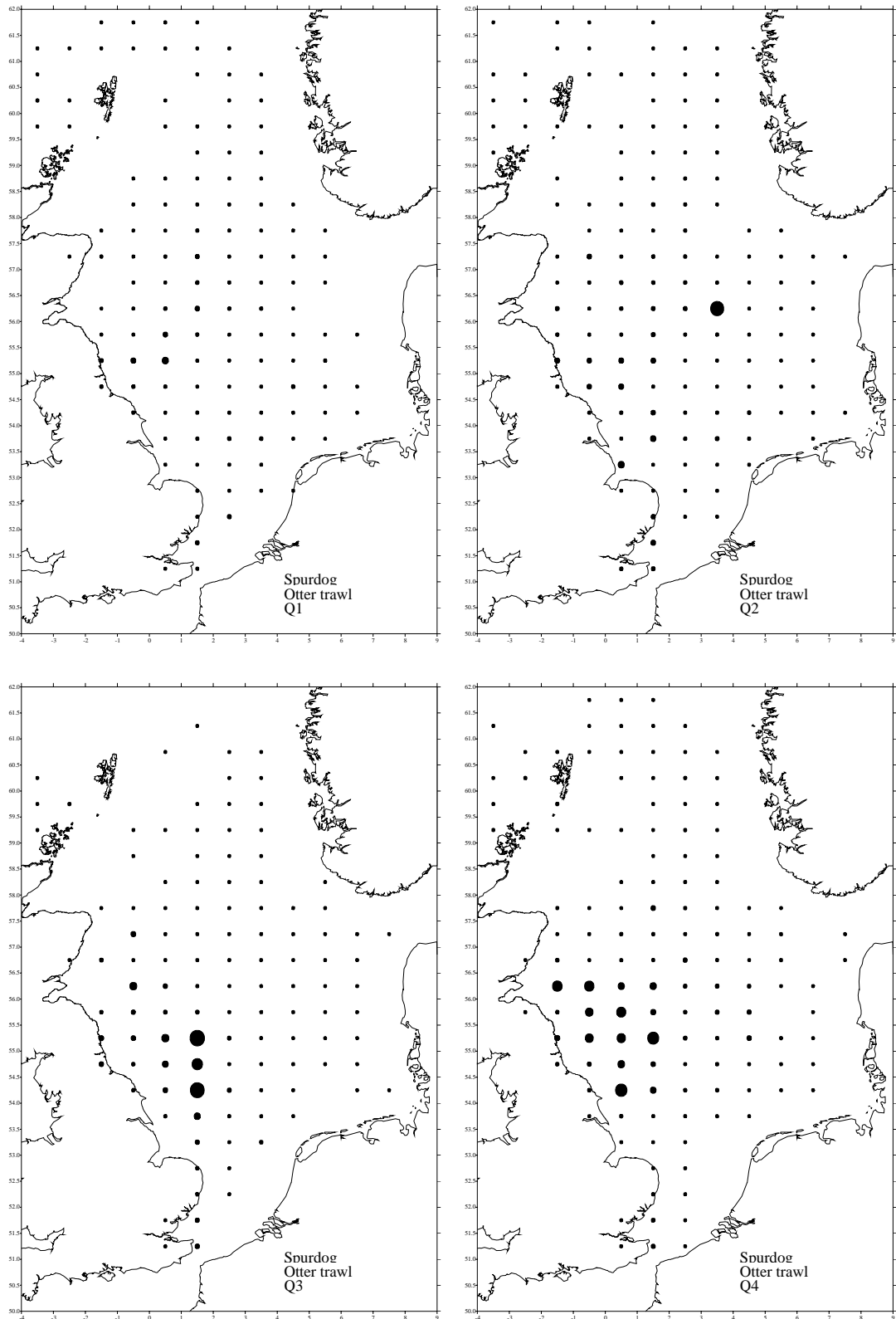
Figure 5.4: Total landings (UK and non-UK vessels landing into the UK, 1995–2004) of spurdog by otter trawl and quarter (largest symbol = 51.0 tonnes).
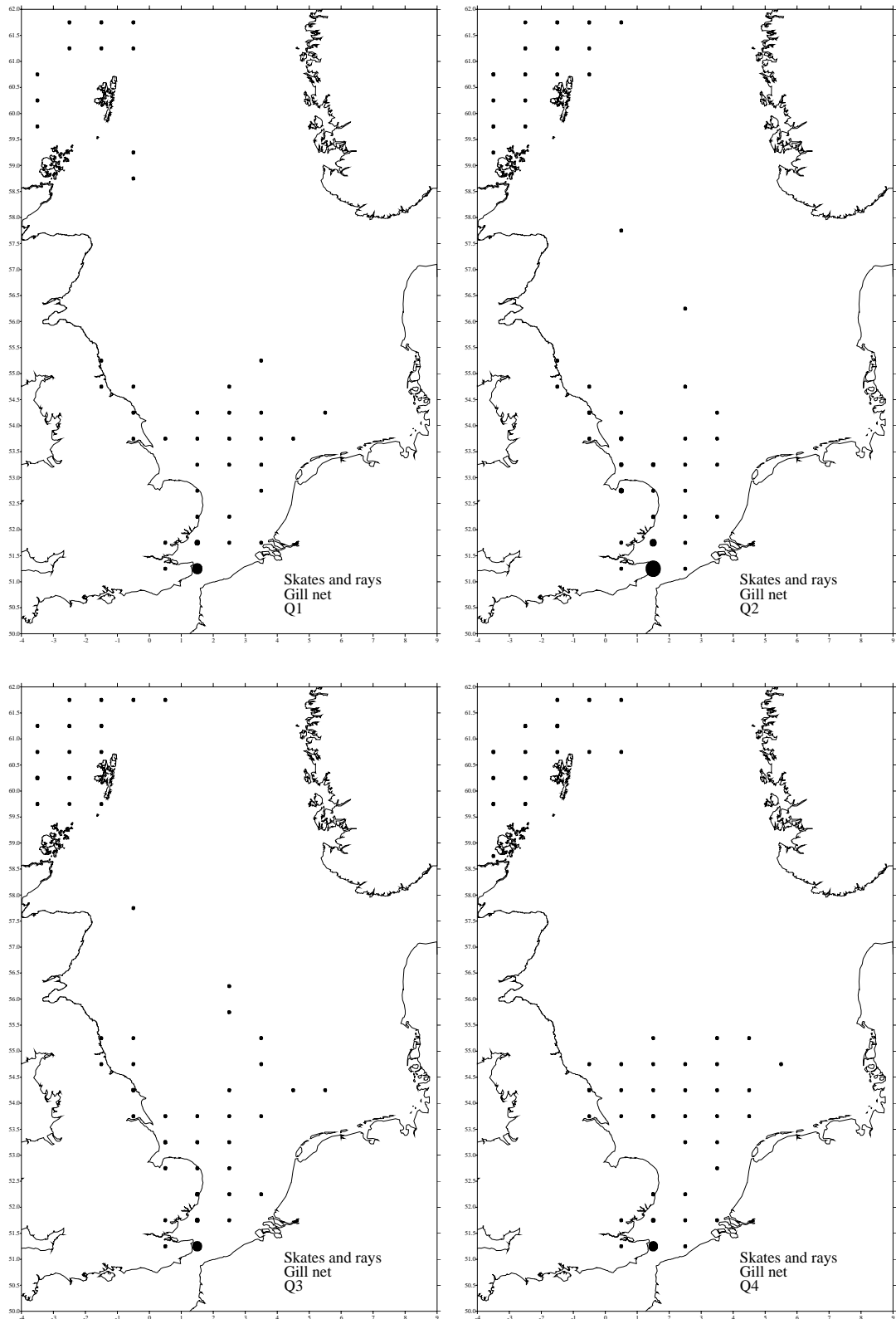
Figure 5.5: Total landings (UK and non-UK vessels landing into the UK, 1995–2004) of skates and rays by gill net and quarter (largest symbol = 316.7 tonnes).
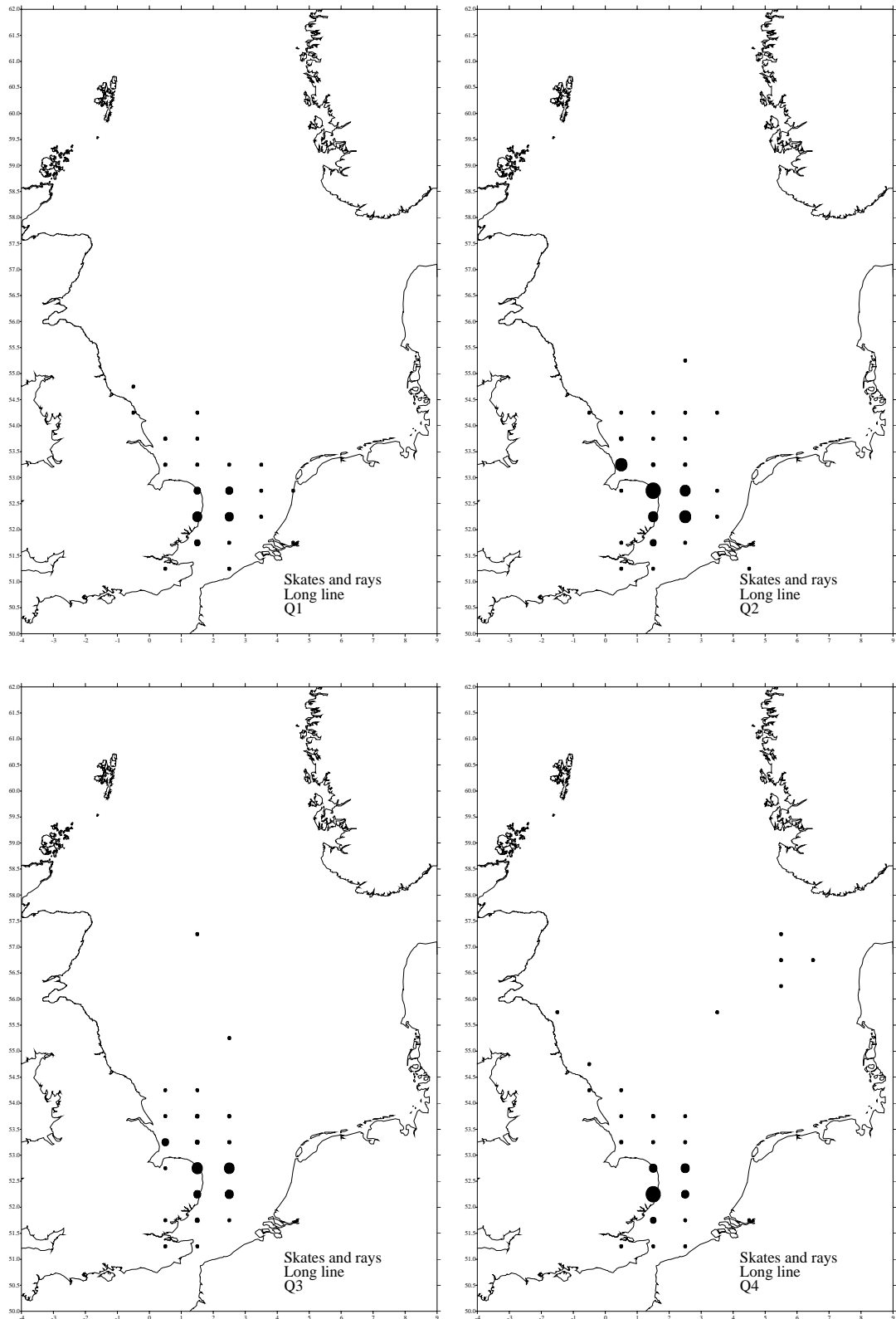
Figure 5.6: Total landings (UK and non-UK vessels landing into the UK, 1995–2004) of skates and rays by long line and quarter (largest symbol = 211.7 tonnes).
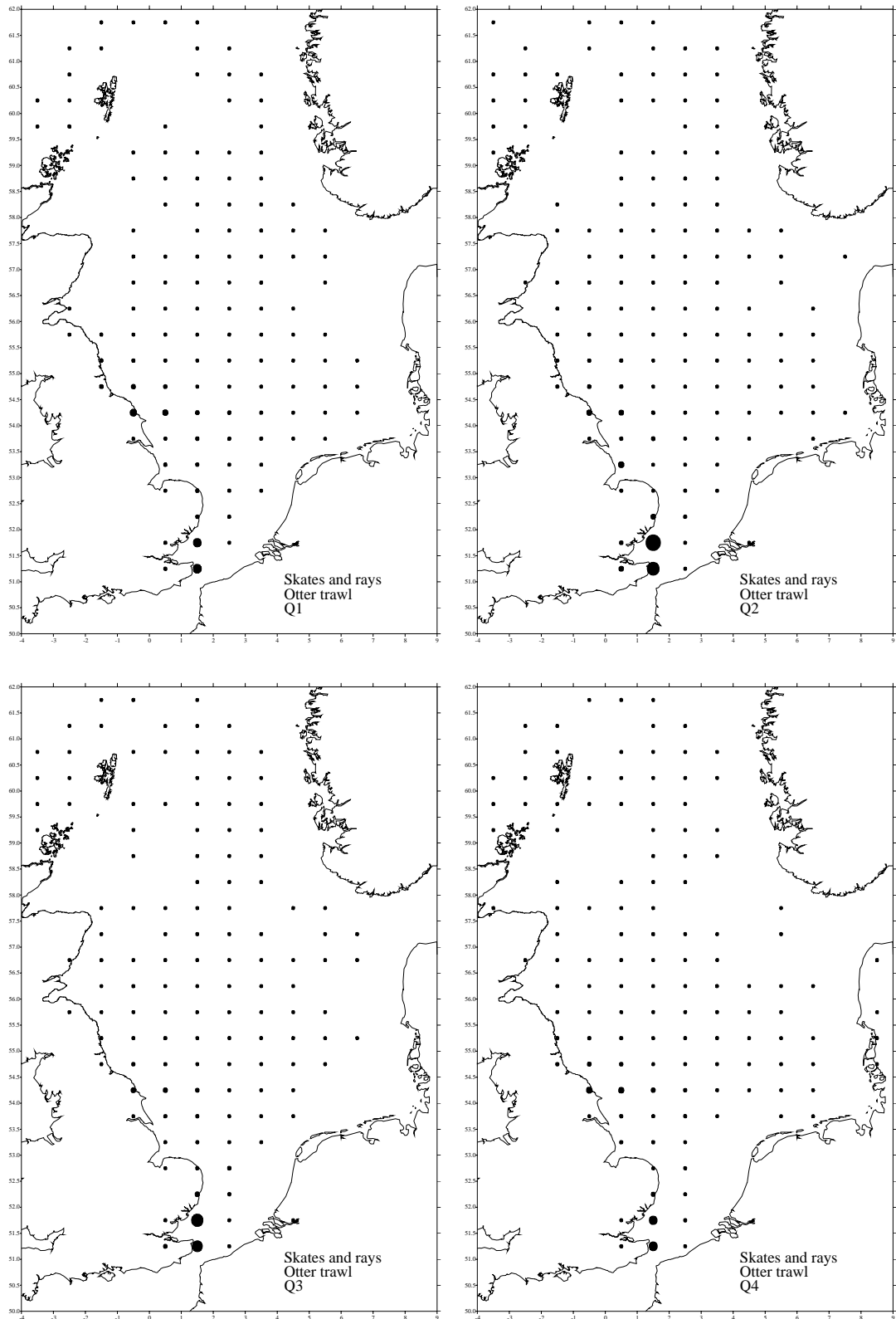
Figure 5.7: Total landings (UK and non-UK vessels landing into the UK, 1995–2004) of skates and rays by otter trawl and quarter (largest symbol = 307.5 tonnes).
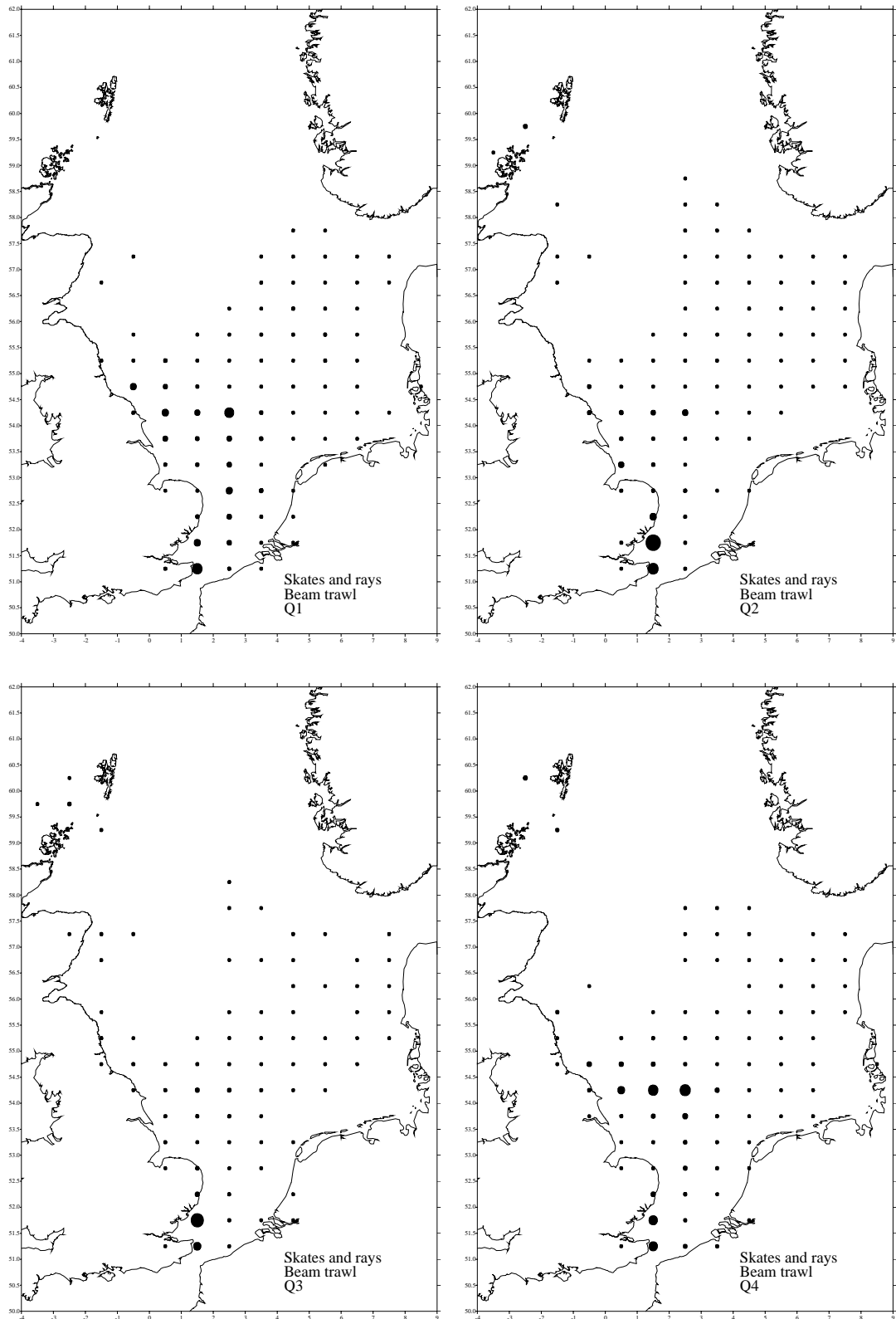
Figure 5.8: Total landings (UK and non-UK vessels landing into the UK, 1995–2004) of skates and rays by beam trawl and quarter (largest symbol = 44.4 tonnes).
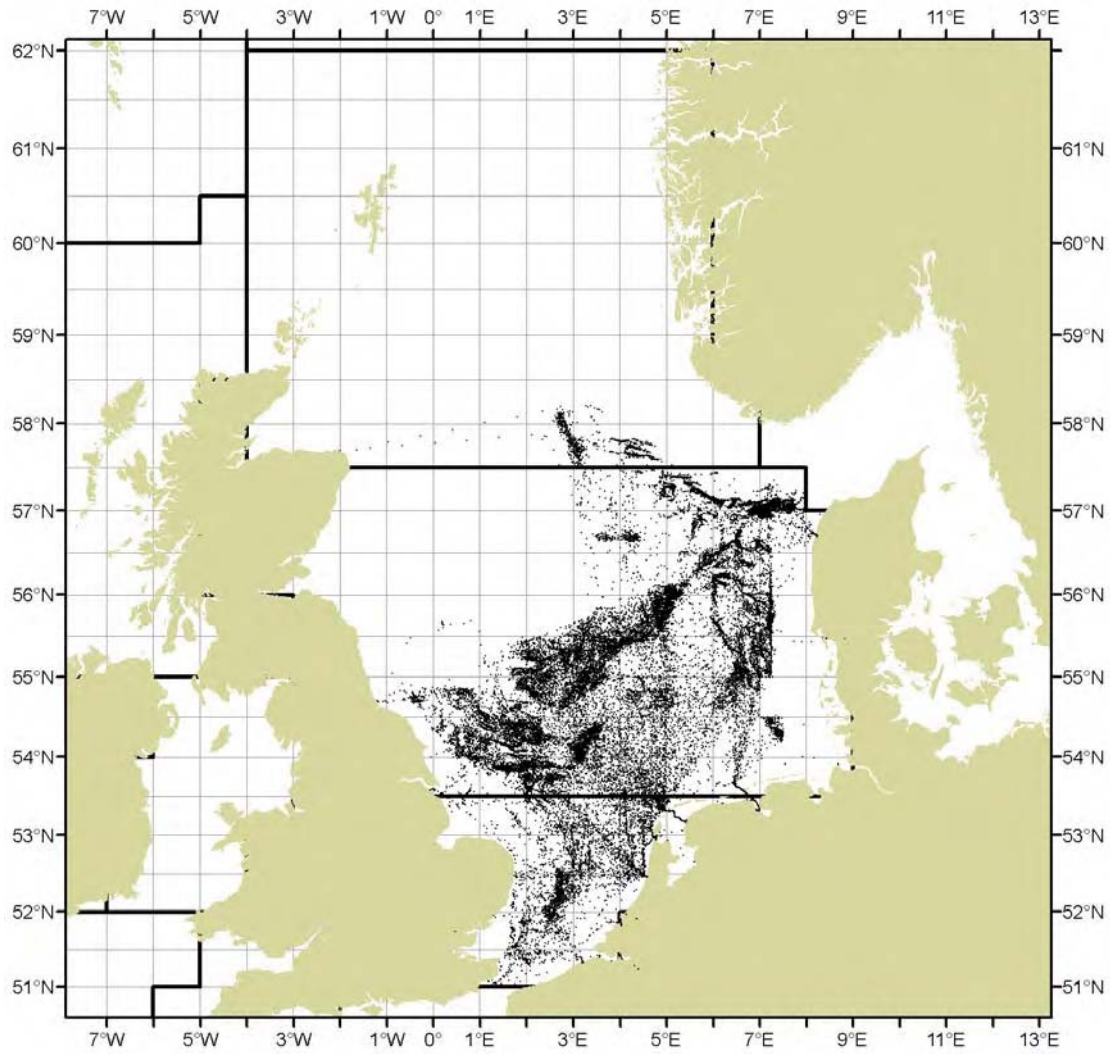
Figure 5.9: Distribution of UK beam trawl fleet based on two hourly VMS locations in 2003.
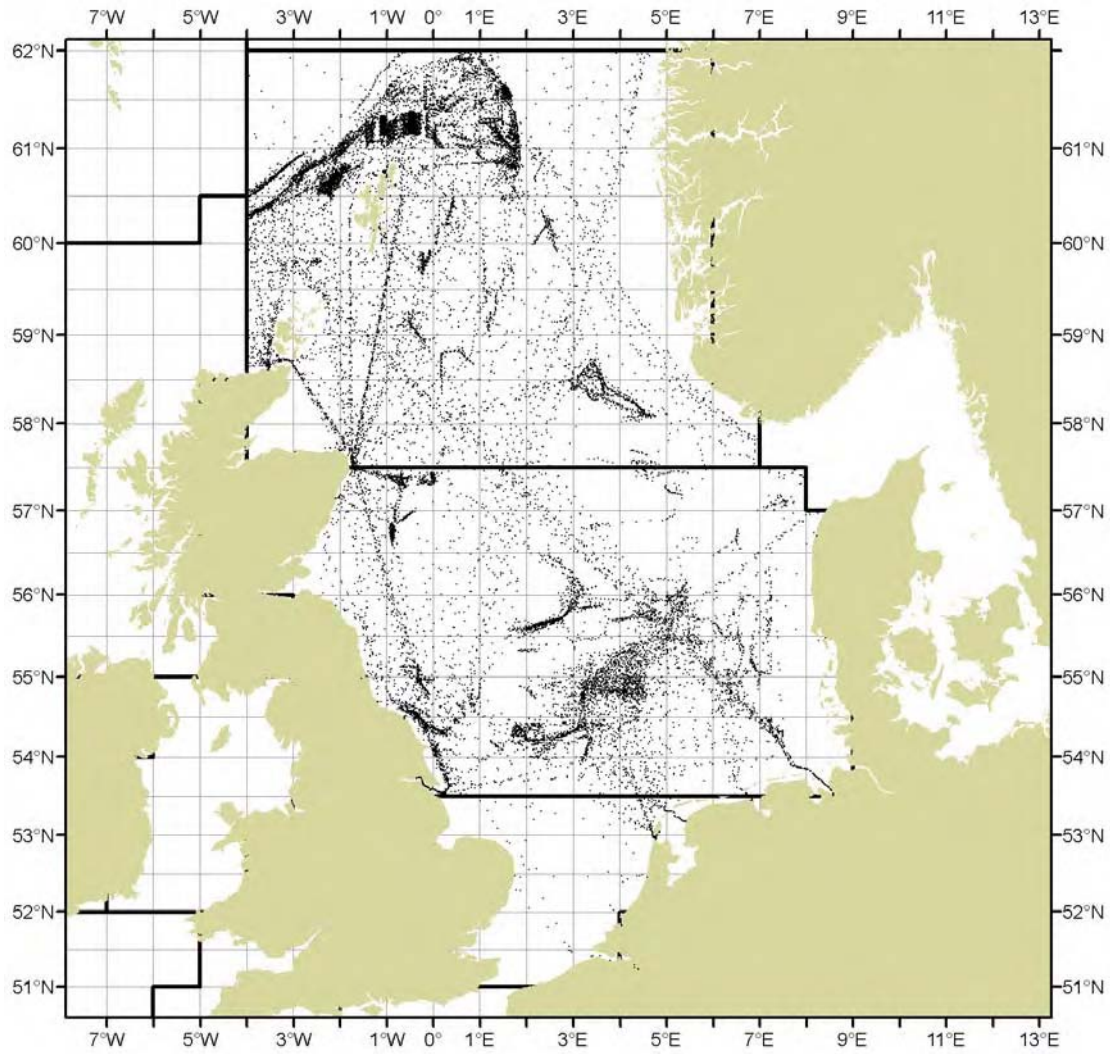
Figure 5.10: Distribution of UK trawler fleet based on two hourly VMS locations in 2003. Gear types include 'Bottom Pair Trawl', 'Heavy Otter Trawl', 'Light Otter Trawl', 'Otter Trawl (un-specified)' and 'Twin *Nephrops* trawl')
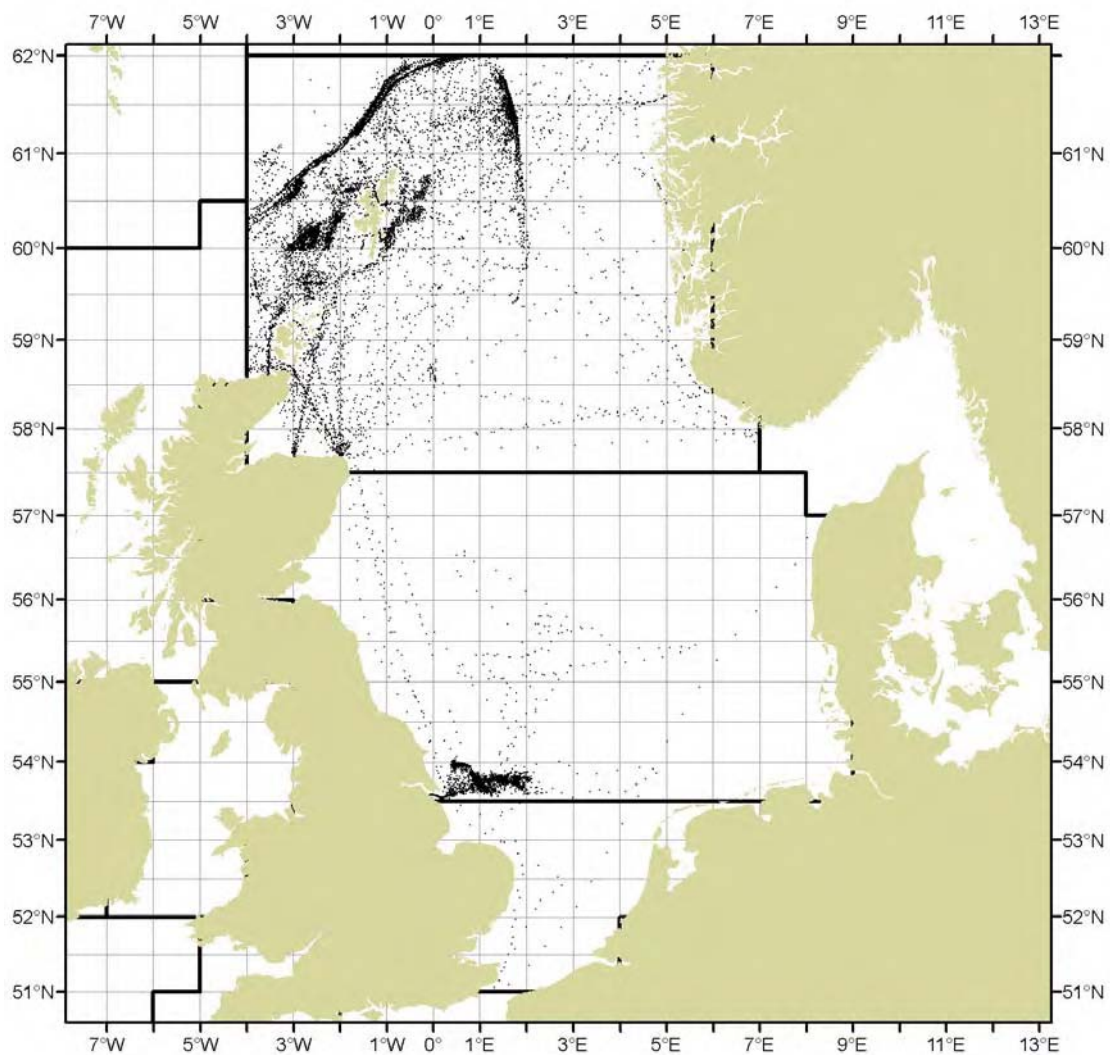
Figure 5.11: Distribution of UK fixed gear fleet based on two hourly VMS locations. Gear types include 'Gill Net (unspecified)','Gill Net', 'Long Lines', 'Other Pots or Mixed', 'Top-opening Pots', 'Tangle Net', 'Trammel Net', 'Purse Seine'
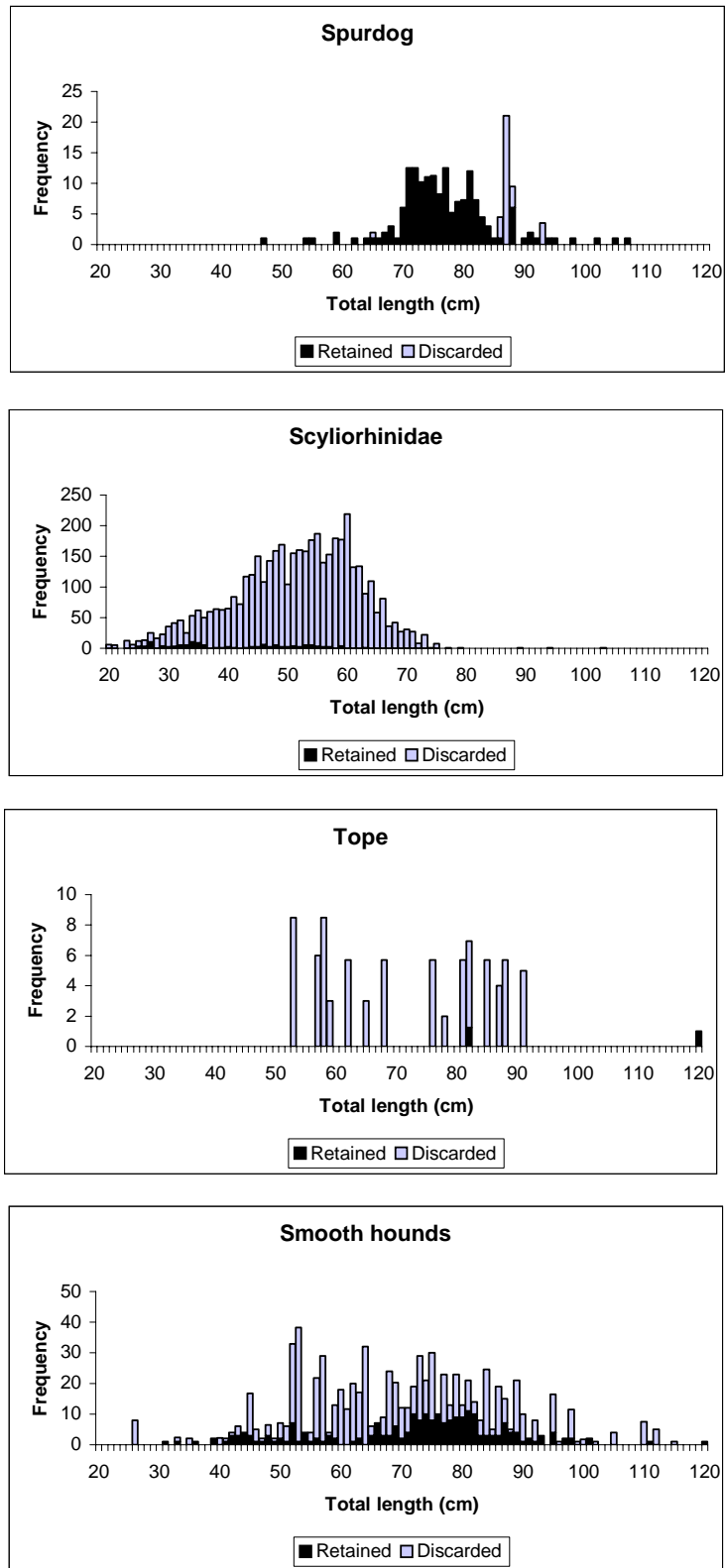
Figure 5.12: Size distribution of spurdog, scyliorhinid catsharks, tope and smooth hounds (*Mustelus* spp.) that were retained or discarded in the North Sea (all gears combined, 2002–2004).
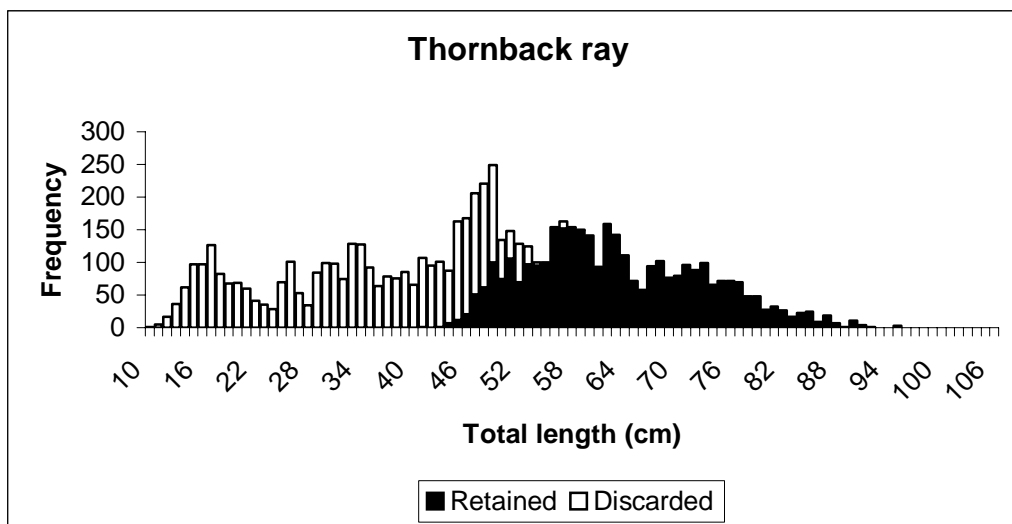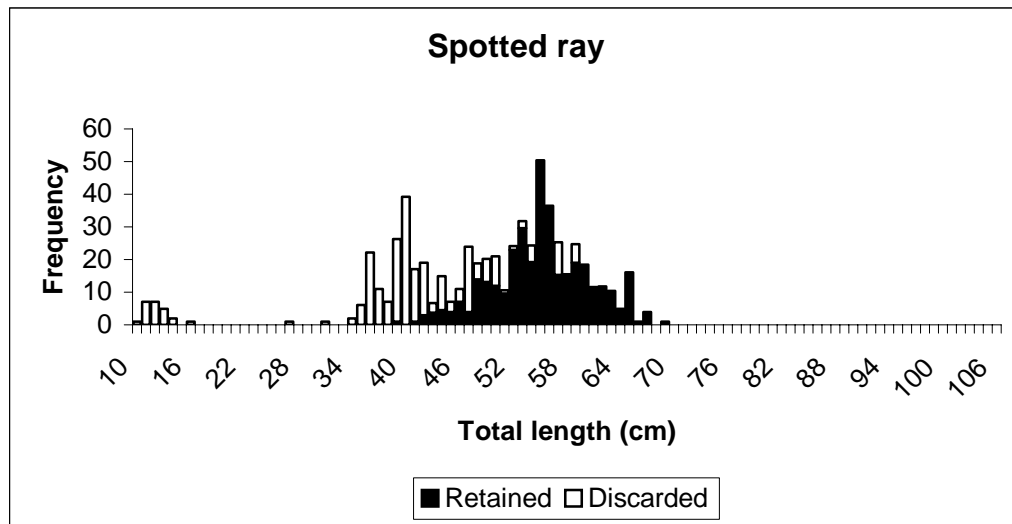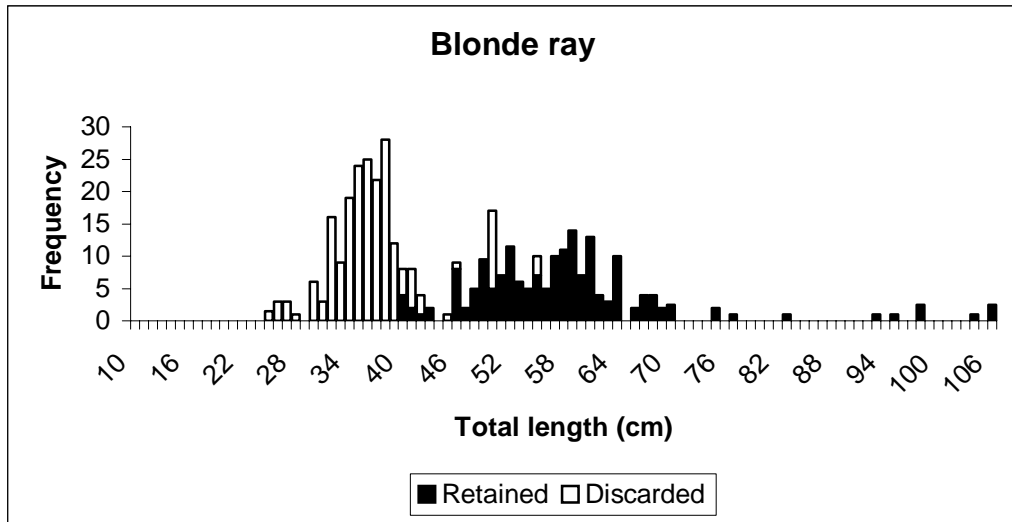
Figure 5.13: Size distribution of blonde, spotted and thornback ray that were retained or discarded in the North Sea (all gears combined, 2002–2004).
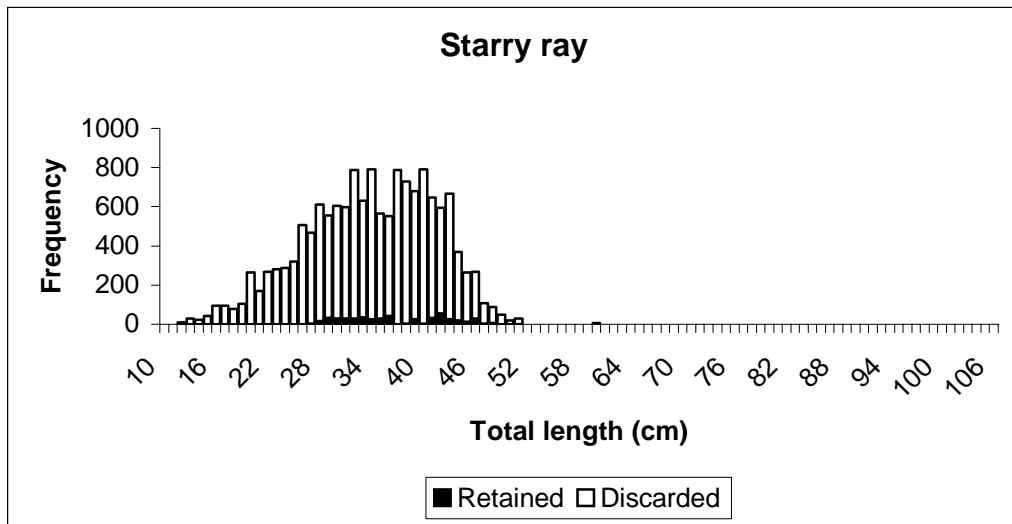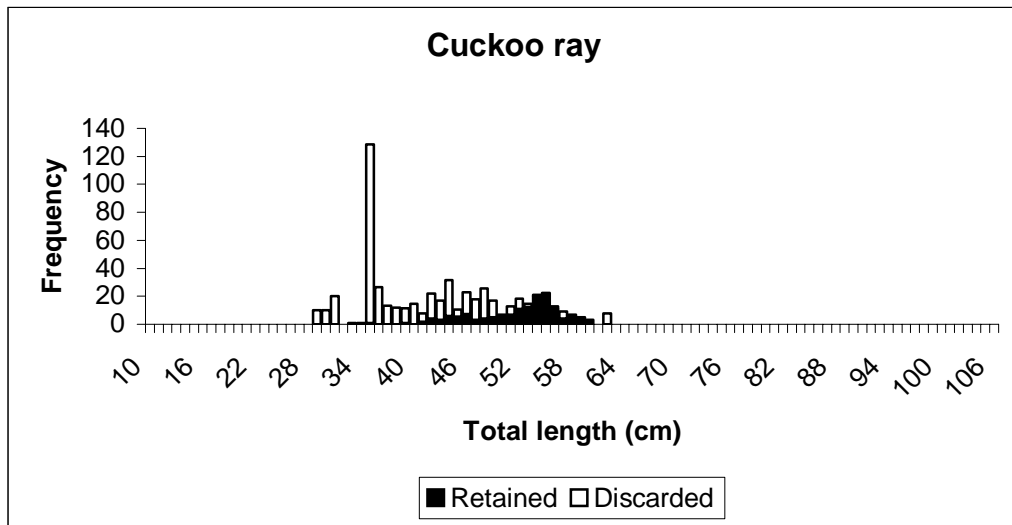
Fiure 5.14: Size distribution of starry and cuckoo ray that were retained or discarded in the North Sea (all gears combined, 2002–2004).
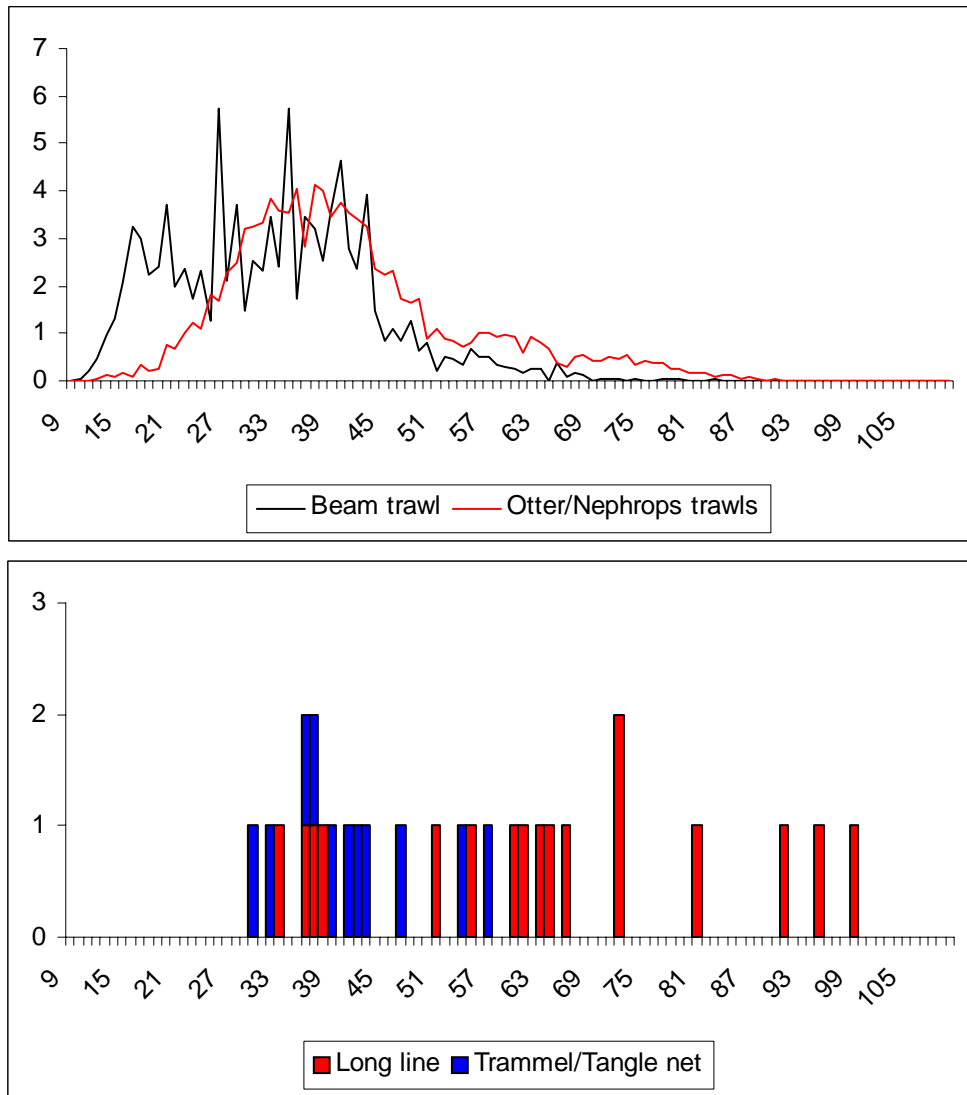
Figure 5.15: Size distributions of rays (all species combined) observed in commercial beam and otter trawl (top) and long line and gill net (bottom) fisheries.

Figure 5.16: The relationship between life history, exploitation pattern and *F(φ)*, the fishing mortality required to reduce SSB/R to 5% of unexploited levels. (a) The relationship between the von Bertalanffy growth rate *k,* and the natural mortality rate, assuming the invariant relationship *M* / *k* = 1.7. (b) The relationship between growth rate *k,* and the age of maturity *α,* assuming the invariant relationship α*M* ~ 2. (c) The relationship between growth rate *k* and the fishing mortality *Fφ,* which will reduce the spawners per recruit (SSB/R) to below 5%, assuming an invariant relationship between length at maturity L_α, and asymptotic length L_∞, L_α / L_∞ = 0.66. The isopleths represent the age at capture t_c expressed as a proportion of age of maturity t_α. Three ratios are plotted here representing ages of capture t_c at 10%, 50% and 90% of the age at maturity t_α.

Figure 5.17: The relationship between growth rate *k* and the fishing mortality *Fφ*, which will reduce spawners per recruit (SSB/R) to below 10% for seven shark and ray species (thick vertical lines represent t_c / $t_\alpha \pm$ 20%). The isopleths represent the age at capture t_c expressed as a proportion of age of maturity t_α. Three ratios are plotted here representing ages of capture t_c at 1%, 10%, 50% and 100% of the age at maturity t_α. The lowest isopleth t_c / t_α = 1% and the highest = 100%.
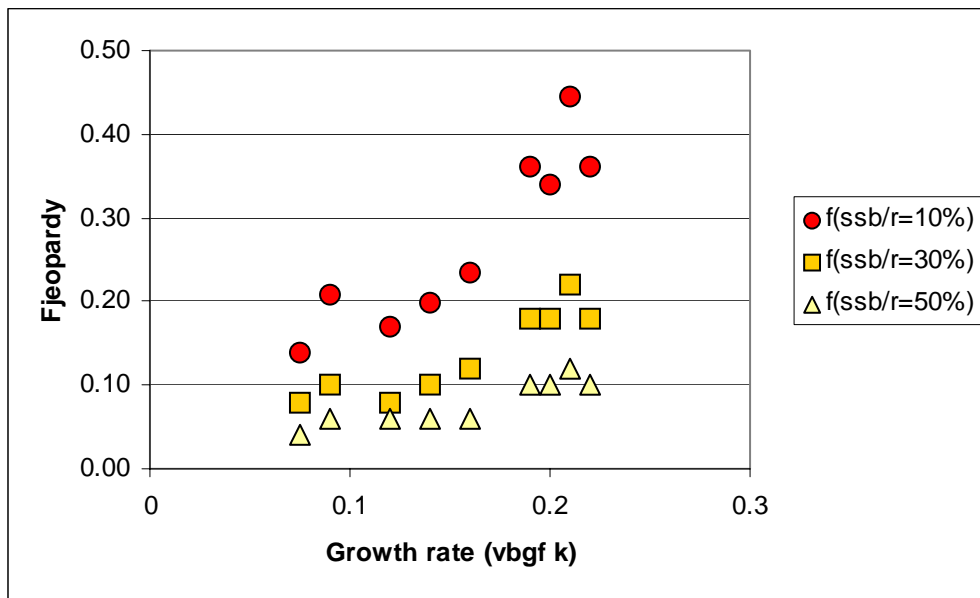
Figure 5.18: The relationship between growth rate *k* and the fishing mortality *Fφ*, which will reduce spawners per recruit (SSB/R) to below 10%, 30% and 50% for seven shark and ray species based on observed age at first capture.
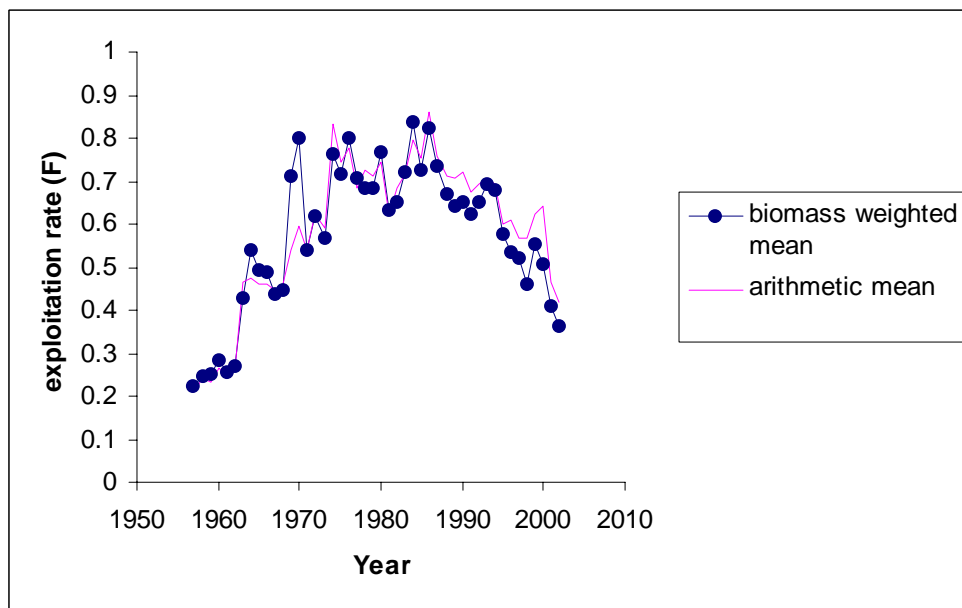
Figure 5.19: Average demersal exploitation rate, expressed as either the arithmetic mean of fishing mortality (*F*), or the mean species *F* weighted by each species biomass of across each of the seven assessed North Sea stocks in each year (Daan *et al.*, 2003).

6   Relative catchabilities of fishes

6.1   Introduction

The probability of being captured by a sampling gear varies between species and with the size of individuals within species. Moreover, corresponding species-size specific catch probabilities are different for different sampling gears. If data from different sampling gears are pooled and used to obtain estimates of relative abundances, species compositions, size compositions etc., it is therefore necessary to standardise data to make density measurements from different gears more comparable (if not the estimates may become biased). For the same reason, it is also important to explore potential catchability changes when gears or vessels are changed over time, and both old and new data are used in time series. Hence, standardisation requires estimates of relative catchabilities. Relative catchability (or capture efficiency) between two sampling gears can be defined as

$$R_{ij} = \frac{\text{absolute capture efficiency of gear } i}{\text{absolute capture efficiency of gear } j} \tag{6.1}$$

where R_{ij} is the capture efficiency of gear i relative to gear j (for a given species-size class). If relative capture efficiency is known, standardisation can be done by using it as a conversion factor and adjust catch rates (cpue) from gear i relative to gear j:

$$cpue_i^* = \frac{cpue_i}{R_{ij}} \tag{6.2}$$

where $cpue_i^*$ is the adjusted/standardised cpue from gear i. Different methods exist to obtain data for estimation of relative catchabilities. The preferred method is inter-calibration experiments where two different sampling gears are operated close in time and space, e.g., many parallel trawl tows for two different trawl types (Wilderbuer *et al.*, 1998). It is then assumed that the two gear types experience equal densities of the target species/group and equal environmental conditions during a comparison. Each of these paired comparisons then gives a single observation (r_{ij}) of relative capture efficiency between the two gears (also termed a cpue-ratio):

$$r_{ij} = \frac{cpue_i}{cpue_j} \text{ for } cpue > 0 \tag{6.3}$$

The estimate of relative capture efficiency ($\hat{R}_{ij}$) based on many pared comparisons can for example be obtained by taking the median of the r_{ij}-values (Salthaug 2002). An argument for using the median is that this estimator will not be heavily affected by extreme outliers (which are quite common in inter-calibration experiments). It should be noted that the arithmetic average of the r_{ij}-values will give a biased estimate because of the skewed distribution of ratios and must therefore not be used without an appropriate transformation of the r_{ij}-values first (e.g., log).

Inter-calibration experiments specifically designed to explore relative catchabilities are preferred, but are expensive and such data are therefore lacking for most gears and species. An alternative method is then to use traditional survey data and treat these as inter-calibration experiments, but this requires that surveys with the two gears overlap in time and space. Cpue-ratios are then calculated each time the different gears have operated close in time and space (a maximum allowed difference in time and distance has to be specified). If the data are still too poor to obtain enough paired comparisons, gear-specific abundance indices from the corresponding area and time period can be used to estimate relative capture efficiency:

$$\hat{R}_{ij} = \frac{index_i}{index_j} \qquad\qquad\qquad\qquad (6.4)$$

A major disadvantage with this method compared with the "paired comparison approach" is that factors other than the gear effect will also influence the estimate of relative catchability (e.g., the spatial structure of the surveyed species relative to the survey coverage, variations in environmental conditions). Though not treated here, different methods exist to assess uncertainty in estimates of relative catchabilities (e.g., Wilderbuer *et al.*, 1998; Salthaug 2002). Potential factors causing differences in catchability between trawl gears were discussed in ICES (2004), and a more general review for various types of gears is given by Fernö and Olsen (1994).

6.2   Case study I: Size specific catchabilities of 8m-beam trawl relative to GOV trawl

Marked differences in the capture efficiency between beam trawl and GOV trawl were found for various fish species (ICES, 2004). It was recommended to conduct further studies on species and size-dependent catchabilities in order to establish conversion factors between the two gear types (where this is possible). In this case study, size-specific capture efficiencies of 8m-beam trawl relative to GOV trawl were calculated for *Amblyraja radiata* and 'all rajids' using the methodology described above.

The IBTS 3rd Quarter cruises, where a GOV trawl is used, partly overlap in time and space with the Dutch 8m-beam trawl survey during the period 1998–2004 (though 2000 and 2001 were excluded because of problems with the data). These data were treated as an intercalibration experiment for each 10-cm length group: when both beam trawl and GOV trawl had been recorded in the same ICES-rectangle and year (termed a year-square cell) a cpue-ratio was calculated relative to GOV (i.e., with cpue from GOV as the denominator in Equation 6.3). Cpue was measured as numbers caught per trawling hour. If more than one trawl haul with the same gear type had been recorded in the same year-square cell the average cpue of these was calculated (including zero-catches) and treated as a single haul. If one or both of the two gear types still had zero-catches in the length group, these comparisons were excluded as they were considered to give limited quantitative information about relative capture efficiency. The final estimate of relative capture efficiency within 10-cm length groups was calculated as the median of the cpue-ratios (Table 6.1).

Table 6.1: Estimated capture efficiency of 8m-beam trawl relative to GOV trawl for different length groups

LENGTH GROUP	CAPTURE EFFICIENCY	
	A. radiata	All rajids
0 – 9 cm	3.0	3.0
10 – 19 cm	5.0	5.0
20 – 29 cm	6.5	6.5
30 – 39 cm	4.0	4.0
40 – 49 cm	2.07	2.0
50 – 59 cm	2.25	2.0

6.3 Case study II: Size specific catchabilities of 4m-beam trawl relative to GOV trawl for *Raja clavata*

The IBTS 3rd Quarter cruise, where a GOV trawl is used, partly overlap in time and space with the English Channel / southern North Sea 4m-beam trawl survey during the period 1992–2003. The resolution of available survey data was too low to obtain paired comparisons. Thus, abundance indices (for the entire time period) for each gear type and length group (10 cm) were calculated for *Raja clavata* and Equation 6.4 was used to estimate size specific catchabilities of 4m-beam trawl relative to GOV. Only observations from ICES Subarea IVc, where the two gear types had some degree of overlapping coverage were used. Figure 6.1 shows the gear-specific abundance indices for different length groups of *Raja clavata*. The abundance indices and the resulting estimates of relative capture efficiencies (Figure 6.2) indicated that the 4m-beam trawl was more efficient for smaller lengths and that the GOV trawl was slightly more efficient for larger lengths.
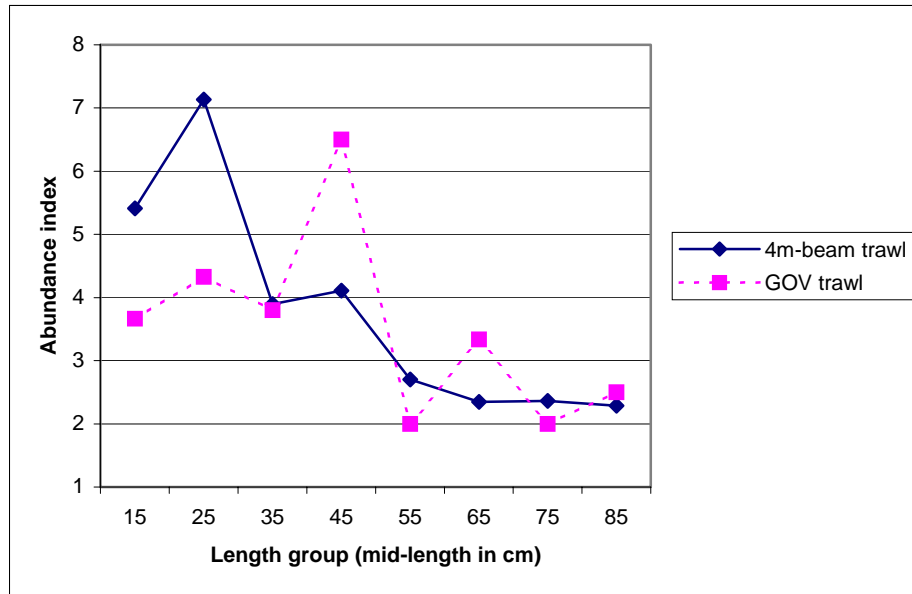
Figure 6.1: Abundance indices per 10 cm length groups of *Raja clavata* (ICES Subarea IVc and time period 1992–2003).
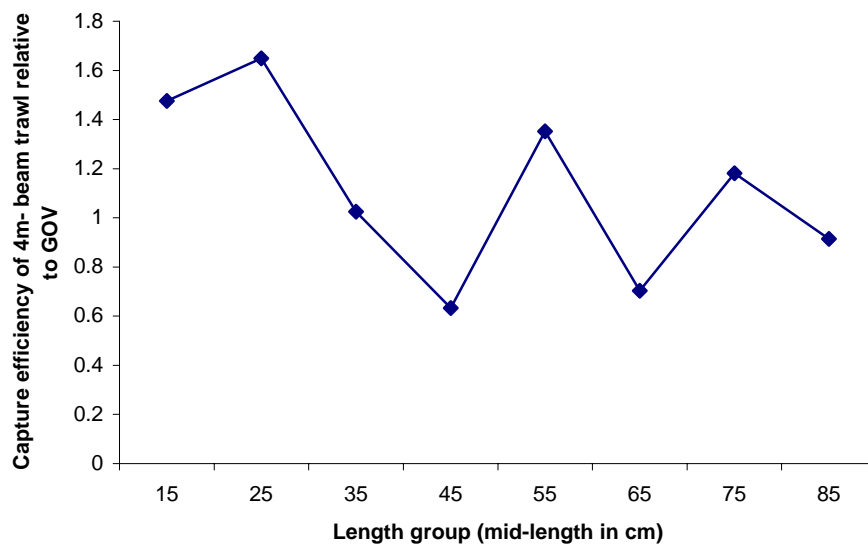
Figure 6.2: Estimated capture efficiency per length group of 4m-beam trawl relative to GOV trawl for *Raja clavata*.

6.4 Case study III: Sensitivity of size spectrum slope to changes in gear, vessel

All fishing gear catches species with different degrees of selectivity (Harley and Myers 2001). In addition, when gear or vessel changes occur there will almost certainly be differences in survey at the change period. Most surveys that have existed for 20 or more years have changed gear and vessel at least once. This is a result of the limited lifespan of most research vessels, and the need to change gear may be due to modifications in data requirements for fisheries management or for improving consistency and comparability with other surveys. Sometimes these surveys have correction factors for certain species that may work quite well (e.g., English Groundfish Survey corrections for commercial species for the Granton – GOV conversion) but the situation is much more difficult when looking for consistency over all

species for community indicators. This problem can be most marked when examining a size-based indicator such as size spectrum slope, where gear changes have altered catchability by size of species.

We have made a brief analysis of abundance and biomass size spectrum slope for all species in the Northern Gulf of St. Lawrence (Canada) survey. The Northern Gulf of St. Lawrence summer survey (1984-present) trawling gear and vessel were changed in 1990 and again in 2004. From 1984 to 1990, the survey was conducted on the *Lady Hammond* with a Western IIA trawl with 19 mm codend liner, and from 1990–2003, the URI 81/114 trawl with 19 mm codend liner was fished from the CCGS *Alfred Needler*. In 1990 comparative tows between vessel and gear were made to enable corrections to be calculated to maintain the consistency of the time series.

The Western IIA trawl used in this survey is essentially a commercial fishing otter trawl and is known to be a fairly good sampler of fish in commercial size ranges though not as good at sampling smaller fish and invertebrates such as shrimp (*Pandalus* spp.). As shrimp became an increasingly important species, both commercially and by biomass, in the Northern Gulf throughout the 1980s and 90s a decision was made to switch gear to a fine mesh shrimp trawl - URI 81/114. This latter gear though good at catching small fish and shrimp is not good at catching large fish and also is much more vulnerable to tearing on rocky or jagged bottoms than the Western IIA trawl. This has prevented the URI trawl from fishing certain areas that are thought to harbour not insignificant numbers of commercial sized cod.

Here we have plotted size spectra (Figure 6.3) and fitted slopes linear size spectrum slope for the 1990 comparative year (Table 6.2). Though correction factors have been developed we have not examined them here. Only fish larger than 15 cm have been included and all invertebrates except squids have been excluded from the data before analysis.
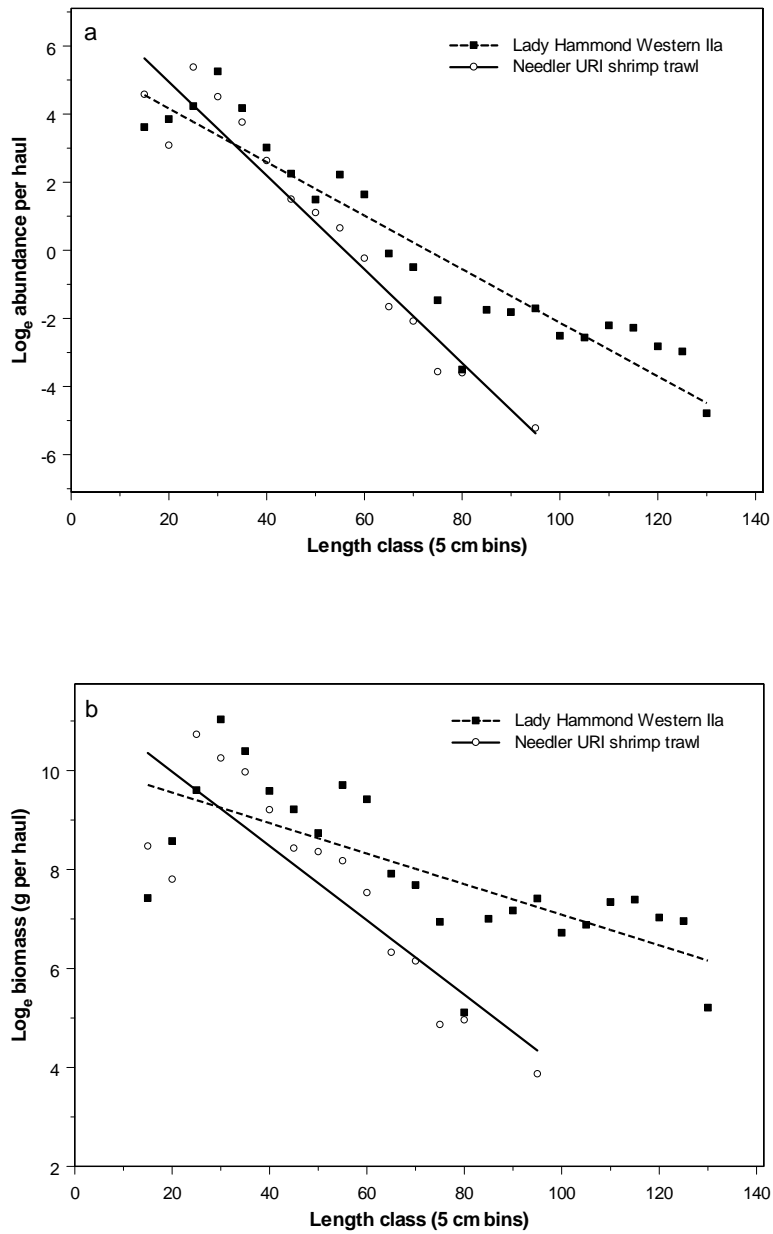
Figure 6.3: (a) Abundance size spectra and (b) biomass size spectra for the northern Gulf of St. Lawrence summer survey in 1990 comparative tows between two gears and vessels.

Table 6.2: Least squares fit of size spectrum slopes and intercepts.

	SLOPE	INTERCEPT	R^2	F	DF	P	CORRELATION OF COEFFICIENTS
Lady Hammond Western IIA abundance	-0.08	5.74	0.88	166	22	0.000	-0.90
Alfred Needler URI abundance	-0.14	7.70	0.94	205	13	0.000	-0.91
Lady Hammond Western IIA biomass	-0.03	10.17	0.52	24	22	0.000	-0.90
Alfred Needler URI biomass	-0.08	11.48	0.74	37	13	0.000	-0.91

All size spectra fits were highly significant (Figure 6.3, Table 6.2), though fits to abundance size spectra were considerably better than to biomass size spectra (Table 6.2). In all cases the *Lady Hammond* survey caught fewer small fish and more large fish than the *Alfred Needler* survey using the shrimp trawl. Both gears seemed to perform equally at about 30 cm fish length. The use of 30 cm as the subjective transition between small and large fish (ICES, 2003) and assessing temporal changes in biomass in these compartments could produce serious bias if using this survey time series uncorrected. The change in sampling efficiency between these gears at the 30 cm mark may lead one to the conclusion that fish >30 cm declined after 1990 while small fish increased. This might appear as a cultivation effect (Walters and Kitchell 2001) if one were not aware of the size dependent sampling biases inherent in the gears. When examining the slope of the size spectrum, it would appear that the slope became much steeper after 1990. Coincidently, the period from the late 1980s and early 1990s is when cod, a major species in this system, declined rapidly in abundance leading to stock collapse and imposition of a fishing moratorium in 1994 (CSAS 2003). Many other systems on the eastern Canadian shelf also showed large and real changes in this period. Therefore, even with the knowledge that gear and survey changed in 1990, it would be tempting to attribute some portion of the change in slope or abundance of large fish resulting simply from survey changes as a real effect.

6.5 Case study IV: The effect of standardisation on the average length of *Amblyraja radiata*

If observations from different gears with different size-dependent catchability patterns are used to obtain estimates of fish metrics, the measured effort must be standardised relative to one of the gears to avoid biased estimates. In this case study, the catch rates of 8m-beam trawl underlying case study I are standardised relative to the GOV, and the effect of standardisation on the average length of *A. radiata* is explored. Only trawl hauls from the area where the two gear types had overlapping coverage were used.

The length distribution of *A. radiata* from 8m-beam and GOV trawl looks different with a proportionately higher frequency of larger individuals in the GOV catches (Figure 6.4). Consequently, the average lengths per year of *A. radiata* increase when beam trawl data are standardised relative to GOV (Figure 6.5).
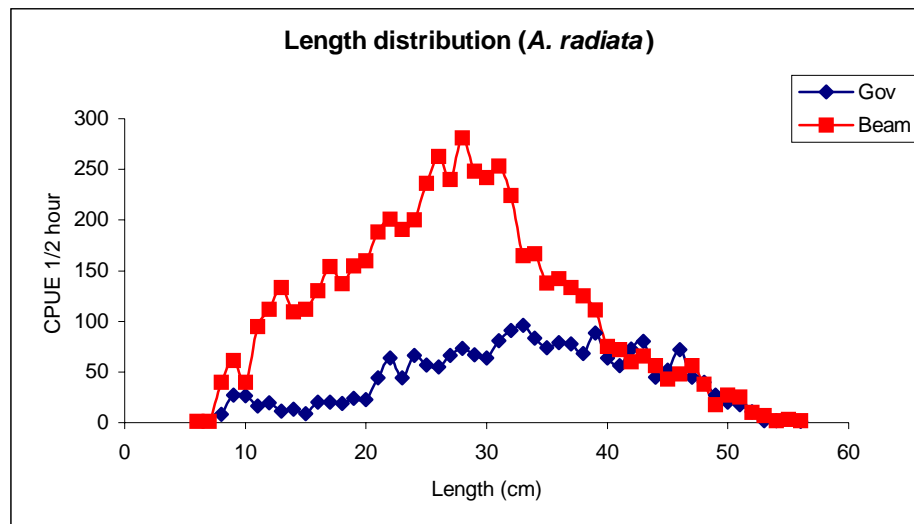
Figure 6.4: Total length distribution of 8m-beam trawl and GOV trawl catches.
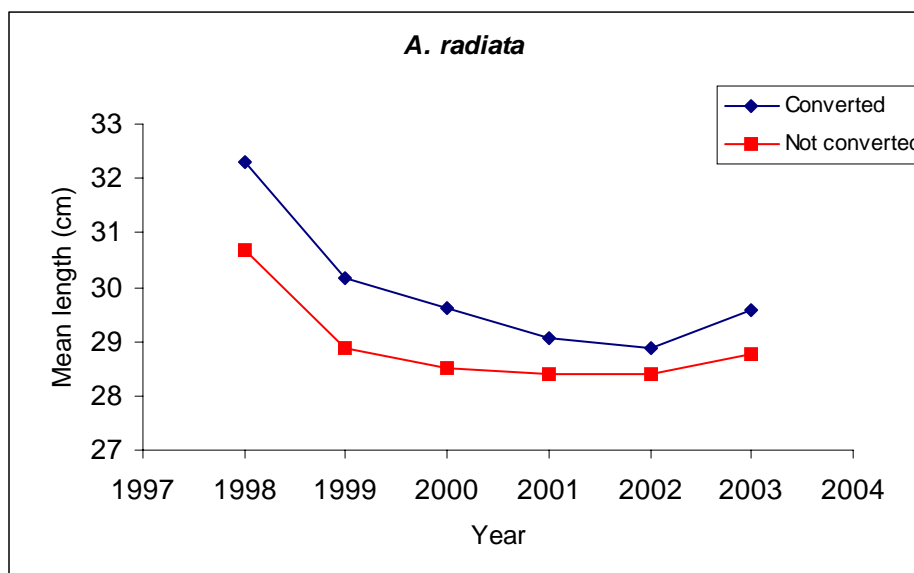
Figure 6.5: Average length of *Amblyraja radiata* based on original and standardised (beam trawl catches converted to GOV level) catch rates.

6.6   Case study V: Barents Sea surveys

Three research surveys conducted by PINRO in the Barents Sea in 2000 were used for testing the effect of gear type on fish community descriptions. The winter survey (winter TAS) is conducted during February in the southern part of the Barents Sea, and this survey used a Norwegian Campelen 1800 trawl (width 50 m, height 5 m, mesh size 22 mm). The Haddock survey is conducted during May-June in the southern part of the Barents Sea, and used a Russian research trawl type 2283–02 (width 20 m, height 8 m, mesh size 30 mm). The demersal fish survey (TAS) is conducted during October-December over the whole Barents Sea, and also uses the Russian research trawl type 2283–02 (width 20 m, height 8 m), though the mesh size is 16 mm. The area of overlap, which comprised 11 local fisheries areas, was chosen for testing, with several stations sampled in each area.

Table 6.3: Mean catches and species number of some fish groups from the data of various surveys in the Barents Sea in 2000.

FISH GROUPS	MEAN CATCH, SPEC PER HOUR			NUMBER OF SPECIES		
	Winter TAS	Haddock survey	TAS	Winter TAS	Haddock survey	TAS
Wolffish	7.67	8.03	2.52	3	3	3
Gadoids	1710.75	663.72	301.82	9	7	6
Capelin	782.73	52.93	10.83	1	1	1
Herring	417.21	3.12	928.22	1	1	1
Redfish	22.22	7.91	3.68	3	3	3
Flatfish	549.52	108.00	120.27	5	5	6
Skates	29.91	5.11	7.73	2	1	3
Eelpouts	17.42	0.07	0.81	6	2	5
Cottoids	209.28	1.01	3.68	9	4	9
Liparids	14.89	0.37	0.68	4	2	4
Stichaeids	14.33	0.06	1.16	3	1	3
Other	26.89	0.82	3.92	4	2	5
Total	3788.472	851.15	1384.189	50	32	50
Trawl number				80	57	96

It is obvious that catches of some small fish species were very different between the different gear types (Table 6.3). Though it is not appropriate here to examine differences in the catches of migratory species, the catches of small non-migratory species are shown, which indicate great variability between surveys. The minimum and maximum catches rates for these groups indicated very large differences in catch rates – 248x (eelpouts), 207 (cottoids), 40 (liparids) and 239 (stichaeids), while the differences in larger species was less pronounced (3–7 times).

As these surveys were not conducted in the same season, some characteristics of the fish community are reflected by the seasonal migrations of fish. Hence, the length distribution of capelin, polar cod and herring were not compared here. Gadoids, especially cod, haddock and blue whiting, are known to be migratory species, and this is reflected in the length distributions of this group (Figure 6.6). Non-migratory species (or species where knowledge concerning migrations is absent) can be assumed to occur during all seasons in the areas covered by the surveys. For some large species (e.g., skates) there were no obvious differences in length distribution, but for wolffish the proportion of the smallest specimens was lowest in the haddock survey. The length distribution of redfish for medium and large specimens was very similar in all surveys, but specimens < 16 cm were not observed in the haddock survey. The length distribution of flatfish was very similar in the haddock survey and TAS in fish with length more then 20 cm, and was different from the length distribution based on data from the winter TAS.

The biggest differences in the length distribution between the three surveys were observed in small non-target species (cottids, eelpouts and liparids). The smallest specimens (< 9–15 cm depending on species) were practically absent in the catches of the haddock survey. Hence, the different gear types (trawl design, mesh size etc.) can have an impact on our perception and description of the fish community and some metrics (e.g., length distributions, mean length in community etc). The total size spectra patterns were variable (Figure 6.7). The slopes were practically the same in the TAS and haddock surveys (-0.1999) and were less then in the winter TAS (-0.0215). Similarly, the intercepts were also very close in the TAS and haddock surveys (2.8316 and 2.8872 respectively) and less than in the winter TAS (3.0577).

Using data collected by different gears for the analysis of fish communities can affect the results, with cluster analysis of trawl data resulting in different patterns in the similarity of fish communities in the different fisheries areas (Figure 6.8).
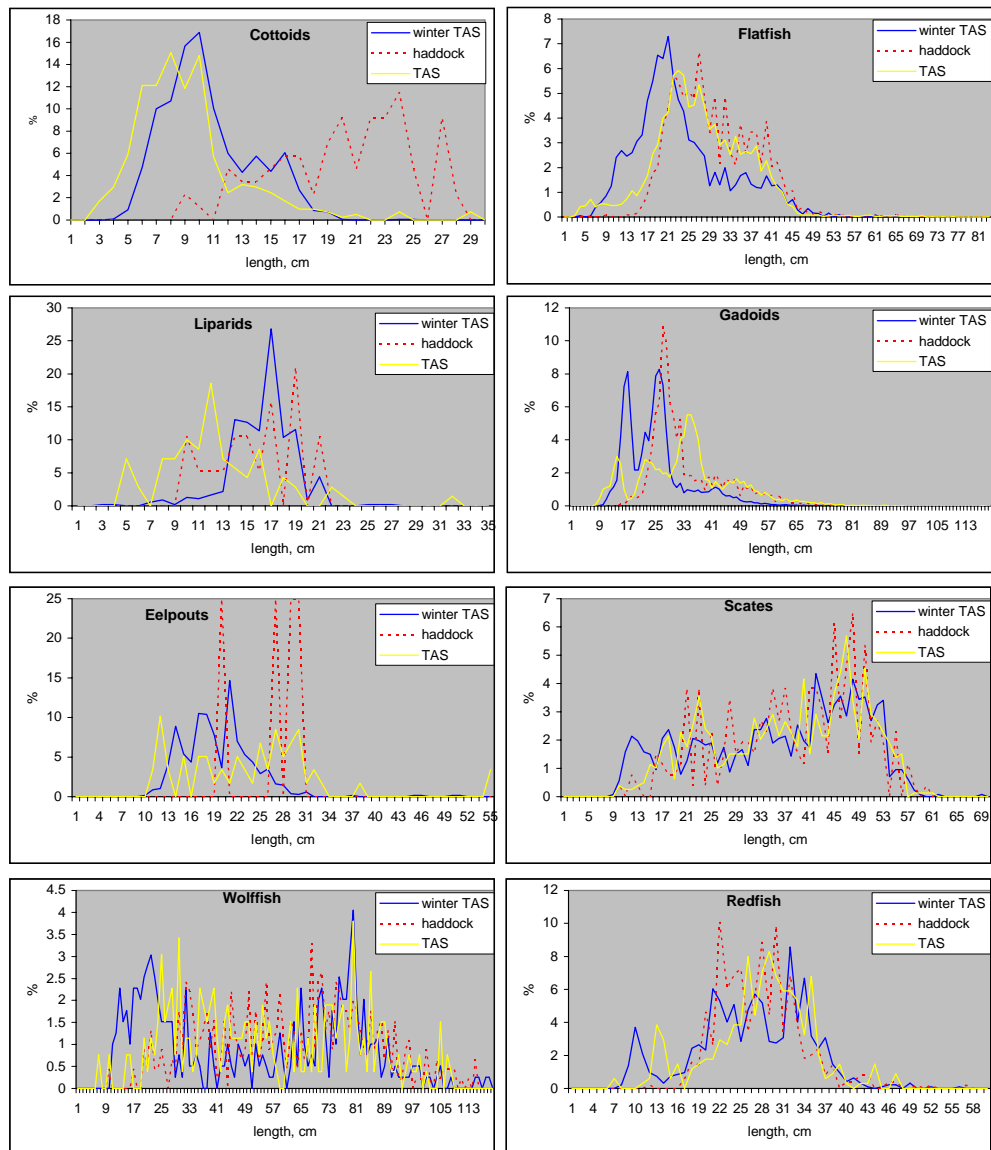
Figure 6.6: Length composition of some fish groups from the data of various surveys in 2000.
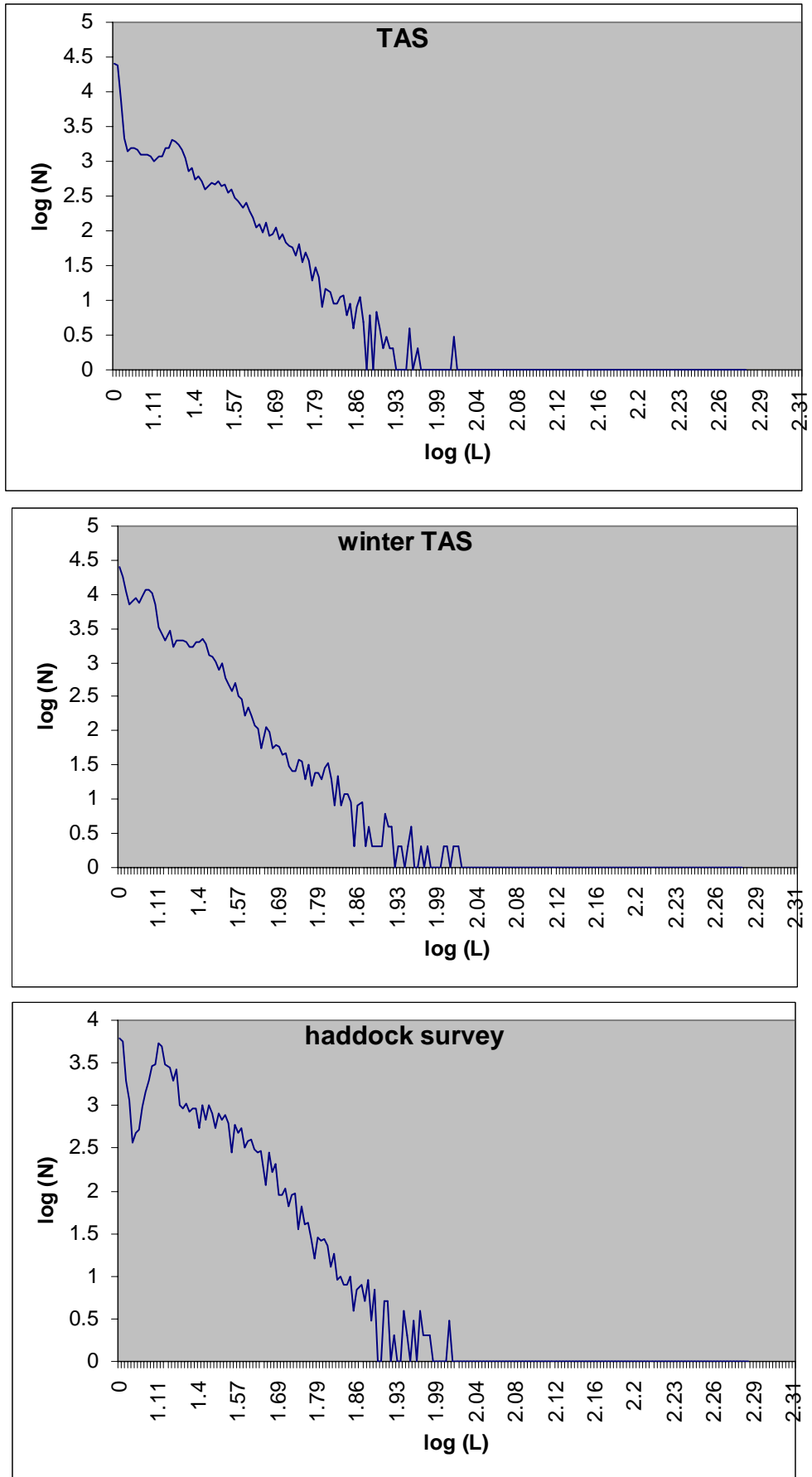
Figure 6.7: Size spectra based on the data collected in the different surveys.

(a)

A

Bray-Curtis Cluster Analysis (Single Link)

| 0. % Similarity | 50. | 100 |

B

Bray-Curtis Cluster Analysis (Single Link)

| 0. % Similarity | 50. | 100 |

C

Bray-Curtis Cluster Analysis (Single Link)

(b)
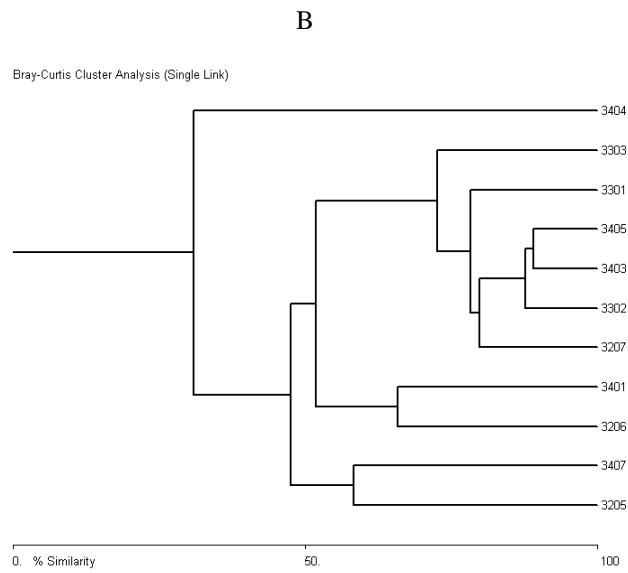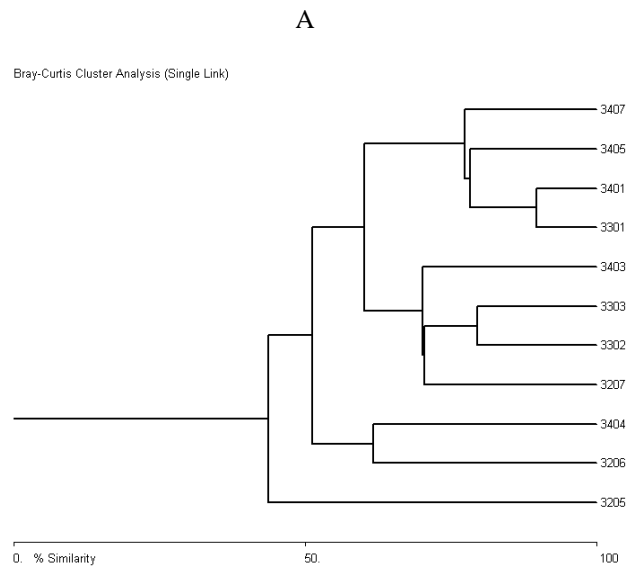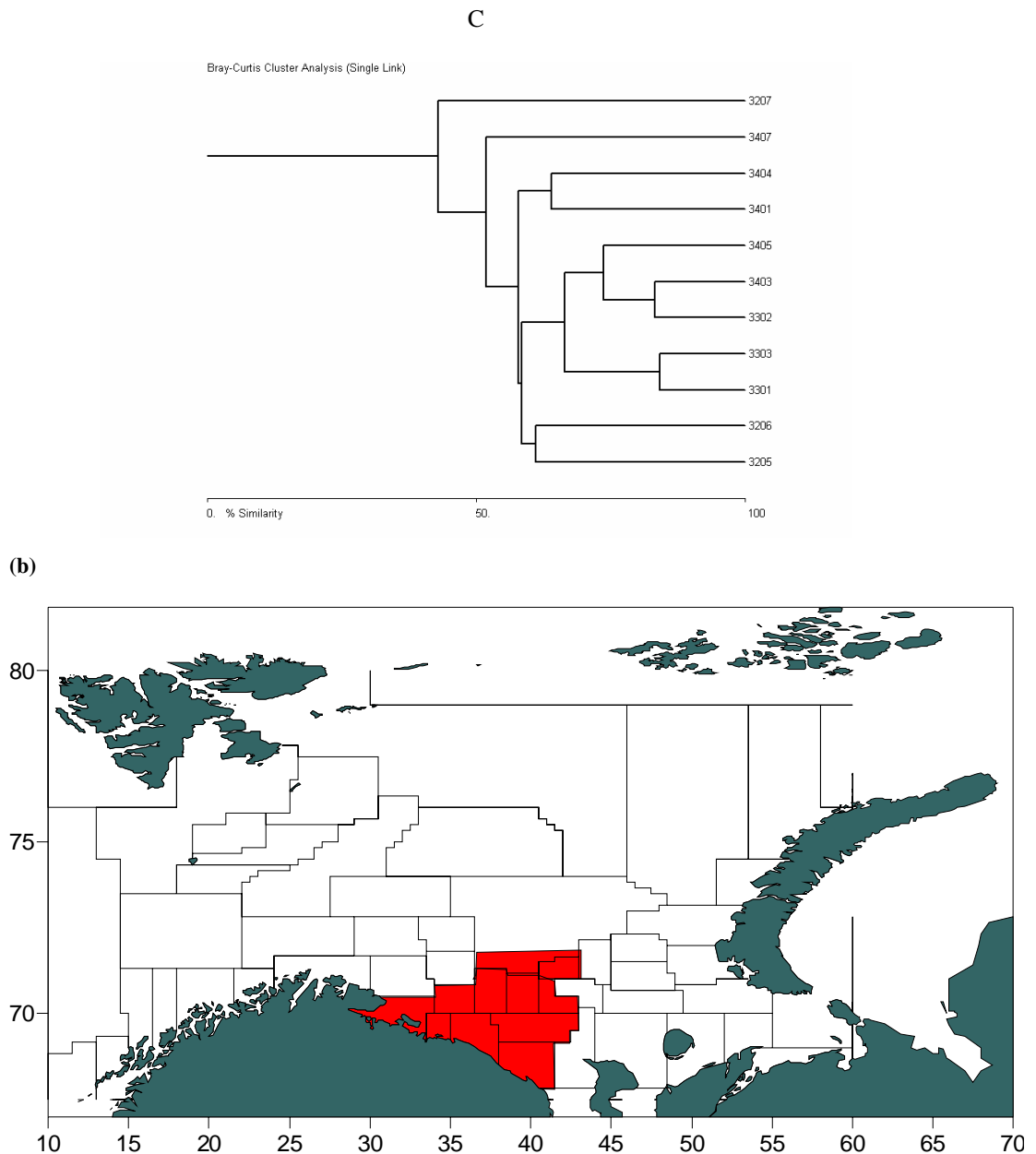
Figure 6.8: Clustering analysis of local fisheries areas using the data collected by different gears (A – the haddock survey, B – TAS, C – winter TAS, right – codes of Russian local fisheries areas) and (b) map of the Russian local fisheries areas in the Barents Sea, with areas used in the cluster analysis in red.

6.7 Case study VI: Differences in catches and fish size between a baca trawl and a 3.5 beam trawl in Galician waters

The Cantabrian shelf demersal communities have been sampled annually since 1983 during bottom trawl surveys conducted by the Instituto Español de Oceanografía. These surveys use a baca trawl (44/60), thus giving information on demersal and benthic megafauna (Sánchez, 1993; Sánchez *et al.*, 2002; Sánchez and Serrano, 2003). Beam trawl sampling was introduced in October 2000, with the aim of improving the quantitative sampling of epibenthic communities (Serrano *et al.*, in press), with special interest on 0-group flatfishes and invertebrate communities, which are prey of fishes (Serrano *et al.*, 2003). After the oil spill caused by the wreck of the vessel *Prestige*, multigear surveys were developed for a monitoring project (ECOPREST). This case study used the information obtained in comparative baca/beam trawl

hauls. Data were analysed for spring and winter, as there is seasonal variability in fish distributions.

Differences in the catchability of fishes between a baca trawl and a 3.5m beam trawl (BT), in the northwestern Spanish shelf (Galician shelf) were analysed. Catches were compared in terms of biomass (kg/km^2) and numbers (individuals/km^2). Data were obtained in 4 surveys, spring 2003 and 2004, and autumn 2003 and 2004. The sampling strategy (Figure 6.9) consisted of eight transects perpendicular to the coastline covering three depth strata (70–120 m, 121–200 m, 201–300 m), resulting in 23 stations per survey (one of the stations was located on non-trawlable rocky grounds). At every station a baca trawl and beam trawl were used. Baca sampling comprised tows of 30-minutes duration during daytime at a speed of 3 knots. Baca mesh size was 60 mm in the net and 10 mm in the cod end. The horizontal opening was 18.9 m and vertical opening 2.0 m. The beam trawl had a horizontal opening of 3.5 m and a vertical opening of 0.6 m, and mesh size was 10 mm. The beam trawl was towed for 15 minutes at a mean speed of 2.5 knots. Both trawl gears were monitored using a Scanmar net control system. The mean area swept was $54728 \pm 1532 \ m^2$ and $3307 \pm 192 \ m^2$ for the baca and beam trawl respectively.

Differences in fish catches between gears were determined using a Mann –Whitney Rank Sum test. In both season a Baca/BT factor (R) was calculated by dividing baca catches (weight and numbers) by beam trawl catches.
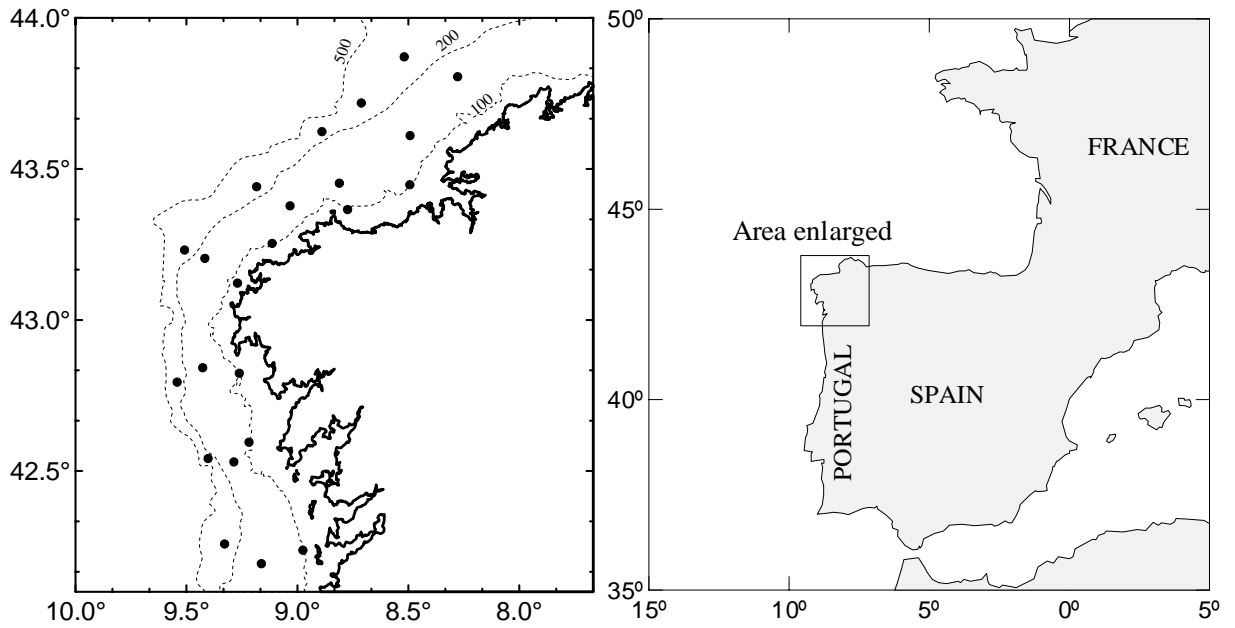
Figure 6.9: Location of sampling stations in the Galician shelf.

Both gears showed clear differences in terms of their catchability, because the baca trawl catch was mostly composed of fishes, whilst the beam trawl catch contained a higher biomass of invertebrates (Figure 6.10). Figure 6.10. The percentage of fish in the catch (by biomass) was higher in autumn for both gears.
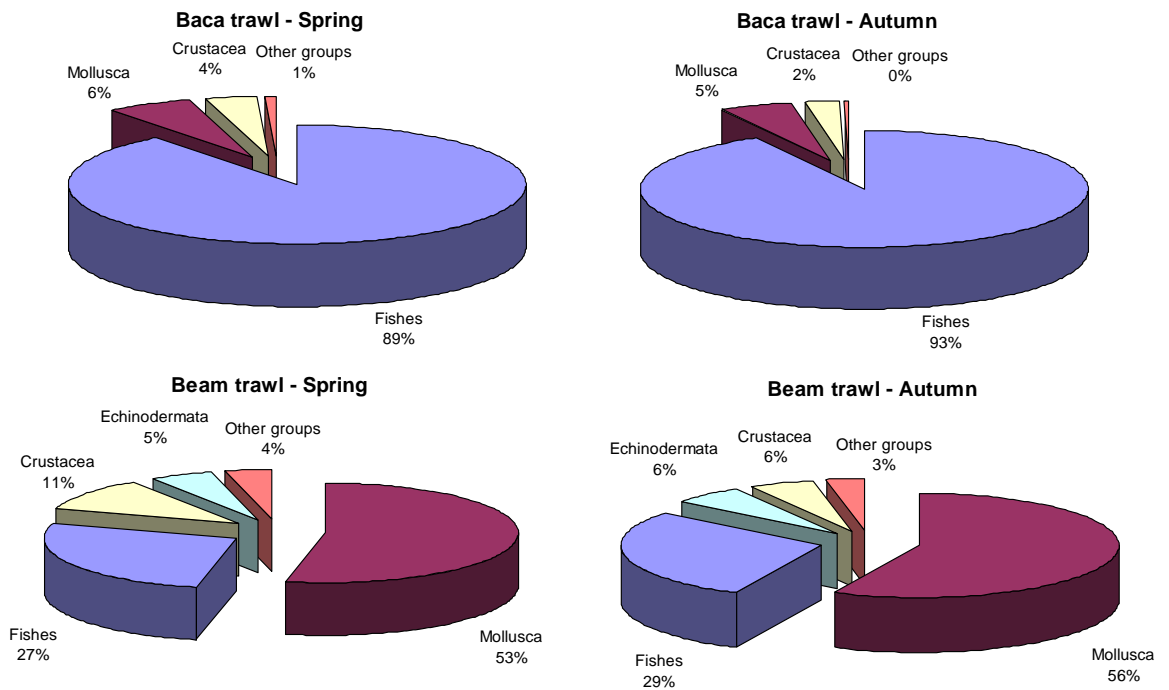
Figure 6.10: Relative catches (biomass) of taxa in both gears by season.

Table 6.4 shows the between-gear differences in weight and numbers of fish caught, and according to the Mann-Whitney test, the Baca trawl caught a higher biomass than the beam trawl, while the beam trawl caught significantly more by numbers.

Table 6.4: Mean catches of fish by biomass (kg km^{-2}) and numbers (1000 individuals km^{-2}) SD= standard deviation; R= catchability coefficient; MW= Mann-Whitney Rank Sum Test.

	SPRING					**AUTUMN**			
	Baca	**BT**	***R***	**MW**		**Baca**	**BT**	***R***	**MW**
W	1288.0	725.7	1.8	2240 / 0.0055		1471.6	823.5	1.8	2278 / 0.0441
SD$_W$	1015.2	446.8				1269.3	499.5		
N	31.1	57.6	0.5	1553 < 0.0001		57.3	137.4	0.4	1485 / < 0.0001
SD$_N$	24.9	39.6				55.2	132.0		

Furthermore, different trends were apparent for the relative catchability of different species (Tables 6.5–6.8). The baca trawl was more efficient at catching semi-pelagic and demersal species, such as blue whiting *Micromesistus poutassou* that resulted in an extremely high value of R (Tables 6.5–6.7). Other large demersal and fast swimming species, like hake *Merluccius merluccius*, bib *Trisopterus luscus*, conger eel *Conger conger*, forkbeard *Phycis blennoides* were also better sampled with the baca. Catches of blackbelly rosefish *Helicolenus dactylopterus* were higher in the baca trawl, in terms of biomass, but more individuals were caught with beam trawl. In contrast, benthic and small-bodies fish species (e.g., megrims, triglids, monkfish, dragonets, gobiids, flatfishes) had a much greater catchability in the beam trawl (Tables 6.5–6.7).

For some groups of species there were no major differences in the length distributions between the two gears, for example hake, anglerfish, bib, gurnards (*Aspitrigla cuculus, Eutrigla gurnardus*), gobies and thickback sole (Figure 6.11). In contrast, beam trawls sampled better blue whiting less than 12 cm, and juvenile four-spotted megrim *Lepidorhombus boscii*, conger eel and forkbeard. A similar pattern was exhibited for the catshark *Scyliorhinus canicula* and rays (*Raja clavata* and *Raja montagui*), with a unimodal curve in Baca and a bimodal curve in beam trawl. Differences in blackbelly rosefish catches were also observed, with a high proportion of juveniles caught by beam trawl.

In summary, the relative efficiency of the beam trawl was higher for juveniles of benthic species, while little difference was apparent for some demersal species. Future studies will examine differences between length distributions using bootstrap technique.

Table 6.5: Baca and beam-trawl catches of main fish groups in biomass (kg/km^2) and numbers (ind./km^2) and by season. _R_= catchability coefficient. Groups ordered in descending order of _R_. Horizontal lines separate groups of baca higher efficiency, BT higher efficiency, and groups absent in one of gears.

	WEIGHT – SPRING				WEIGHT - AUTUMN		
Taxon	**Baca**	**BT**	**_R_**	**Taxon**	**Baca**	**BT**	**_R_**
Blue whiting	742.68	3.37	220.063	Blue whiting	942.57	0.42	2249.118
Rays	27.75	1.85	14.992	_Trisopterus_	50.33	6.53	7.713
Sharks	149.07	12.89	11.563	Hake	73.43	12.60	5.826
Conger eel	2.68	0.36	7.542	Myctophids	0.13	0.02	5.330
Forkbeard	1.39	0.18	7.524	Rays	14.32	4.34	3.303
Hake	59.31	10.77	5.506	Conger eel	19.48	6.67	2.923
Megrim	0.88	0.16	5.316	Sharks	25.15	10.31	2.440
Macrourids	2.81	0.62	4.493	Blackbelly rosefish	10.97	8.23	1.333
Blackbelly rosefish	18.24	7.12	2.561	Forkbeard	1.27	1.02	1.253
Trisopterus	26.63	30.12	0.884	Silvery pout	75.45	154.06	0.490
Triglidae	27.59	55.23	0.499	Four spot megrim	32.48	86.28	0.377
Four spot megrim	40.77	101.83	0.400	Triglids	18.22	62.78	0.290
Monkfish	40.51	117.24	0.346	Monkfish	21.62	77.19	0.280
Silvery pout	23.88	90.31	0.264	Dragonets	9.19	49.57	0.185
Dragonets	6.95	27.23	0.255	Other flatfishes	33.75	283.12	0.119
Other flatfishes	24.55	236.60	0.104	Gobiids	0.29	13.06	0.022
Gobiids	0.02	2.89	0.007	Macrourids	0.03		
Myctophids				Megrim	0.63		
Sparids	37.20			Sparids	93.61		

	Numbers - Spring				Numbers - Autumn		
Taxon	**Baca**	**BT**	**_R_**	**Taxon**	**Baca**	**BT**	**_R_**
Blue whiting	19334.1	105.4	183.433	Blue whiting	29105.4	12.9	2257.122
Macrourids	46.9	13.2	3.557	_Trisopterus_	1589.3	103.2	15.406
Conger	16.6	6.6	2.515	Hake	2936.4	419.1	7.007
Sharks	446.1	177.9	2.508	Myctophids	117.4	19.3	6.067
Hake	961.6	408.4	2.354	Forkbeard	27.7	19.3	1.431
Rays	44.4	19.8	2.249	Conger	122.7	90.3	1.360
Trisopterus	476.0	250.3	1.901	Rays	24.8	32.2	0.769
Blackbelly rosefish	371.7	283.3	1.312	Blackbelly rosefish	335.1	451.3	0.743
Megrim	4.8	6.6	0.736	Sharks	75.2	148.3	0.507
Triglidae	427.5	1198.9	0.357	Silvery pout	17909.1	42888.5	0.418
Monkfish	29.5	98.8	0.298	Triglidae	314.2	1237.9	0.254
Silvery pout	3564.4	18333.3	0.194	Monkfish	31.4	167.6	0.187
Four spot megrim	649.3	3965.7	0.164	Four spot megrim	608.3	4900.1	0.124
Forkbeard	8.9	118.6	0.075	Other flatfishes	2071.1	38252.7	0.054
Other flatfishes	1361.2	25362.3	0.054	Dragonets	240.9	17214.7	0.014
Dragonets	126.5	2931.5	0.043	Gobiids	248.8	20606.1	0.012
Gobiids	6.1	2022.4	0.003	Macrourids	0.4		
Myctophids	0.4			Megrim	2.5		
Sparids	123.2			Sparids	290.9		

Table 6.6: Catches of all fish species present in both gears by biomass (kg/km^2) and by season. _R_= catchability coefficient. Ranked by descending values of _R_ for the autumn.

	SPRING			AUTUMN		
	Baca	BT	_R_	Baca	BT	_R_
Micromesistius poutassou	742.68	3.37	220.063	942.57	0.42	2249.118
Zeus faber	10.09	0.01	1094.581	13.67	0.15	94.228
Maurolicus muelleri				316.00	3.80	83.158
Trisopterus minutus	9.02	0.18	50.318	14.93	0.94	15.806
Cepola rubescens	3.74	0.08	44.299	5.19	0.55	9.391
Argentina sphyraena	16.61	1.75	9.493	22.41	3.34	6.719
Trisopterus luscus	17.61	29.94	0.588	35.40	5.58	6.344
Merluccius merluccius	59.31	10.77	5.506	73.43	12.60	5.826
Molva dypterygia				2.42	0.47	5.167
Conger conger	2.68	0.36	7.542	19.48	6.67	2.923
Trigla lucerna	10.61	7.55	1.404	7.62	2.84	2.684
Scyliorhinus canicula	144.68	12.89	11.223	24.18	10.31	2.346
Raja clavata	21.76	1.54	14.116	9.47	4.34	2.184
Lepidotrigla cavillone	0.84	0.11	7.973	1.72	0.81	2.135
Chelidonichthys obscurus	2.97	8.77	0.339	0.73	0.36	2.013
Macroramphosus scolopax	0.02			0.02	0.01	1.526
Helicolenus dactylopterus	18.24	7.12	2.561	10.97	8.23	1.333
Aspitrigla cuculus	8.73	5.21	1.675	3.55	2.67	1.331
Phycis blennoides	1.39	0.18	7.524	1.27	1.02	1.253
Blennius ocellaris	2.48	2.70	0.921	1.80	1.57	1.148
Lophius piscatorius	37.26	77.47	0.481	16.21	27.98	0.579
Gadiculus argenteus	23.88	90.31	0.264	75.45	154.06	0.490
Lepidorhombus boscii	40.77	101.83	0.400	32.48	86.28	0.377
Microchirus variegatus	13.58	55.17	0.246	15.96	53.03	0.301
Callionymus lyra	6.89	22.78	0.303	8.75	29.14	0.300
Capros aper	17.92	2.59	6.920	2.42	8.91	0.272
Solea solea	2.68	13.56	0.198	3.64	16.16	0.225
Serranus cabrilla		1.37		0.09	0.45	0.198
Lophius budegassa	3.24	39.76	0.082	5.41	49.21	0.110
Lesueurigobius friesii	0.01	0.66	0.015	0.12	1.17	0.106
Buglossidium luteum	0.11	2.29	0.048	0.31	3.58	0.087
Eutrigla gurnardus	4.26	33.59	0.127	4.40	56.10	0.078
Arnoglossus laterna	8.04	153.33	0.052	13.32	205.76	0.065
Pomatoschistus sp.	0.00	1.85	0.001	0.16	7.30	0.022
Callionymus maculatus	0.05	4.45	0.012	0.45	20.42	0.022
Gaidropsarus macrophthal- mus	0.02	9.78	0.002	0.45	31.05	0.014
Arnoglossus imperialis	0.02	0.16	0.101			
Crystallogobius linearis	0.01	0.01	0.836	0.00		
Deltentosteus quadrimacula- tus	0.00	0.37	0.002		4.56	
Lepidorhombus whiffiagonis	0.88	0.16	5.316	0.63		
Malacocephalus laevis	2.81	0.62	4.493	80.00		
Mullus surmuletus	3.83	1.17	3.263	0.05		
Raja montagui	5.98	0.31	19.313	4.85		
Pegusa lascaris	0.12	12.09	0.010		4.50	

Table 6.7: Catches of all fish species present in both gears by numbers (ind./km^2) and by season. R= catchability coefficient. Ranked by descending values of R for the autumn.

	NUMBERS - SPRING			NUMBERS - AUTUMN		
	Baca	**BT**	**R**	**Baca**	**BT**	**R**
Micromesistius poutassou	19334.1	105.4	183.433	29105.4	12.9	2257.122
Trisopterus minutus	686.0	1.0	686.000	1327.7	64.5	20.592
Merluccius merluccius	961.6	408.4	2.354	2936.4	419.1	7.007
Trisopterus luscus	492.0	37.0	13.297	261.6	38.7	6.762
Maurolicus muelleri	0.4			117.4	19.3	6.067
Molva dipterygia				95.9	25.8	3.717
Argentina sphyraena	801.2	138.3	5.792	2600.0	767.2	3.389
Aspitrigla cuculus	212.5	131.8	1.613	45.0	19.3	2.329
Trigla lucernas	46.5	32.9	1.411	29.8	12.9	2.307
Cepola rubescens	51.3	26.4	1.947	80.6	38.7	2.083
Lepidotrigla cavillone	24.6	6.6	3.741	49.6	25.8	1.923
Zeus faber	52.0	1.0	52.000	21.1	12.9	1.634
Phycis blennoides	8.9	118.6	0.075	27.7	19.3	1.431
Conger conger	16.6	6.6	2.515	122.7	90.3	1.360
Chelidonichthys obscurus	35.6	131.8	0.270	7.4	6.4	1.154
Blennius ocellaris	101.0	112.0	0.902	81.4	90.3	0.902
Helicolenus dactylopterus	371.7	283.3	1.312	335.1	451.3	0.743
Scyliorhinus canicula	1100.0	27.0	40.741	74.4	148.3	0.502
Gadiculus argenteus	3564.4	18333.3	0.194	17909.1	42888.5	0.418
Raja clavata	81.0	2.0	40.500	12.4	32.2	0.385
Lophius piscatorius	26.3	32.9	0.797	25.6	90.3	0.284
Callionymus lyra	116.4	487.5	0.239	151.7	541.6	0.280
Solea solea	14.0	6.0	2.333	5.8	25.8	0.224
Microchirus variegatus	581.4	3267.5	0.178	702.5	3204.4	0.219
Eutrigla gurnardus	105.9	895.9	0.118	179.3	1173.4	0.153
Macroramphosus scolopax	1.2			0.8	6.4	0.128
Lepidorhombus boscii	649.3	3965.7	0.164	608.3	4900.1	0.124
Capros aper	2122.0	553.4	3.835	703.3	6299.2	0.112
Lophius budegassa	3.2	65.9	0.049	5.8	77.4	0.075
Buglossidium luteum	4.4	118.6	0.037	14.9	219.2	0.068
Lesueurigobius friesii	2.4	388.7	0.006	44.6	683.4	0.065
Arnoglossus laterna	768.5	21864.3	0.035	1347.5	34751.8	0.039
Serranus cabrilla		1.0		0.4	19.3	0.021
Pomatoschistus sp.	1.2	1093.5	0.001	201.2	12056.7	0.017
Gaidropsarus macrophthalmus	1.6	1383.4	0.001	33.5	3539.7	0.009
Callionymus maculatus	10.1	2444.0	0.004	89.3	16673.1	0.005
Arnoglossus imperialis	0.4	6.6	0.061			
Crystallogobius linearis	2.0	19.8	0.102	2.9		
Deltentosteus quadrimaculatus	0.4	520.4	0.001		7846.6	
Lepidorhombus whiffiagonis	4.8	6.6	0.736	2.5		
Malacocephalus laevis	46.9	13.2	3.557	0.4		
Mullus surmuletus	18.6	13.2	1.411	0.4		
Raja montagui	28.0	1.0	28.000	12.4		
Pegusa lascaris	2.0	10.0	0.200		38.7	

Table 6.8: Fish species present in one gear but absent in the other.

BACA TRAWL	BEAM TRAWL
Boops boops	*Citharus linguatula*
Chelidonichthys lastoviza	*Echiichthys vipera*
Chimaera monstrosa	*Hyperoplus inmaculatus*
Echiodon dentatus	*Labrus bimaculatus*
Galeus melastomus	*Petromyzon marinus*
Hexanchus griseus	*Scorpaena scrofa*
Lampanyctus crocodilus	*Symphurus nigrescens*
Lepidopus caudatus	*Syngnathus acus*
Leucoraja circularis	
Notoscopelus kroeyerii	
Ophidion barbatum	
Pagellus acarne	
Pagellus bogaraveo	
Scophthalmus rhombus	
Scyliorhinus stellaris	
Spondyliosoma cantharus	
Trigla lyra	
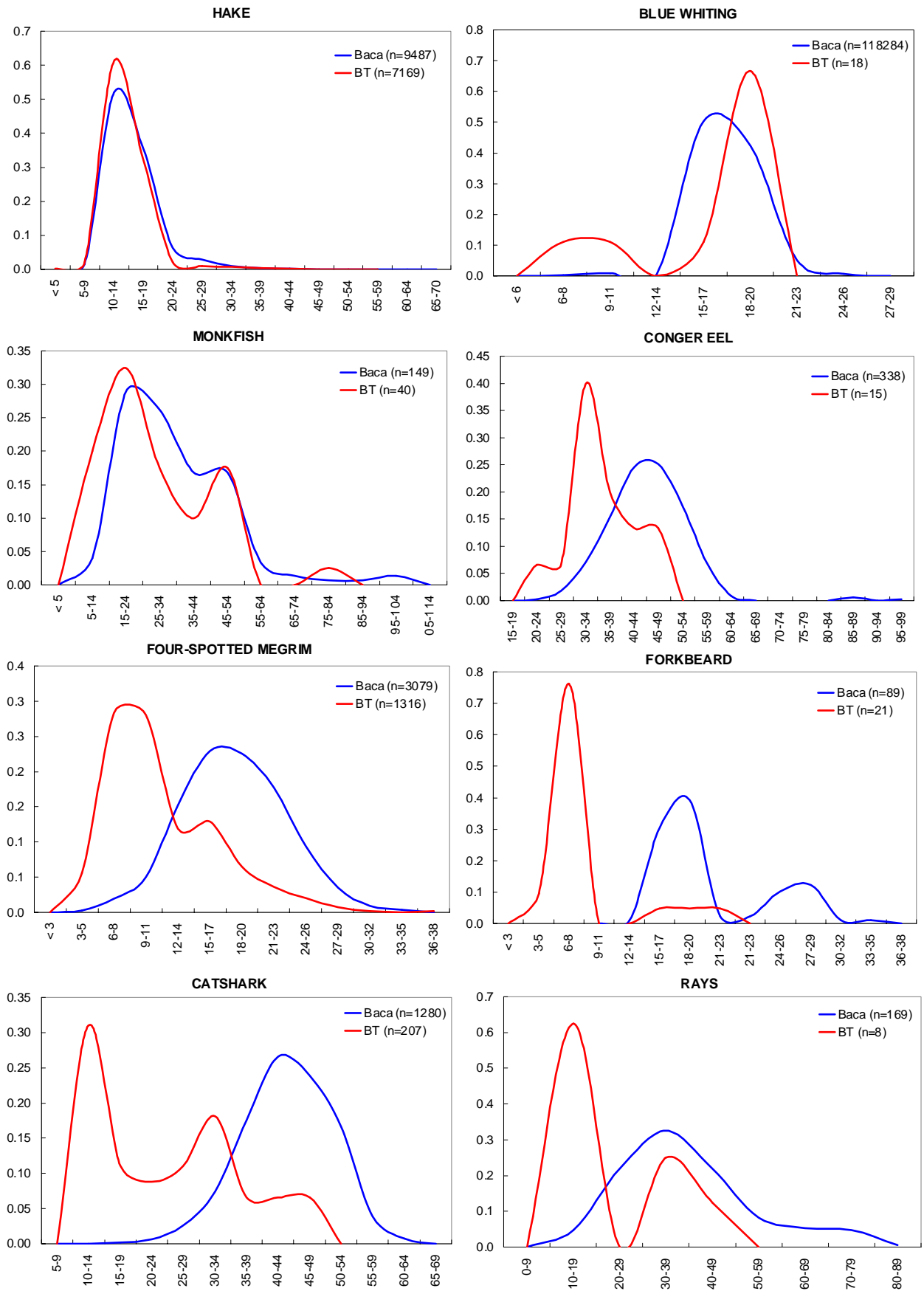
Figure 6.11: Relative length distribution of main species in beam and baca trawl.
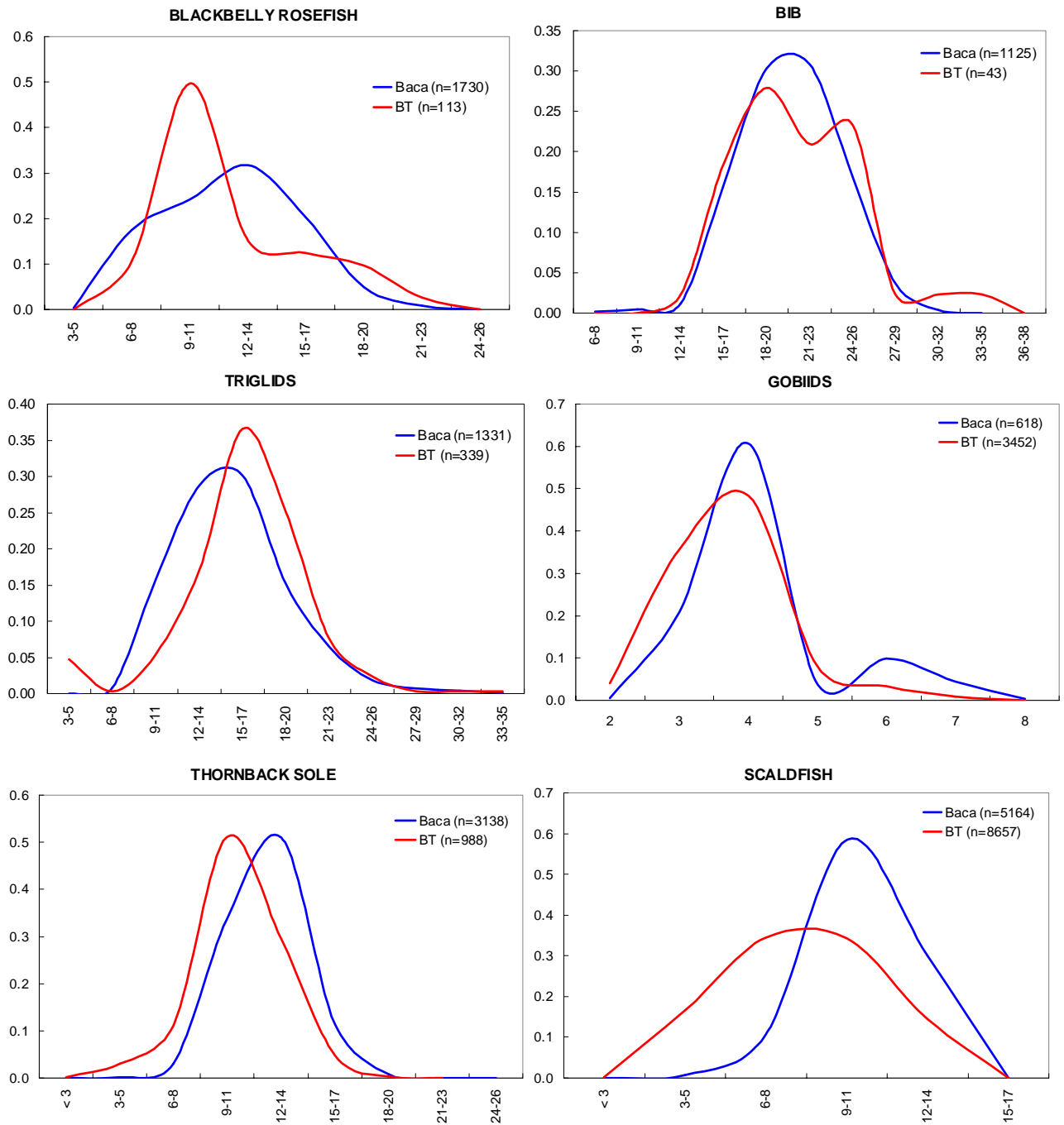
Figure 6.11 (continued). Relative length distribution of main species in beam and baca trawl.

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7 Abundance-occupancy relationships in fishes

7.1 Introduction

The effects of exploitation on macroecological patterns have received only limited attention (Fisher and Frank, 2004). Abundance-distribution relationships have been found over a broad range of species (Gaston *et al.* 1998). Such relationships may be useful for highlighting species of concern. A consequence of intraspecific relationships in abundance-distribution for fisheries is that catch rates will be proportionately higher for a given level of effort, given a positive abundance-distribution relationship (Paloheimo and Dickie, 1964; Fisher and Frank, 2004). The collapse of the northern cod stock at Newfoundland co-occurred with a hyper aggregation of the cod at low stock abundance. This increased the CPUE, in spite of a strong reduction in population abundance, and is likely to have contributed to the collapse of this stock (Rose and Kulka, 1999). Knowing how a stock responds spatially to changes in abundance can therefore be important to prevent stock collapse. An improved knowledge of abundance-occupancy relationships are also important for the effective implementation of any spatial management actions: for example, related to establishing marine protected areas (Jennings, 2000; Fisher and Frank, 2004).

Sampling can have an impact on the detection of abundance-distribution relationships. Density dependent catchability is considered a potential source of sampling bias that can affect density estimates and thus establishment of abundance-distribution relationships. Another issue is age dependent catchability. Many species are very widespread during the young life stages but catchability is often low and variable. This is likely to vary between species and hence possibly influence both intra- and interspecific abundance-distribution relationships. Thus there are likely to be some effects of sampling issues on the abundance-distribution relationship. A more thorough analysis of this topic should be considered.

Theoretical aspects of abundance-distribution relationships were discussed in detail in last years report of WGFE (ICES, 2004b). A recent study by Fisher and Frank (2004) examined both intra- and inter-specific abundance range size-relationships in 24 commercial and non-commercial species in Canadian waters, with 13 of these species and 16 of 34 stocks exhibiting significant intraspecific relationships. The interspecific relationship was also significant. Here we present a related analysis looking at the intra- and interspecific abundance-distribution relationship in fishes in the Northwest Atlantic, the North Sea and the Barents Sea.

7.2 Intra and interspecific abundance – occupancy

Fisher and Frank (2004) noted differing relationships between abundance of a species and distribution (spatial variation). Two case studies examined here showed very different relationships. Area occupied and abundance for Norway pout (*Trisopterus esmarki*) in the North Sea, were positively correlated during the period 1998–2004 while for thorny skate (= starry ray, *Amblyraja radiata*) on the Grand Banks, the relationship changed following a period of decline in abundance as described below.

7.2.1 Western North Atlantic

Thorny skate is the dominant species of skate on the Grand Banks and is presently the target of the largest elasmobranch fishery in the Canadian and surrounding waters of the northwest Atlantic, fished by Canada (inside 200 miles), and Spain, Portugal and Russia (outside 200 miles). The stock has been under quota and managed by Canada inside 200 miles since the mid-1990's. Prior to 2004, however, thorny skate was not regulated by quota outside of 200 miles, though mesh size was restricted to a minimum of 280 mm in the codend as of 2002. In

2004, skate occurring in NAFO (Northwest Atlantic Fisheries Organization) Divisions 3LNO, that part of the Grand Bank overlapping the NAFO Regulatory Area, were placed under NAFO management. An initial quota was set at 13,500 tonnes, which approximated current catch levels. Thus, this is an important exploited elasmobranch stock. Further details of the thorny skate fishery and its management can be found in Kulka *et al.* (2004).

Kulka *et al.* (2004) noted not only significant reductions in abundance for Grand Banks thorny skate but also distributional changes. The proportion of the surveyed area containing no skates increased from about 3% in 1980–1988 to 25% in 2001–2003 (Figure 7.1, left panels). At the same time, the biomass became increasingly more concentrated. During 1980–1988, about 57% of the biomass was located within 20% of the survey area and the concentrations of skate located over the southern Grand Banks (see Figure 7.2). By 2001–2003, 83% of the biomass was concentrated into 20% of the surveyed area, primarily along the southwest slope of the Grand Bank. At the same time, the catch rate, both in terms of number and kg has increased where skates were concentrated within each year since 1992; more so within the area where skate concentrated in 2001–2003. This indicates that the skate are becoming progressively concentrated, or hyper-aggregated, within a small portion (20%) of the available area on the Grand Banks. Plotting relative biomass against percent of biomass in 20% of the area (x-axis) shows a trajectory moving to the right on the x-axis starting in 1992, following the decline in biomass (Figure 7.3).
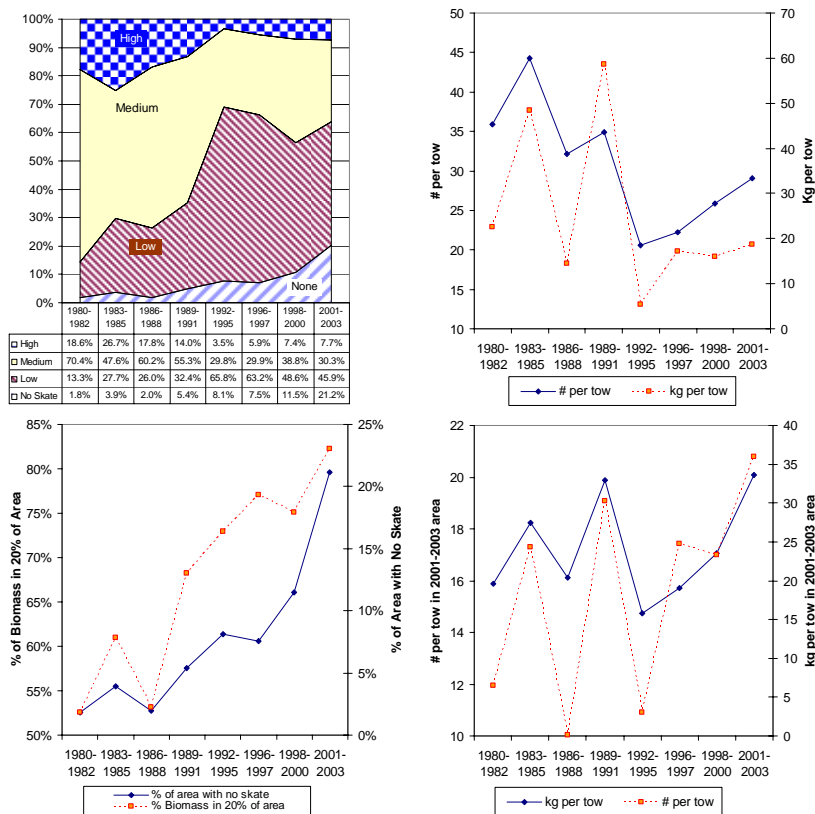
Figure 7.1: Changes in area of occupancy 1980–2003: Upper Left – percent of the area with none, low, medium, and high densities of thorny skate; Lower Left - percent of the area without thorny skate (solid line), and percent of biomass contained within 20% of the total area (dotted line); Upper Right – density (number and weight per tow) of skate in 20% of the area for those years; Lower Right – density (number and weight per tow) of skate in 20% of the 2001–2003 high density area (after Kulka *et al.*, 2004).

Examining trends in relative biomass and abundance (the traditional method for investigating population change) does not encompass all of the population dynamics of thorny skate, or indeed for any species. The analyses of spatial dynamics from fishery-independent survey data have revealed changes in the skate populations that would otherwise be difficult or impossible to detect, using aggregated statistics from commercial or survey sources. In addition to changes in relative biomass, thorny skate has also undergone substantial changes in its distribution since the 1980s (prior to the beginning of its decline in biomass). The analyses of spatial dynamics have revealed changes in the thorny skate population that would be difficult or impossible to detect with standard analytical (non-spatial) analyses. Hyper-aggregation (increasing density at the centre of distribution, decreasing at the periphery) was observed after the mid-1990s, when the skate population size was stable at a low level. More than 50 years of data suggest that this is a relatively recent phenomenon. A continuation of this trend could result in a further reduction of the stock, given that thorny skates are mostly concentrated in the area that is commercially fished. Commercial catch rates have remained and will probably continue to remain steady or increase in the short term even if the thorny skate population declines further. Spanish catch rates increased between 1995–1997 and 1998–1999, and del Rio and Junquera (2001) noted a further increase in skate catch rates on the Tail of the Grand Banks in 2001. This increasing rate corroborates the spatial contraction and increasing density observed in spring research surveys following the period of decline.
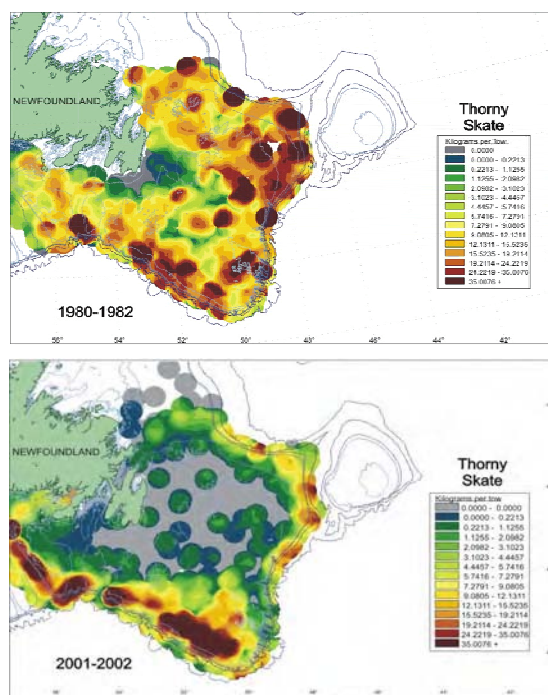
Figure 7.2: Distribution of thorny skate, 1980–1982 compared to 2001–2003 based on research trawl surveys. Grey area represents surveyed area with no catch. Brown areas represent highest concentrations (after Kulka *et al.*, 2004).

A similar pattern of aggregation was observed for northern cod *Gadus morhua* prior to its collapse (Rose and Kulka 1999). Similar to skate, 64–75% of the Atlantic cod biomass in 1983–1988 occurred in 20% of the habitat. In contrast, during the decline of northern cod (1989–1991), hyper-aggregation (87–89% in 20% of the area) was observed. Aggregation and reduced area of occupancy led to the cod being increasingly more vulnerable to exploitation, because they became more densely concentrated where levels of fishing effort could remain high. This is very similar to what is now happening to thorny skate. Whether these spatial dynamics are an indication of a skate stock under stress is uncertain, but such changes clearly

lead to increased vulnerability to exploitation. Thus, examination of thorny skate distribution dynamics constitutes an important part of understanding the status of the stock. This approach would benefit any assessment where geo-referenced fishery-independent data were available.
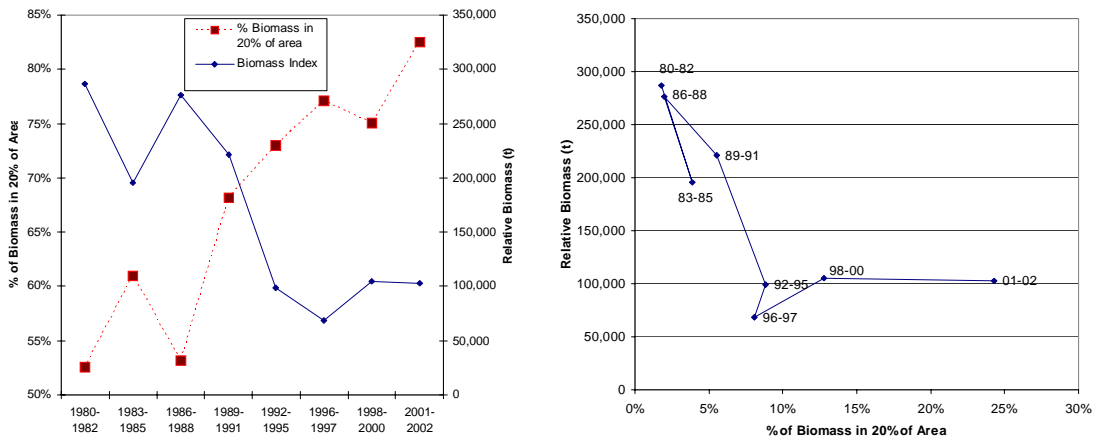
Figure 7.3: Changes in area of occupancy, 1980–2003: Left – percent of biomass contained within 20% of the total area (dotted line) compared to the survey biomass trajectory; Right – Relative biomass in relation to biomass concentration as a time line (after Kulka *et al.,* 2004).

7.2.2 North Sea

We analysed abundance–occupancy patterns for eight different North Sea species using IBTS data for the first quarter of the year. The analysis includes data on one year and older Norway pout *Trisopterus esmarki*, cod *Gadus morhua*, herring *Clupea harengus*, sprat *Sprattus sprattus*, whiting *Merlangius merlangus*, haddock *Melanogrammus aeglefinus*, mackerel *Scomber scombrus* and saithe *Pollachius virens*.

When the entire data set was taken together, a positive relationship between the log mean abundance and occupancy was found (Figure 7.4). This supports the general finding of many previous studies that the more abundant species tend to occupy larger areas. However, the analysis reveals considerable variation and several clusters can be recognised in Figure 7.4. When the data were broken down by species (Figure 7.5), the picture changed somewhat. Positive correlations where found for some species, including cod, sprat and Norway pout, while mackerel, saithe and haddock showed a low correlation between occupancy and abundance (Figure 7.6). Herring and whiting had intermediate slopes (Figure 7.6). In light of previous findings, spatial management action could be appropriate for North Sea cod and Norway pout.
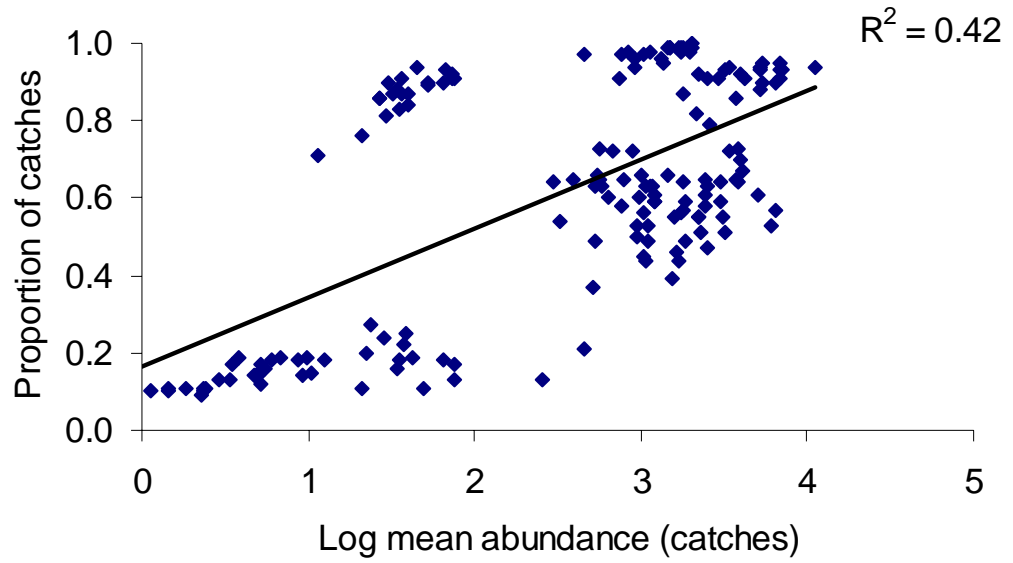
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Figure 7.4: Relationship between year-averaged occupancy (presence/absence) and log catch abundance in IBTS survey of the eight different species.
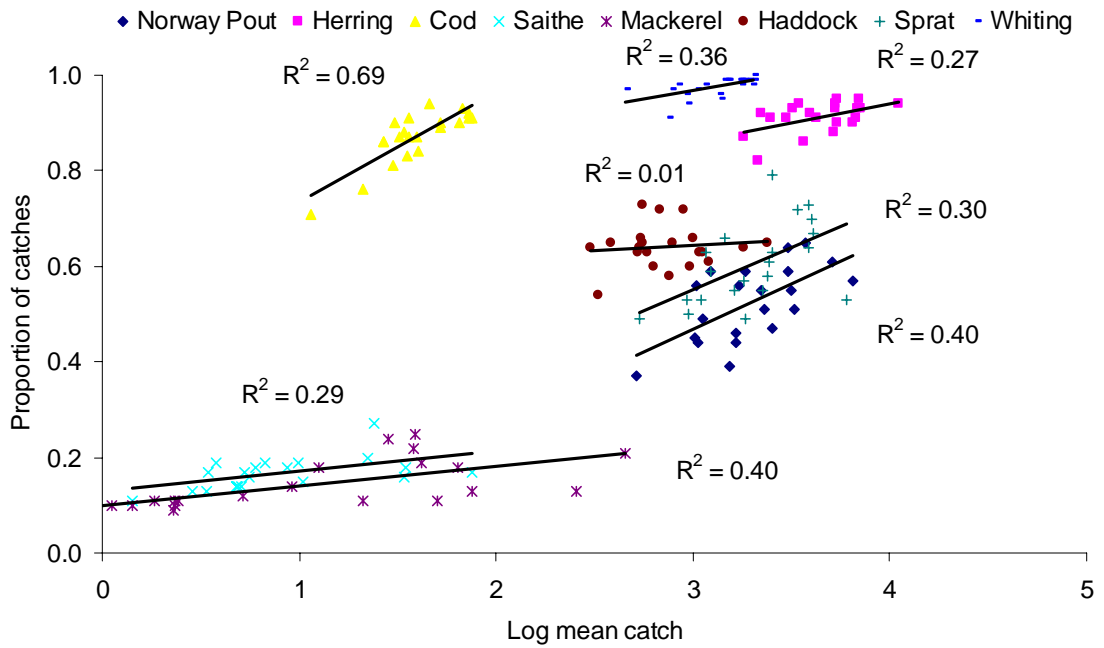
Figure 7.5: Occupancy-abundance relationship for eight different North Sea species from IBTS survey.
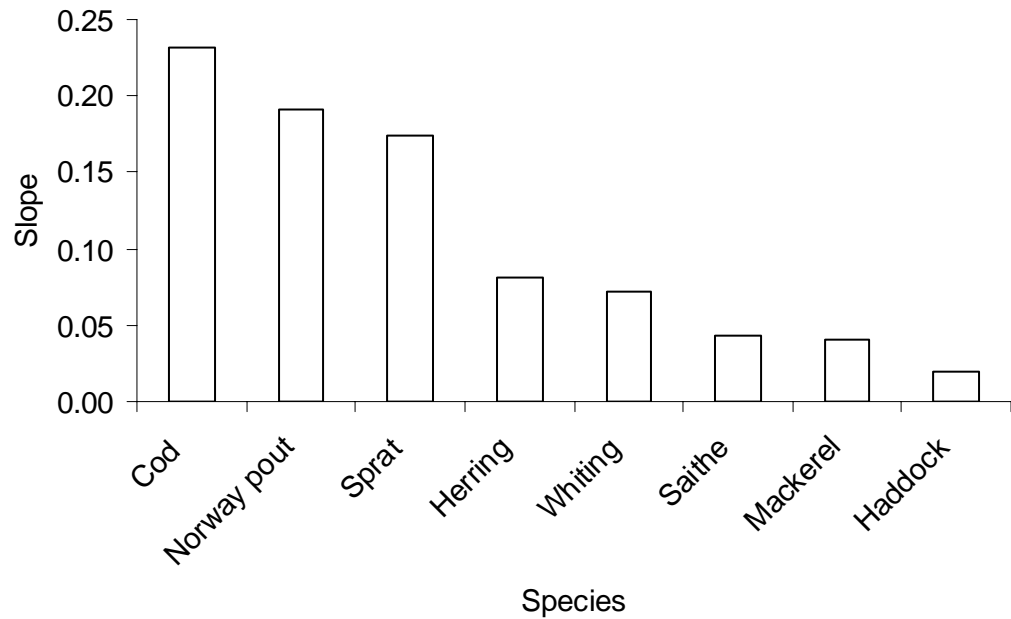
Figure 7.6: Slope between proportion of catches and abundance of different species.

Norway pout

The Norway pout stock in the North Sea is considered to be outside safe biological limits, and the forecast for this stock for 2005 suggests that even a zero fishing mortality in 2005 would leave the stock below B_{lim} (ICES, 2004a). The decline in this stock has taken place in spite of a historically low fishing mortality (ICES, 2004a). Following the recruitment of a strong year class in 1999, there has been a consistent decline in the Norway pout stock (Figure 7.7).
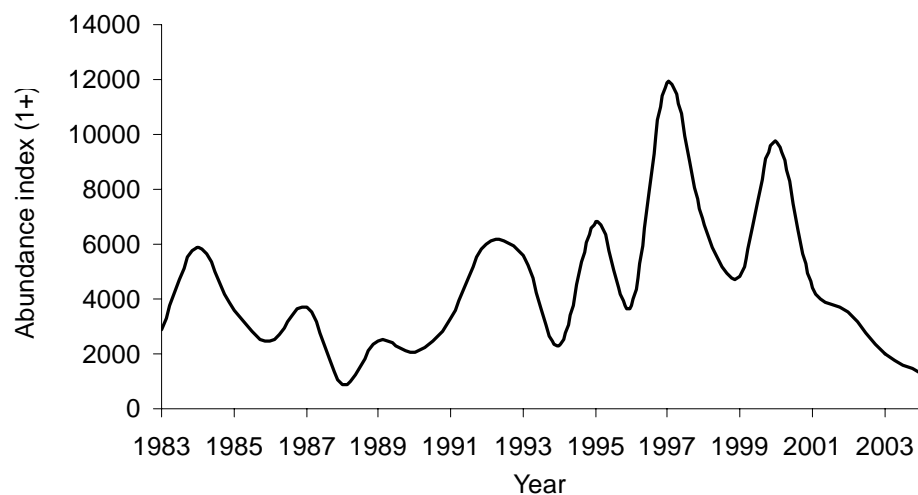
Figure 7.7: Abundance of Norway pout during the period 1983–2004.

Norway pout is distributed in the northern part of the North Sea at, mainly at depths of 100–160m (Figure 7.8). The change in distribution during the period 1998 to 2004 (Figures 7.9 and 7.10) shows that there has been a considerable decrease in areas with high abundance of Norway pout. While such areas were widely distributed during 1998–2002, there was a pronounced decline in 2003 and 2004. Indeed, the most recent survey (2004) revels high-density

areas only in the northwestern part of the distribution area (Figure 7.10). Concurrent with the decrease in the area of high abundance, a reduction in the occupancy was observed (Figure 7.11). The latter is in line with the occupancy-abundance relationship, suggesting that the stock has a fairly high colonising ability and that the spatial dynamics are driven by differences in vital rates or density dependent habitat selection. Natural mortality due to predation and post spawning mortality is very high in this stock and this along with several years of below average recruitment has contributed to the decline. Thus during the period 1983–2004, vital rates rather than exploitation appear to have driven the changes in spatial pattern. The short-lived nature of the species infers a density independent mortality on the Norway pout that will be homogenous over the area. This is different to the relationship reported thorny skate above and suggests that Norway pout is less likely to undergo rapid collapses due to over-exploitation, since pronounced reduction in CPUE is likely to precede such a collapse.
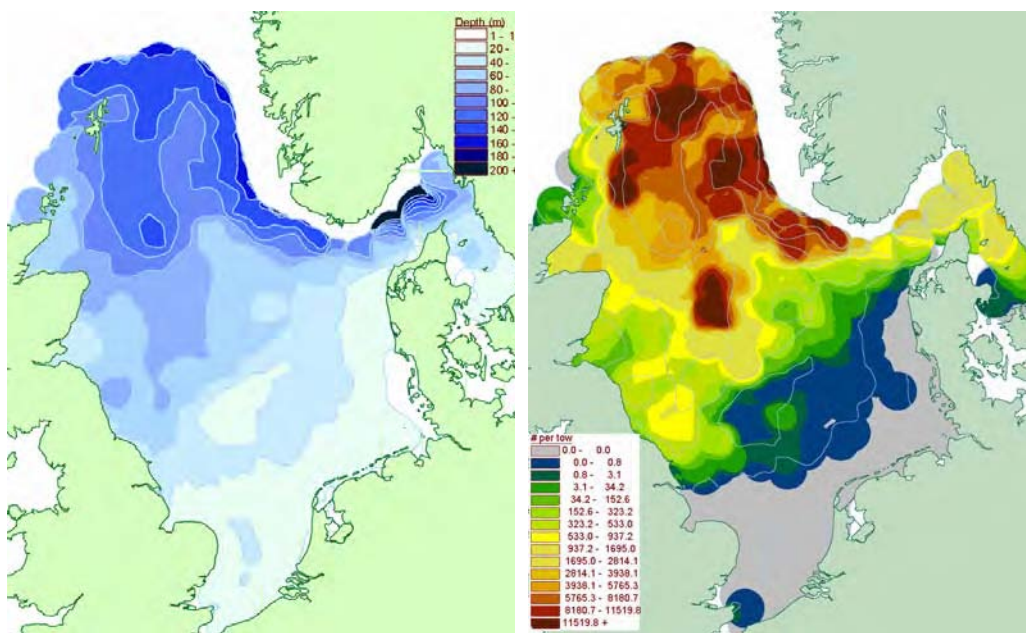
Figure 7.8: Interpolated bathymetry of the North Sea and the distribution of Norway pout sampled during IBTS surveys during the period 1983–2004. Grey area on the distribution map indicated areas sampled with no catches.
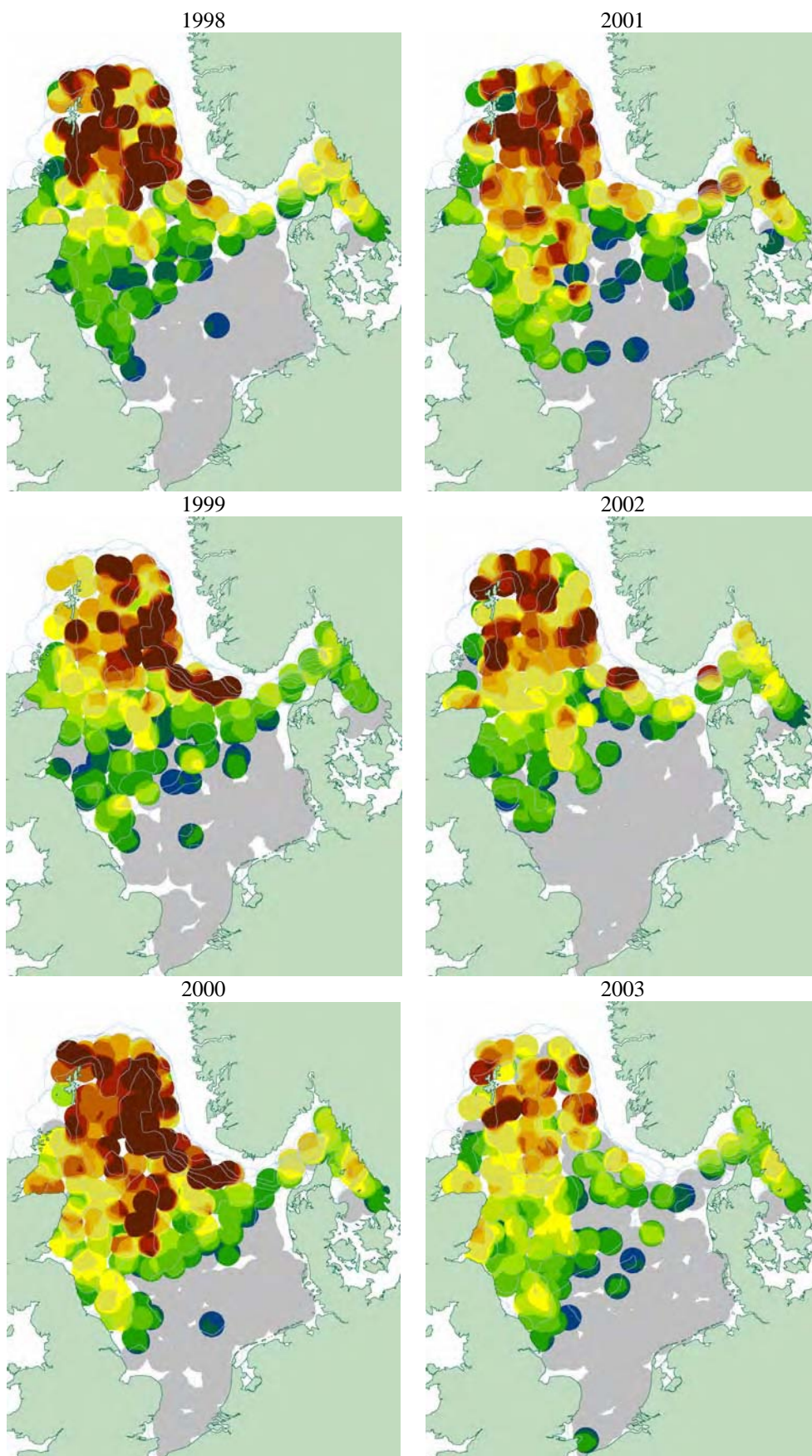
Fgure 7.9: Distribution of Norway pout (1+) abundance during 1998–2003.
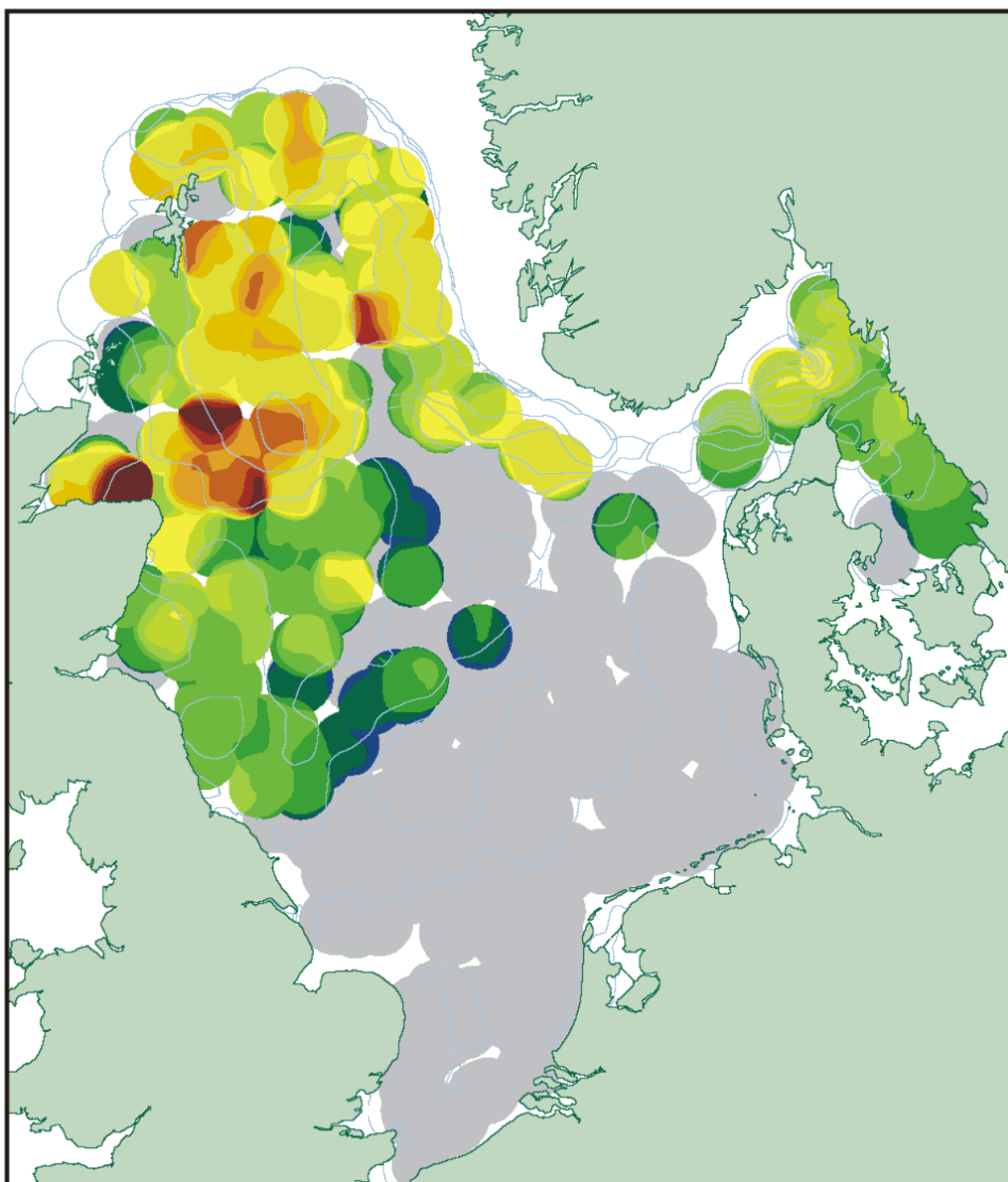
Figure 7.10: Distribution of Norway pout (1+) abundance in 2004.
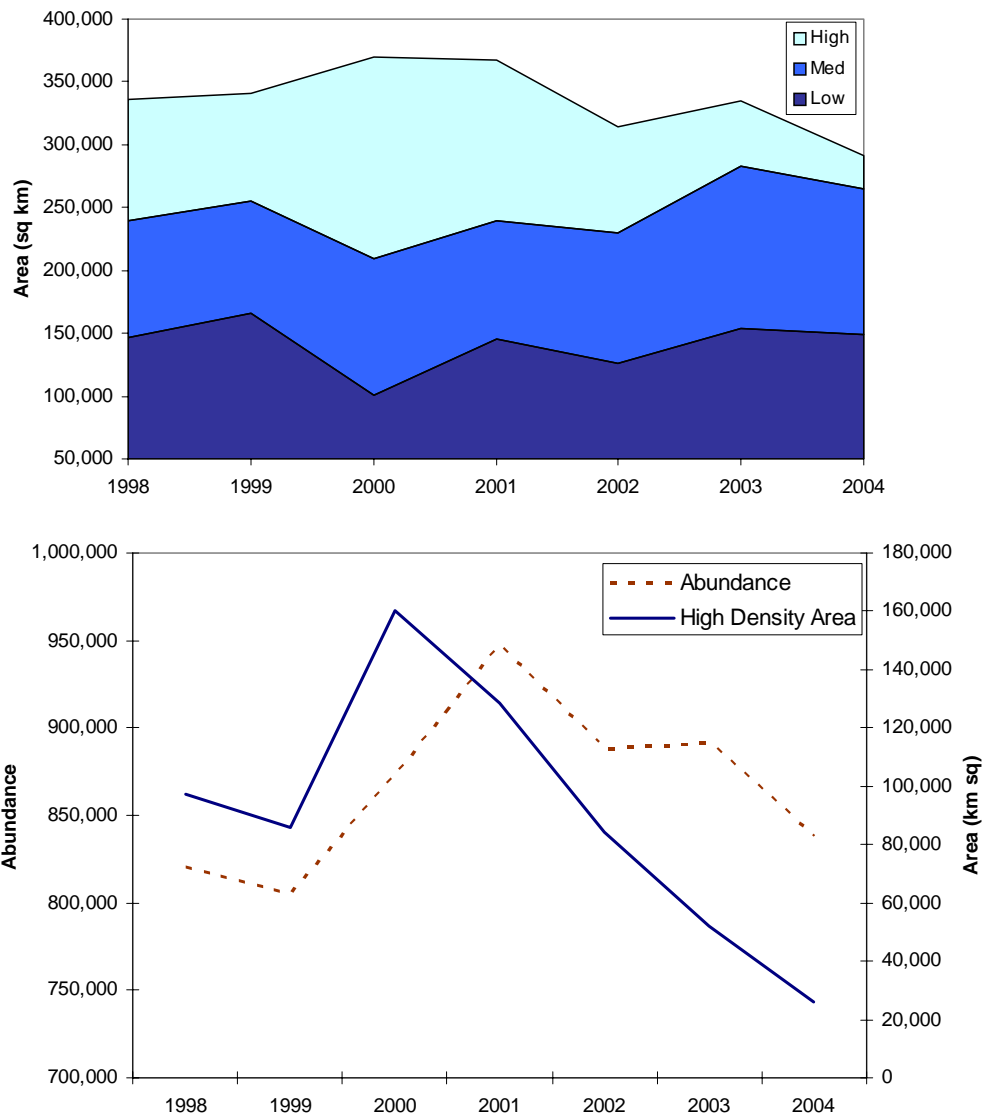
Figure 7.11: Upper panel: Change in area occupied by Norway pout in the North Sea at varying density, 1998–2004. Low = <323 individuals/ km^2; Med = 324–3938; High = > 3938 (refer to density classes in Figures 7.9 and 7.10). Lower panel: Relationship between year-averaged occupancy and log catch abundance in IBTS surveys. High density areas are defined as areas with catches > 3938 individuals.

7.2.3 Barents Sea

Data from the research survey conducted by PINRO during October-December 1998–2002 over the entire Barents Sea were analysed. The mean catch of each species (specimens per one hour haul) was used as an estimate of relative abundance. The proportion of trawl stations where a given species was captured was used as the indicator of the range of distribution. The data on the total stock biomass of cod, haddock, saithe and Greenland halibut *Reinhardtius hippoglossoides* were taken from the report of the Arctic Fisheries Working Group (AFWG) (ICES, 2004c). The data on the spawning stock biomass of blue whiting *Micromesistius poutassou* were taken from the report of Northern Pelagic and Blue Whiting Fisheries Working Group (WGNPBW) (ICES, 2004d).

A close relationship between relative abundance and the distribution range was observed for most of the species examined (Figure 7.12). Such relationships were found in species from all taxonomic, ecological and zoogeographic groups. In commercial fish species these relationships were tested using both relative abundance and the biomass estimated by the survey data or mathematic methods (Figure 7.13). Relatively strong relationships were observed for blue whiting and Greenland halibut, and the relationship was weaker for cod, haddock and saithe.

Comparison of influence the stock level and water temperature on the distribution range:

Blue whiting is abundant in the Barents Sea, where the northern border of its distribution is observed (Belikov *et al.*, 2004). The positions of the northern and eastern borders of its distribution in the Barents Sea are variable and depend on different factors. The good relationship between water temperature on some standard oceanographic section (Figure 7.14) and the latitude and longitude of distributional borders was revealed (Dolgov and Karsakov, in prep.; Table 7.1).

An analysis of the influence of the blue whiting stock level on the range size was conducted. Distribution borders in blue whiting in the Barents Sea were generally less well correlated with stock level than with water temperature. Hence, though abundance-range size relationships are often a significant feature of marine fishes, they can be constrained by environmental boundaries.

Table 7.1: Pearson's coefficient values of relationships between the distributional borders of blue whiting and water temperature and stock levels (Spawning Stock Biomass) in the Barents Sea

PARAMETERS	DISTRIBUTION BORDER IN THE BARENTS SEA	
	Latitude	Longitude
Fugløye-Bjørnøye Section	0.27	0.02
Kola Section (stations 3–7)	0.40	**0.74**
Kola Section (stations 8–10)	**0.87**	0.18
Section 29	-0.07	**0.65**
Section 31	0.49	**0.60**
SSB (Norwegian data), millions spec.	**0.77**	**0.57**
SSB (ISVPA estimations), thousand tons	**0.58**	**0.57**
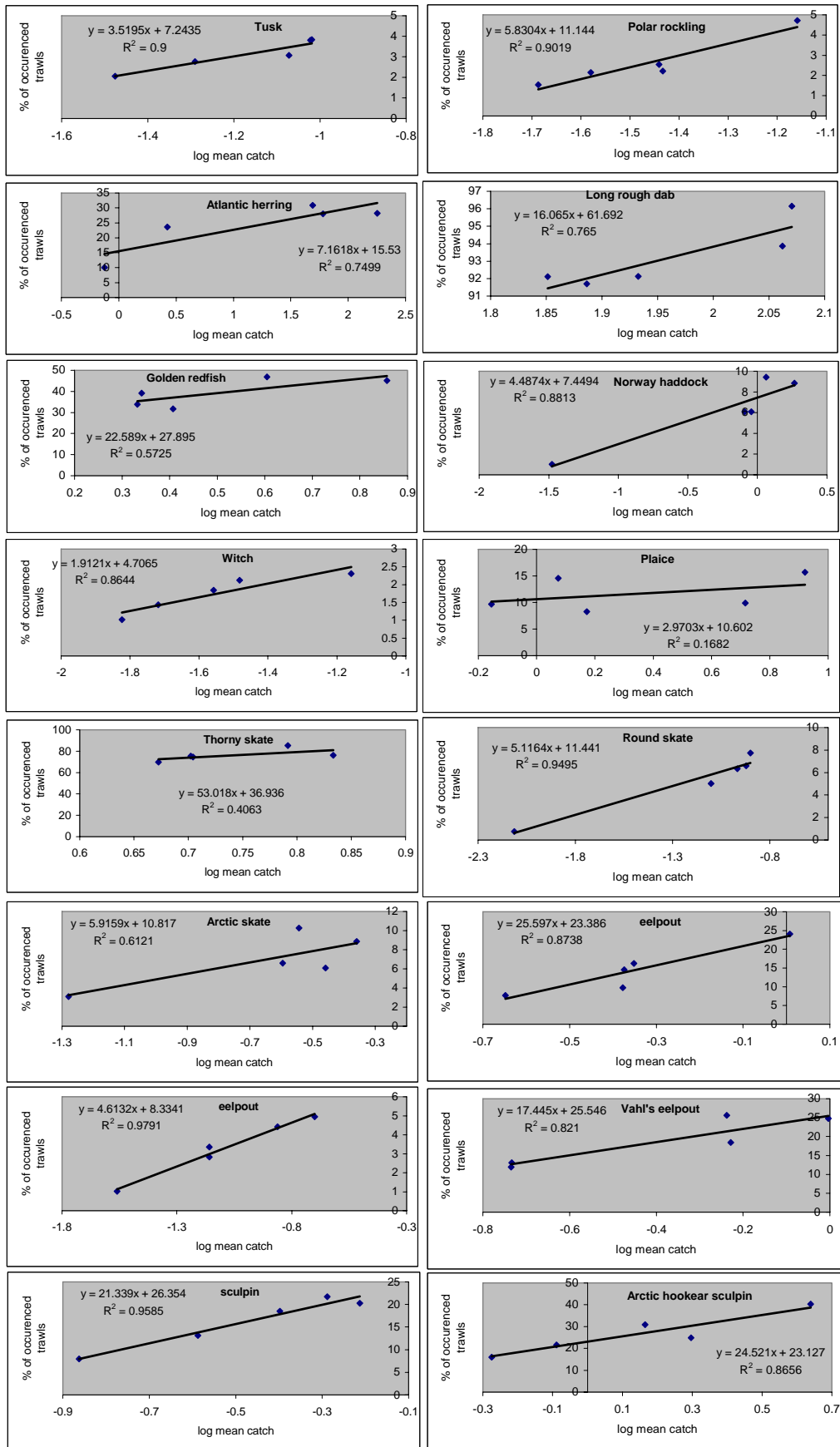
Figure 7.12: Relationships between distribution range and mean catches in the most important commercial fish species of the Barents Sea.
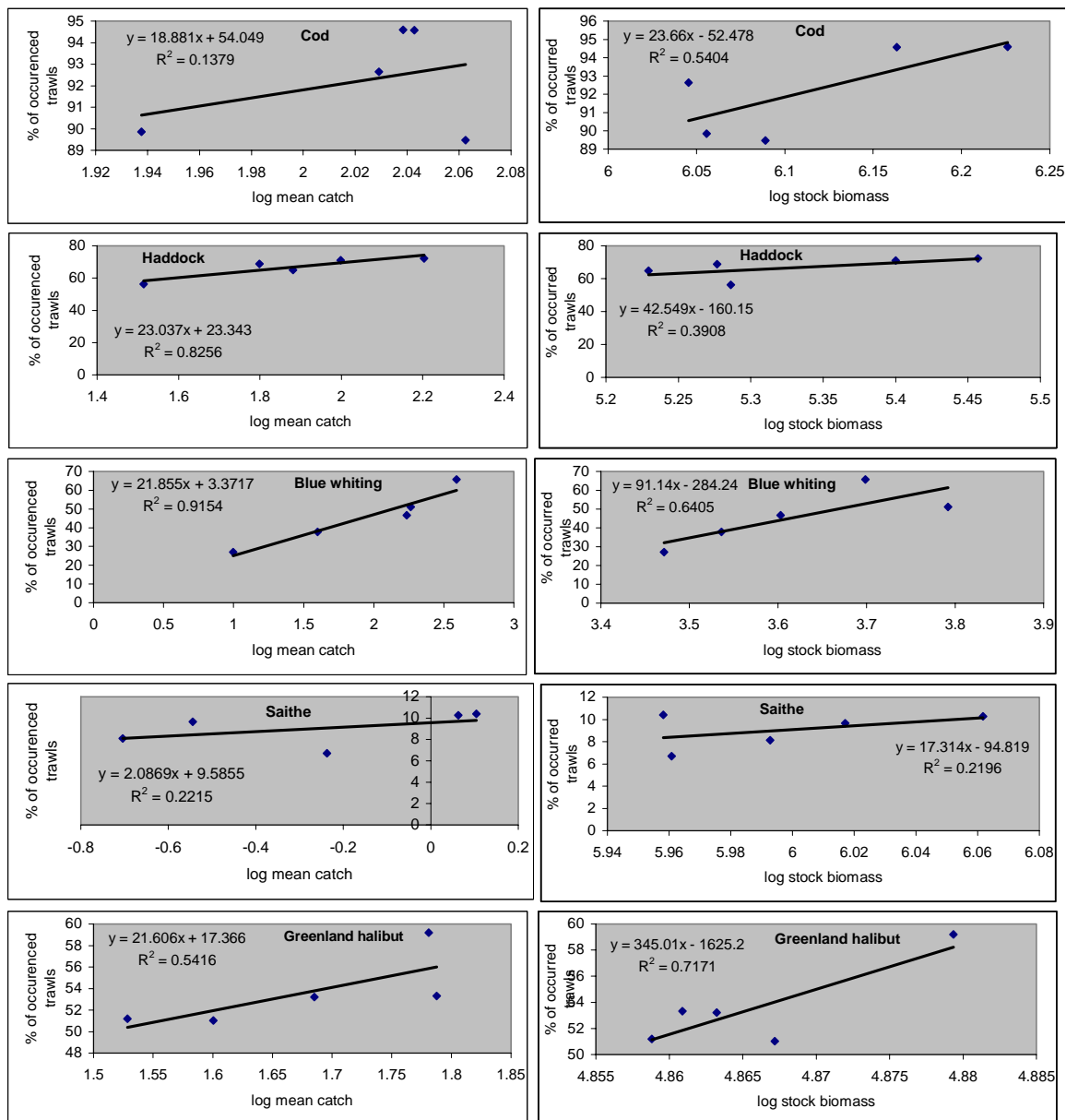
Figure 7.13: Relationships between distribution range and stock level in the most important commercial fish species of the Barents Sea.
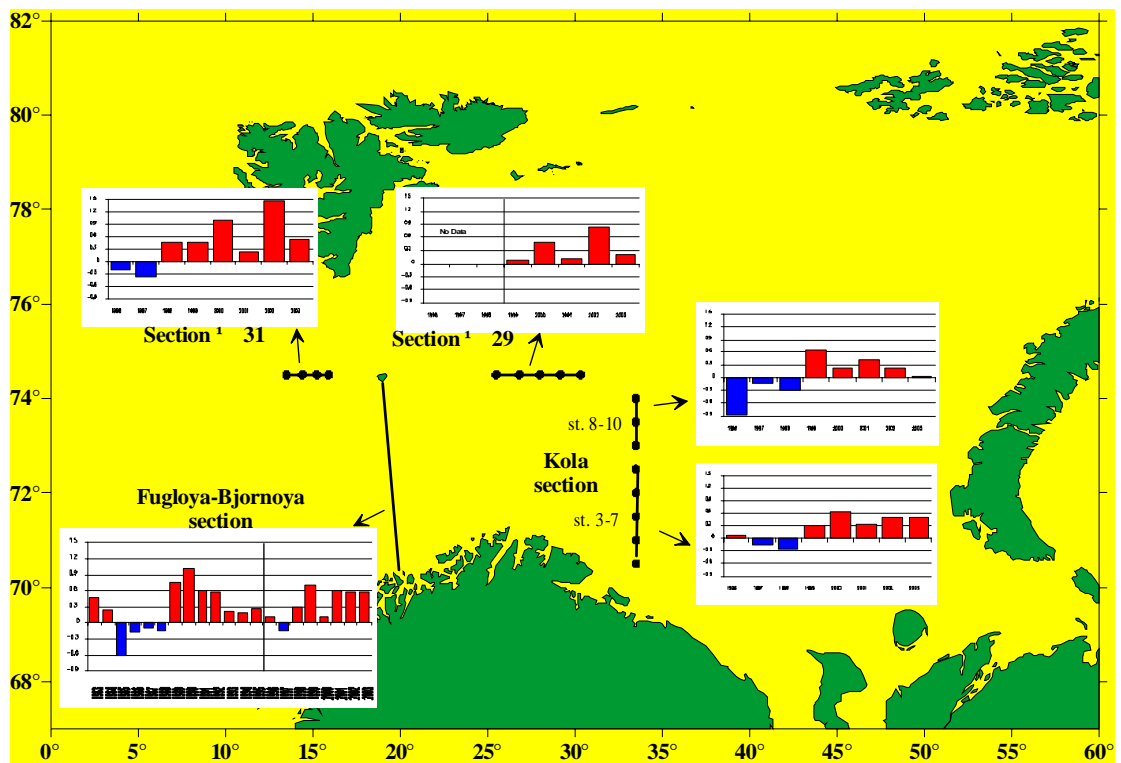
Figure 7.14: Standard oceanographic sections in the Barents Sea showing temperature anomalies (variance from the long-term mean bottom water temperature)

7.3   Processes and mechanisms

The macroecological comparative approach has generated a number of repeated patterns, and has provided some insight into the underlying mechanisms (Gaston and Blackburn, 2000). Three mechanisms for explaining abundance-distribution relationships have been explored previously: vital rates, density dependence and metapopulation. Predictions from habitat filling models include;

- For species with low colonisation rates, there should be no abundance-occupancy relationship;
- For species with high colonisation ability, a simple hyperbolic abundance-occupancy relationship is predicted; and
- Species with moderate colonisation ability will exhibit metapopulation dynamics and a minimum density exists below which populations cannot persist.

This suggests that abundance–occupancy relationships are important by linking local and regional population processes and thus allowing local and regional impacts to be explored simultaneously (Watkinson *et al.*, 2003). It is valuable to consider basic elements driving the spatial dynamics of fish in order to understand the dynamics of abundance-distribution relationships. Species distributions are affected by habitat availability, migratory and dispersal capabilities, predator-prey interactions and environmental conditions. A conceptual framework for understanding the spatial dynamics of fish is shown in Figure 7.15. During the early life stages of broadcast spawners, distribution is mainly a function of the initial spawning position and subsequent transport processes. As the fish grows it increasingly controls its own movement and distribution and migration needs to be viewed in light of individual responses to the physical environment, as well as the internal states of individuals (Figure 7.15). The behavioural responses are eventually evaluated in light of their ultimate function. Thus fish responses will vary not only between species, but also between different life stages and envi-

ronmental conditions, which varies both seasonally and inter- annually. Thus while density dependence is likely to be an important factor in the distribution of fish, there are a number of other processes that will impact on the distribution of fish and that may preclude simple abundance-distribution relations.

Ultimate causes:
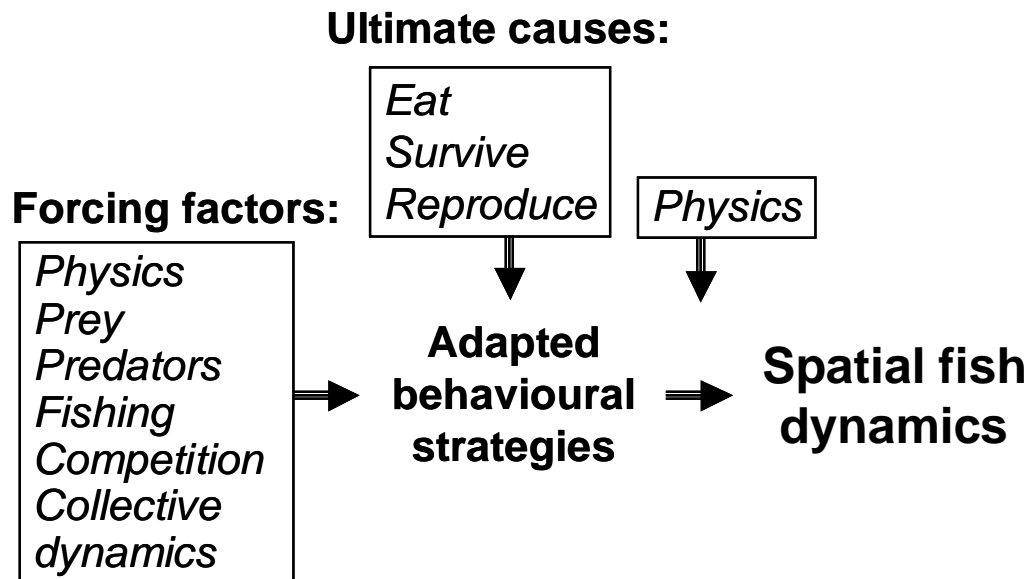
Figure 7.15: A conceptual illustration of important factors underlying the spatial dynamics of fish.

While natural mortality is recognised as important for the removal of prey from the population, predators may also impact on prey dynamics by scaring them: this is referred to as "the ecology of fear" (Brown *et al.*, 1999). In cases where both the predator and the prey are able to relocate, the fear of predation may be more important for the interaction between predators and prey than the actual predation. Such "fear" inflicts costs through prey spending time on vigilance instead of eating, and in sub-optimal food patches that have reduced predation risk. The northwards migration of the Barents Sea capelin *Mallotus villosus* has been suggested to be a case of "the ecology of fear" (Fiksen *et al.*, 1995; Huse, 1998). Thus in such cases where refuges exist, the degree of prey aggregation in the refuge may be more a function of predator density than of prey density.

As indicated in Figure 7.15, fish movements are influenced both by passive transport, as well as active movement behaviour. Active movements can further be grouped into migrations and habitat choice, where the former can be defined as the periodic passage of groups of animals from one region to another. Habitat choice on the other hand is a more local process involving selecting between patches, and the specific strategy employed will depend on whether the motive is growing, surviving, reproducing or combinations of these actions. There have been theoretical studies addressing fish movements based on life history theory (see reviews by Tyler and Rose, 1994; and Giske *et al.*, 1998). Under this theory individuals choose habitat and migration paths in order to maximise their Darwinian fitness. This entails trading off a number of factors related to growth and survival, including predator and prey distributions, temperature, light, number of conspecifics, etc. In obligate schooling species such as herring, the decisions made collectively at the school level can be in conflict with individual fitness maximisation. Collective decision making and leadership can therefore have a strong affect on the spatial dynamics of fish such as herring (Fernö *et al.*, 1998; Huse *et al.*, 2002) and, to a lesser degree, cod (Rose, 1993).

In summary it is important to view the spatial dynamics of fish as an interplay between environmental factors, individual behavioural, collective behaviour moulded by the ultimate motivation. This framework for addressing spatial fish dynamics has implications for expectations

about abundance-distribution relationships. Thus when using simple metrics such as abundance-distribution relationships it is important to keep in mind the processes that might confound simple density dependent distributions.

7.4 Utility of abundance- occupancy relationships for management

Obviously the management strategies required for fisheries targeting widespread species and those fisheries targeting species with more restricted distributions could be very different. The two case studies on thorny skate and Norway pout above illustrate different patterns in abundance-distributions relation. While Norway pout showed a general decrease both in abundance and distribution, thorny skate showed an increase in density in some places, a phenomenon referred to as hyper- aggregation (Rose and Kulka, 1999). This may make thorny skate more vulnerable to rapid stock collapses, as seen previously for northern cod, while reduced Norway pout abundance is accompanied by a similar decrease in CPUE and provides diminishing returns for fishers. Whether the current level of exploitation of thorny skate stock is sustainable is uncertain. Biomass has remained stable for about 12 years, but concentration (hyperaggregation) of the stock into small, accessible areas can make them increasingly susceptible to over-exploitation. Potential management considerations under such circumstances, if the stock is deemed depressed, could be an area closure. In this case, closing a relatively small area could potentially have a large benefit to the stock because the fish are so highly concentrated.

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8   Evaluation of the decline criterion

8.1   Introduction

WGFE had the following ToR: to "Evaluate the decline criteria used by existing nature conservation organisations, and address any upcoming nature conservation issues for marine fishes".

Previous work by WGFE detailed threat and decline metrics used by international, regional and national organisations, and qualitatively critiqued the Texel-Faial criteria (Dulvy *et al.*, 2004; ICES, 2004). The Texel-Faial criteria were proposed by OSPAR to assess 'which species and habitats need to be protected and those human activities that are likely to have an actual or potential adverse affect on these species and habitats or on ecological processes' within the OSPAR area.

More generally there is a need to understand how threat metrics compare to the fisheries management reference points to determine the degree to which threat metrics support or conflict with fisheries management reference points (Musick, 1999; Powles *et al.*, 2000). Threat listing of exploited marine species has been controversial because of the scientific uncertainty of extinction risk as well as the social, economic and political costs of management procedures that may be triggered by designating a species as threatened. For example, in Canada a species listed as threatened by the Species at Risk Act (SARA), may result in the closure of fisheries impacting the threatened species, with high associated social and economic costs (see Section 8.4). Clearly in such situations overly sensitive threat criteria run the risk of raising *false alarms* that would result in unnecessarily large social and economic impacts. There have been limited comparisons of threat metrics and some studies suggest decline criteria may be prone to *false alarms* (Matsuda *et al.*, 1998; Punt, 2000), others studies suggest a high rate of decline is associated with a low probability of recovery and also possibly consistent with high risk of extinction (Beverton, 1990; Hutchings, 2001b).

8.2   Evaluation of the decline criterion

Recent work has compared the widely used World Conservation Union (IUCN) decline criterion A, and American Fisheries Society (AFS) decline criteria, with population viability analyses (PVA) and ICES ACFM stock status (Dulvy *et al.* working paper 1). Decline criteria and PVA were calculated using time series of numbers at age and spawner-recruit data for 76 commercially exploited Northeast Atlantic ICES stocks, comprising 62 stocks from 21 fish species: anglerfish *Lophius* spp. (2 stocks), anchovy *Engraulis encrasicolus*, blue whiting *Micromesistius poutassou*, cod *Gadus morhua* (11), haddock *Melanogrammus aeglefinus* (5), hake *Merluccius merluccius* (2), Greenland halibut *Reinhardtius hippoglossoides*, herring *Clupea harengus* (11), horse mackerel *Trachurus trachurus*, mackerel *Scomber scombrus*, megrim *Lepidorhombus* spp. (3), Norway pout *Trisopterus esmarki*, plaice *Pleuronectes platessa* (5), saithe *Pollachius virens* (4), sandeel, sardine *Sardina pilchardus*, sprat *Sprattus sprattus*, sole *Solea solea* (6), and whiting *Merlangius merlangus* (4); and 14 stocks from three invertebrate species [Solent oyster *Ostrea edulis*, Northern prawn *Pandalus borealis*, and Norway lobster *Nephrops norvegicus* (12). Threat assessments and fisheries management assessments were compared using the *hits*, *misses* and *false alarm* framework derived from signal detection theory (ICES, 2004; Piet and Rice, 2004; Rice, 2003). There are two types of *hit*: a true positive and a true negative, resulting in a 2 x 2 table of outcomes.

Within this framework:

- a true positive *hit* is where a stock is exploited unsustainably (outside safe biological limits) and the threat criterion is met;

- a *miss* is where the species is exploited unsustainably (outside safe biological limits) but does not meet the threat criterion;
- a true negative *hit* is where a stock is exploited sustainably (within safe biological limits) and the threat criterion is not met, and
- a *false alarm* is where the stock is exploited sustainably (within safe biological limits) but the threat criterion is (erroneously) met.

A good set of criteria should avoid *false alarms* and minimise *misses* (Rice, 2003).

A total of 19 stocks met one or more threat criteria (Table 8.1). Not all threat criteria could be applied to all stocks, while decline criteria could be calculated for most stocks (IUCN A = 64, AFS = 58), the PVA methods could be applied to relatively few stocks (IUCN E – Dennis, Munholland and Scott (1991) method = 11, IUCN E – Holmes (2001) = 12). The PVA methods assume populations do not exhibit density dependence and could only be applied to stocks with low variance and exponential declines or increases.

Table 8.1: Stocks meeting one or more of IUCN A, AFS and IUCN E threat criteria. IUCN E threat criteria were undertaken using the methods of Dennis *et al.* (1991) and Holmes (2001). CR = Critically endangered; EN = Endangered, VU = Vulnerable.

STOCK	IUCN A	AFS	IUCN E(D)	IUCN E(H)
Cod (E Baltic)	EN			
Cod (Irish Sea)	VU			
Cod (Kattegat)			CR	CR
Cod (North Sea)	EN			
Cod (Norway coast)	VU		CR	CR
Cod (W Scotland)	VU			
Greenland Halibut	EN			
Hake (Iberia)			CR	CR
Hake (Northern)				CR
Herring (E. Baltic)			CR	
Herring (Iceland spring)	CR	VU		
Herring (SW Scot.)	EN			
Herring (W. Baltic)	VU			
Nephrops (Bay of Biscay)				CR
Nephrops (N Galicia)	VU		CR	CR
Nephrops (S Port.)				CR
Nephrops (W Galicia+N Port.)	EN			
Sole (Baltic)	VU			

A total of 46 (67%) stocks were outside safe biological limits (exploited unsustainably) and 23 (33%) were inside safe biological limits (exploited sustainably). None of the threat methods raised *false alarms*, and stocks that were categorized as threatened had also been categorized by ICES ACFM as being exploited "outside safe biological limits" (Table 8.2).

The study concluded that threat criteria and fisheries stock assessments provide comparable information on the status of populations exploited by NE Atlantic European fisheries (Dulvy *et al.*, Working paper 1). There was no evidence that the application of threat criteria would raise *false alarms*: none of the stocks identified as threatened were classed as exploited within safe biological limits by ICES. These results lead to two conclusions. First, in every case where a stock was classified as threatened, they were also regarded as being unsustainably exploited 'outside safe biological limits'. Second, stocks identified as 'outside safe biological limits' may, in some cases, also be considered threatened. Thus management advice to reduce fishing mortality on stocks outside safe biological limits is consistent with the requirement to reduce the risk of extinction and reductions in fishing mortality should meet the concerns of both fisheries and conservation interests.

Table 8.2: The proportion (%) of stocks meeting each of four possible outcomes (true positive *hit*, true negative *hit*, *miss* and *false alarm*) and the total number of stocks for which both stock status and threat status were available. IUCN E$^{(D)}$ was calculated using the Dennis *et al.* (1991) method and IUCN E$^{(H)}$ was calculated using the Holmes (2001).

THREAT CRITERIA	FALSE ALARM	HIT (POSITIVE)	HIT (NEGATIVE)	MISS	NUMBER OF STOCKS COMPARED
IUCN A1	0	16	36	48	64
AFS	0	2	36	62	58
IUCN E$^{(D)}$	0	36	45	18	11
IUCN E$^{(H)}$	0	50	25	25	12

This analysis supports suggestions that AFS criteria decline rate thresholds are overly conservative and may overlook threatened species (Hutchings, 2001a). Apart from the Iceland spring spawning herring, the AFS criteria overlooked all other stocks exploited outside of safe biological limits and also those categorised as threatened by other quantitative approaches. The AFS criteria assume high resilience in teleost fishes from high interannual variability and reproductive output, but these assumptions have no theoretical or empirical basis (Denney, Jennings and Reynolds, 2002; Dulvy, Sadovy and Reynolds, 2003; Hutchings, 2001a; Hutchings and Reynolds, 2004; Sadovy, 2001).

While the Texel-Faial criteria include a decline criterion, the thresholds are undefined and hence could not be applied to the stocks considered here. Such criteria often have a weaker theoretical grounding than the existing IUCN criteria, and given the apparently reliable performance, international recognition and peer-review of the IUCN criteria and the consistency with which IUCN criteria reflect ICES fisheries management stock assessments, there is little reason to invent new threat criteria without a strong scientific case (Dulvy *et al.*, Working paper 1).

8.3 EcoQOs for threatened and declining fish species

One of the 10 OSPAR Ecological Quality issues is "Threatened and declining species" and the element is, "presence and extent of threatened and declining species in the North Sea". The most obvious objective for threatened and declining species is to reverse the downward trend in abundance and/or extent. In order to accomplish this, there will need to develop simple, effective and relevant metrics, and agreed objectives that deliver a year-on-year assessment in population or habitat status with which to monitor improvements. To achieve this, there will also need to be suitable monitoring and assessment methods, which have sufficient statistical power to show statistically significant improvement (ICES, 2003). To date an Ecological Quality objective has yet to be considered.

The initial list of fish species identified by OSPAR as 'threatened and declining' comprises 13 species (Table 8.3), and the statistical power for monitoring trends for many of these species

in groundfish surveys will be low, though for many of the diadromous fish species more appropriate sampling programmes for estuarine and fluvial stages may be available.

Table 8.3: Table of fish species listed by OSPAR as 'Threatened and Declining' (Adapted from OSPAR (2004) Initial OSPAR List of Threatened and/or Declining Species and Habitats, Reference Number: 2004–06)

SCIENTIFIC NAME	COMMON NAME	OSPAR REGIONS WHERE THE SPECIES OCCURS	OSPAR REGIONS WHERE THE SPECIES IS UNDER THREAT AND/OR IN DECLINE
Petromyzon marinus (Linnæus, 1758)	Sea lamprey	I, II, III, IV	All where it occurs
Cetorhinus maximus (Gunnerus, 1763)	Basking shark	All	All where it occurs
Dipturus batis (Linnæus, 1758)	Common Skate	All	All where it occurs
Raja montagui (Fowler, 1910)	Spotted Ray	II, III, IV, V	All where it occurs
Acipenser sturio (Linnæus, 1758)	Sturgeon	II, IV	All where it occurs
Alosa alosa (Linnæus, 1758)	Allis shad	II, III, IV	All where it occurs
Coregonus lavaretus oxyrinchus (Linnæus, 1758)	Houting	II	All where it occurs
Salmo salar (Linnæus, 1758)	Salmon	I, II, III, IV	All where it occurs (the status of different stocks is still to be taken into account)
Gadus morhua (Linnæus, 1758)	Cod	All	II, III (Populations in OSPAR regions II and III, including North Sea and Skagerrak cod stock, Kattegat cod stock, Cod west of Scotland, Cod in the Irish Sea, Cod in the Irish Channel and Celtic Sea)
Hoplostethus atlanticus (Collett, 1889)	Orange roughy	I, V	All where it occurs
Hippocampus guttulatus (Cuvier, 1820) (synonym: *Hippocampus ramulosus*)	Long-snouted seahorse	II, III, IV, V	All where it occurs
Hippocampus hippocampus (Linnæus, 1758)	Short-snouted seahorse	II, III, IV, V	All where it occurs
Thunnus thynnus (Linnæus, 1758)	Bluefin tuna	V	All where it occurs

Here new and ongoing work on a possible threat index for the North Sea fish assemblage is briefly summarised, full details are available in Dulvy *et al*. (Working paper 2). Recent work highlights the consistency of decline criteria with fisheries management reference points (Section 8.2) and highlights the suitability of the IUCN decline criteria A over the American Fisheries Society criteria and Texel-Faial criteria for assessing threat and decline. Here the IUCN A criteria have been applied to the North Sea fish assemblage over time (Dulvy *et al*., Working paper 2, Annex 2). Decline rates were calculated for 23 North Sea fish species and used to determine how many species were threatened and the degree of threat (*Critically Endangered*, *Endangered*, *Vulnerable*) in any given year. Decline rate was calculated from adult abundance estimates at start and end points estimated from a linear model fitted to portions of each species time series. Two methods of assessing decline were assessed: *extent of decline*, where the decline is assessed from the start of the time series compared to the latest survey year and the *rate of decline*, where the decline is calculated from the latest survey year and *x* years prior to the latest survey year, where *x* represents 10, 12 and 15-year moving time windows (Figure 8.1). The only possible advantage of the rate of decline over the extent of decline approach is that the threat indicator is much more sensitive to recovery. However this advantage is outweighed by the problem of a shifting baseline - where the baseline represents an increasingly exploited state over time, and may mask the true magnitude of fishing effects on biodiversity (Pauly 1995).
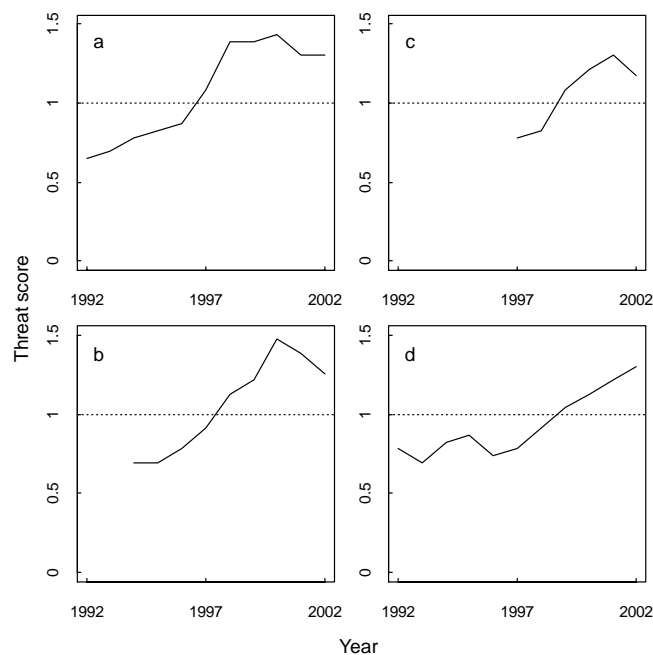
Figure 8.1: An indicator of threat over time for a suite of North Sea demersal fishes measured as, (a) rate of decline with 10-yr window, (b) rate of decline with 12-yr window, (c) rate of decline with 15-yr window and (d) extent of decline. A score of 1 is equivalent to each species meeting the *Vulnerable* criterion and is indicated with a dotted line.

Individual species threat categorisations were scored as *Critically Endangered* = 3, *Endangered* = 2 and *Vulnerable* =1 (Baillie, Hilton-Taylor and Stuart, 2004). The composite threat index was averaged across species for each year. The threat index score varies from 0 to 3. This index is readily interpreted; a score of 0 is equivalent to none of the species meeting any of the threat criteria, and a score of 3 is equivalent to each species being *Critically Endangered*.

To date power analysis has not been conducted on this composite threat index. While the choice of a suitable EcoQO for threat and declining species depends largely on the degree of

public interest in threat it is simpler to suggest an acceptable reference direction (Jennings and Dulvy, in press). It is widely agreed that threat in the marine environment is undesirable (Hutchings and Reynolds, 2004; Jennings, 2004) and a defensible reference direction would be to minimise the index score. If threat and decline were unacceptable by the general public the EcoQO or reference point should be zero. This may be appropriate given the losses of populations that have already occurred in parts of the North Sea (Dulvy *et al.*, 2003; Walker and Heessen, 1996; Wolff, 2000). If public concern for threat and decline were lower a less stringent reference point of 0.25 or 0.5 could be set. Such threat index values would occur if one quarter (8 species) or a half (15 species) of the assemblage qualifies as *Vulnerable* (i.e., population decline rate between ≥50% and <70% over 10 years), or if fewer species met higher threat criteria. A limit reference point that should not be exceeded is a composite threat score of 1. This is based on previous findings that fish species qualifying as threatened under the IUCN A decline criteria must also be exploited beyond safe biological limits (Dulvy et al., in press). Consequently a composite threat score of 1 indicates that all species are unsustainably exploited.

Good indicators have a number of properties, including being: readily interpreted, measurable, accurate/precise, representative of the system, specificity to forcing factor, able to set reference points, sensitivity, responsiveness, and sound theoretical basis (Piet and Jennings, 2005; Rice, 2003). This threat index meets many of these indicator selection criteria, it is: readily interpreted, relatively easily measured and data are readily available, fairly accurate/precise, representative, sensitive, responsive, specific to exploitation and reference points and directions can be set within a well-understood theoretical framework.

8.4   Canadian Species at Risk Act (SARA)

In Canada, since 1978, the role of COSEWIC (Committee on the Status of Endangered Wildlife in Canada) has been to assess the status of species suspected to be at risk nationally, to report the list of species at risk and its findings to the Canadian Endangered Species Conservation Council, and subsequently to make the list public. This Committee uses criteria very similar to those employed by the IUCN (see ICES, 2004):

- Endangered Species - a wildlife species that is facing imminent extirpation or extinction
- Threatened Species - a wildlife species that is likely to become an endangered species if nothing is done to reverse the factors leading to its extirpation or extinction

where the "threatened" and "endangered" categories refer specifically to imminent extirpation or extinction (see www.cosewic.gc.ca and ICES (2004) for further information).

The Species at Risk Act (SARA, Canada Gazette, Statutes of Canada, 2002, Pt. 3, Vol. 25, no 3, Ch. 29. http://canadagazette.gc.ca), proclaimed in June 2003, for the first time affords legal protection for listed wildlife in Canada. The Act requires the imposition of automatic prohibitions on the killing and harming of species on **Schedule 1**, those species legally listed as "threatened" or "endangered". Thus, starting 1 June 2004, Sect. 32 (1), General Prohibitions, states that "no person shall kill, harm, harass, capture or take an individual of a wildlife species that is listed as an extirpated species, an endangered species or a threatened species". Fines imposed under the Act for harm or possession of an individual are substantial. The socio-economic implication associated with such widespread species such as the wolffish *Anarhichas denticulatus* and spotted wolfish *A. minor* classified by COSEWIC as "threatened" may be large given their widespread distribution and their incidental occurrence in most Canadian Atlantic fisheries. Previously, the classification of risk by COSEWIC did not carry the associated prohibitions and associated socio-economic implications tied to the legal listing of a species under the Act.

However, Sect. 73 (1) also provides for exceptions where: "The competent Minister may enter into an agreement with a person, or issue a permit to a person, authorizing the person to engage in an activity affecting a listed wildlife species, any part of its critical habitat or the residences of its individuals", subject to certain conditions. Effectively, this means that the SARA legislation provides the competent Minister the option to issue a **Permit** to allow for unavoidable "incidental harm" to a listed species providing that certain conditions are met, as has been done for wolffish.

The issuance of a Permit allowing harm is predicated on demonstrating that incidental capture will not impair or prevent recovery of the species and that all possible actions are being taken to minimize the harm. In the case of wolffish specifically, live release has been chosen as the key permit criterion and this action allows recovery to occur. However, the implications of this are unclear for other marine fish species, such as cod, which generally do not survive capture. For such species, legal listing based on COSEWIC recommendations could have substantial consequences in the form of widespread fishery closures. Under such circumstances, careful consideration must be given to the application of the COSEWIC decline criteria in regard to determining if a species is truly at risk of biological extinction.

The current list of Atlantic marine fish that are designated or under review by COSEWIC is summarized in Table 8.4. In terms of marine fishes, only the three wolffish species are presently on Schedule 1 of the Act (i.e., legally listed.) (see: http://www.sararegistry.gc.ca/). Some estuarine and diadromous populations of species such as striped bass *Morone saxatilis,* Atlantic Salmon *Salmo salar* and Pacific Salmon *Oncorhynchus* spp. have also been listed by COSEWIC and, in the Pacific Ocean, the bocaccio *Sebastes paucispinis* has been listed as threatened.

Table 8.4: Current list of Atlantic fishes that are designated or under review by COSEWIC (adapted from http://www.cosewic.gc.ca)

STATUS	SPECIES	POPULATIONS	TREND	RECOVERY STRATEGY	ACTION PLAN
Endangered	Porbeagle *Lamna nasus*	Atlantic	Stable	No	No
	Atlantic Cod *Gadus morhua*	Newfoundland / Labrador	Stable	No	No
	Atlantic Cod *Gadus morhua*	Laurentian North	Stable	No	No
Threatened	Cusk *Brosme brosme*	Scotian Shelf	?	No	No
	Northern Wolffish Anarhichas denticulatus	Atlantic	Stable	Yes (draft)	Yes (Draft)
	Spotted Wolffish *Anarhichas minor*	Atlantic	Increase	Yes (draft)	Yes (Draft)
Special Concern	Atlantic Cod *Gadus morhua*	Maritimes	Stable	No	No
	Striped Wolffish *Anarhichas lupus*	Atlantic	Increase	Yes (draft)	Yes (Draft)
Under Review	White shark *Carcharodon carcharias*	Atlantic		No	No
	Basking shark *Cetorhinus maximus*	Atlantic		No	No
	Blue shark *Prionace glauca*	Atlantic		No	No

STATUS	SPECIES	POPULATIONS	TREND	RECOVERY STRATEGY	ACTION PLAN
	Spinytail skate *Bathyraja spinicauda*	Atlantic		No	No
	Thorny skate *Amblyraja radiata*	Atlantic		No	No
	Barndoor skate *Dipturus laevis*	Atlantic		No	No
	Winter skate *Leucoraja ocellata*	Atlantic		No	No
	Spiny Eel Notocanthus chemnitzi	Atlantic		No	No
	Blue hake *Antimora rostrata*	Atlantic		No	No
	White hake *Urophysis chus*	Atlantic		No	No
	Roundnose grenadier *Cory-phaenoides rupestris*	Atlantic		No	No
	Roughead grenadier Macrourus berglax	Atlantic		No	No
	Acadian redfish *Sebastes mentella*	Atlantic		No	No

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9   Food rations and prey composition of North Sea fishes

9.1   Re-evaluation of MSVPA interactions

New data on gastric evacuation of different crustacean prey (crabs and shrimps) with a robust exoskeleton are now available. The evacuation of these prey types can be coupled to general prey characteristics. The results will therefore be used during 2005 to improve the new mechanistic evacuation model, as described in previous reports (ICES, 2003, 2004), as a generic tool for the prediction of evacuation of invertebrate prey as well as fish prey. Haddock feeds on a variety of invertebrate prey, as well as fish prey. The food ration and diet composition of this gadoid predator needs, therefore, to be revised when the new model improvement is accomplished. Accordingly, relevant runs of MSVPA/FOR to examine the consequences when using revised estimates of food ration and diet composition of the MSVPA-predators, rather than the ones used at present by ICES, has been postponed until revised feeding estimates are available for haddock.

9.2   Food rations of grey gurnard

Acquisition of sufficient information for parameterisation of the new evacuation model for grey gurnard has been delayed. The information is, however, available by now and a full model covering the main variables affecting evacuation in this important North Sea will be established in 2005 so that it is possible to estimate food ration and diet composition before the next meeting in 2006.

9.3   Future studies

It is recommended that this TOR is postponed until the 2006 meeting.

9.4   References

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10 Diet, feeding habits and foraging behaviour of fishes in the North Sea and adjacent waters

10.1 Introduction

Studies on the feeding habits of marine fishes have been published regularly in the scientific literature since the late 1800's (e.g., Smith, 1889–1892; Scott, 1903, 1903; Todd, 1903, 1905, 1907; Blegvad, 1916; Larsen, 1936), though many early studies were more descriptive and the diets were not always fully quantified. There are now many published studies quantifying the diets of fishes in the eastern North Atlantic, and a preliminary summary of the dietary studies of fishes in the North Sea and adjacent waters is given in Table 10.1.

The most comprehensive database of stomach contents of North Sea fish is the International Stomach Content Data Base, maintained by ICES (Daan *et al.*, 1989; ICES, 1997). Sources of information for stomach contents data for species included in the year of the stomach and additional species for the North Sea, Celtic Sea, Barents Sea, Baltic Sea, northern Spanish shelf and other regions is summarised in Table 10.1. In the "Atlas of Sea Fishes", Knijn *et al.* (1993) reported on the diet composition of 98 species. Diet compositions, in terms of numbers and percentage weight, and estimates of daily consumption rates are available for cod, haddock, whiting, saithe, mackerel, horse mackerel, Norway pout, plaice and herring in the North Sea (Greenstreet, 1996). Studies off the northern Spanish shelf have provided diet compositions, in terms of numbers and percentage volume, for 25 predatory fish species from autumnal surveys conducted since 1988. These data include information on the length of fish prey and size of decapod crustacean prey.

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Table 10.1: Published studies on the diets of selected fishes in the North Sea and adjacent regions

SPECIES	COMMON NAME	AREA	SOURCE
ELASMOBRANCHII			
Lamna nasus	Porbeagle	All areas combined	Scott (1902); Aasen (1961); Stevens (1973); Capapé (1975d); Ellis and Shackley (1995)
Alopias vulpinus	Thresher shark	English Channel	Pascoe (1986)
Squatina squatina	Angel shark	All areas combined	Todd (1903); Ellis *et al*. (1996); Capapé (1975d); Azouz and Capapé (1971);
Squalus acanthias	Spurdog	North Sea	Todd (1905); Todd (1907); Blegvad (1916); Scott (1903); Todd (1903); Holden (1965); Bennet (1967); Rae (1967);
		Western areas	Ellis *et al*. (1996); Holden (1965); Rae (1967);Ford (1921); Symonds and Elson (1983)
		Other areas	Bowman *et al*. (1984);. Ebert *et al*. (1992);. Hanchet (1991); Jones and Geen (1977); Rae (1967); Robinson *et al*. (1982); Tanasichuk *et al*. (1991); Jardas (1972);
Galeus melastomus	Blackmouth dogfish	North Sea	Mattson (1981); Rae and Shelton (1982)
		Western areas	Mauchline and Gordon (1983)
		Mediterranean	Relini Orsi and Wurtz (1975); Capapé and Zaouali (1976); Carrassón *et al*. (1992); Capapé (1975d); Azouz and Capapé (1971); Macpherson (1980)
		Northern Spain shelf	Olaso and Rodríguez-Marín (1995); Velasco *et al*. (1996); Serrano *et al*. (2003a, b); Olaso *et al*. (2005)
Scyliorhinus canicula	Lesser-spotted dogfish	North Sea	Todd (1905); Todd (1903);Rae and Shelton (1982);
		Western areas	Gibson and Ezzi (1987); Scott (1903); Ellis *et al*. (1996); Eales (1949); Lyle (1983); Kaiser and Spencer (1994); Ford (1921); Steven (1930); Symonds and Elson (1983)
		Mediterranean	Capapé (1974); Lazzaretto (1964); Azouz and Capapé (1971); Jardas (1972)
		Northern Spain shelf	Olaso and Rodríguez-Marín (1995); Velasco *et al*. (1996); Olaso *et al*. (1998); Olaso *et al*. (2002); Serrano *et al*. (2003a, b); Preciado *et al*. (2003); Olaso *et al*. (2005)
Scyliorhinus stellaris	Greater-spotted dogfish	All areas combined	Ellis *et al*. (1996); Capapé (1975c); Ford (1921); Lazzaretto (1964); Azouz and Capapé (1971);
Galeorhinus galeus	Tope	North Sea	Todd (1905); Todd (1907);
		Western areas	Ellis *et al*. (1996); Ford (1921)
		Mediterranean	Capapé (1975d);
Mustelus spp.	Smooth hounds	All areas combined	Ellis *et al*. (1996); Ford (1921); Capapé (1975d); Lazzaretto (1964); Azouz and Capapé (1971);
Amblyraja radiata	Starry ray	North Sea	Blegvad (1916); Scott (1902); Scott (1903); Smith (1890); Daan *et al*. (1993); Rae and Shelton (1982); ICES (1997)
		Other areas	McEachran *et al*. (1976); Tyler (1971); Templeman (1982); Pedersen (1995); Bjelland *et al*. (2000); Skjæraasen and Bergstad (2000)
		Barents Sea	Zenkevich and Brotskaya (1931); Antipova and Nikiforova (1983, 1990); Berstovsky (1989); Dolgov (1997, 2005)
Dipturus batis	Common skate	North Sea	Todd (1905); Todd (1907); Ray (1914); Blegvad (1916); Rae and Shelton (1982);
		Western areas	Herdman and Scott (1895); Steven (1932); Du Buit (1968).

SPECIES	COMMON NAME	AREA	SOURCE
		Scottish waters	Scott (1902); Scott (1903); Smith (1889); Smith (1891);
		Barents Sea	Dolgov (2005)
Leucoraja circularis	Sandy ray	Western areas	Du Buit (1968)
Leucoraja fullonica	Shagreen ray	All areas combined	Scott (1902); Scott (1903); Rae and Shelton (1982); Du Buit (1968);
Leucoraja naevus	Cuckoo ray	All areas combined	Herdman and Scott (1895) (as R.circularis); Scott (1902) (as R.circularis); Smith (1891); Daan *et al.* (1993); Ellis *et al.* (1996); Holden and Tucker (1974); Rae and Shelton (1982); Cunha *et al.* (1986); Du Buit (1968); Marques and Ré (1978); Steven (1930);
		Northern Spain shelf	Serrano *et al.* (2003b); Velasco *et al.* (In prep.)
Raja brachyura	Blonde ray	All areas combined	Todd (1905); Todd (1903); Ellis *et al.* (1996); Holden and Tucker (1974); Rae and Shelton (1982); Cunha *et al.* (1986); Marques and Ré (1978); Quiniou and Andriamirado (1979); Steven (1930);
Raja clavata	Thornback ray	All areas combined	Todd (1905); Todd (1907); Herdman and Scott (1895); Ray (1914); Blegvad (1916); Gibson and Ezzi (1987); Scott (1902); Smith (1889); Smith (1890); Smith (1891); Daan *et al.* (1993); Ellis *et al.* (1996); Ajayi (1977, 1982); Holden and Tucker (1974); Rae and Shelton (1982); Abdel-Aziz (1986); Capapé (1975b); Cunha *et al.* (1986); Du Buit (1968); Ebert *et al.* (1991); Fitzmaurice (1974); Marques and Ré (1978); Macpherson (1986); Nottage and Perkins (1978); Nottage and Perkins (1980); Quiniou and Andriamirado (1979);. Smale and Cowley (1992); Lazzaretto (1964); Steven (1930); Azouz and Capapé (1971); Jardas (1972); Ebeling (1988)
		Northern Spain shelf	Olaso and Rodríguez-Marín (1995); Velasco *et al.* (1996); Velasco *et al.* (In prep.)
Raja microocellata	Smalleyed ray	Western areas	Ajayi (1982); Rousett (1987);
Raja montagui	Spotted ray	North Sea	Todd (1905); Todd (1907); Smith (1889); Daan *et al.* (1993); Holden and Tucker (1974); Rae and Shelton (1982);
		Western areas	Herdman and Scott (1895); Smith (1889); Ellis *et al.* (1996); Ajayi (1982); Holden and Tucker (1974); Cunha *et al.* (1986); Du Buit (1968); Marques and Ré (1978); Quiniou and Andriamirado (1979); Steven (1930);
		Northern Spain shelf	Velasco *et al.* (1996); Velasco *et al.* (In prep.)
Rostroraja alba	White skate	Mediterranean	Azouz and Capapé (1971)
Dasyatis pastinaca	Sting ray	Mediterranean	Capapé (1975a)
Myliobatis aquila	Eagle ray	Mediterranean	Capapé (1976a)
Pteromylaeus bovinus	Bull ray	Mediterranean	Capapé (1976b)
GADIFORMES			
Gadiculus argenteus	Silvery Pout	Norwegian Deep	Albert (1993); Matson (1981)
		Mediterranean Sea	Macpherson (1978)
Gadus morhua	Cod	North Sea	Daan (1973); (1989); ICES (1988, 1997); Dahl and Kirkegaard (1986); Cranmer (1986); Rae (1967); Floeter and Temming (2003)
		Southern Bight	Bromley (1995)
		Irish Sea	Armstrong (1982)
		Baltic Sea	Hertling (1928)

SPECIES	COMMON NAME	AREA	SOURCE
		Celtic Sea	Pinnegar *et al* (2003); Du Biut (1995)
		Kattegat	Pihl (1994)
		Barents Sea	Zenkevich and Brotskaya (1931); Zatsepin and Petrova (1939); Grinkevich (1957); Ponomarenko and Yaragina (1996); Bogstad and Mehl (1996); Dolgov (2000)
Melanogrammus aeglefinus	Haddock	North Sea	Daan (1989); ICES (1997); Hertling (1938); Jones (1954); Adlerstein *et al* (2002); Albert (1994a); Greenstreet *et al* (1998); Temming *et al* (2004); Cramer and Daan (1986)
		Barents Sea	Zenkevich and Brotskaya (1931); Zatsepin (1939); Tseeb (1964); Antipova *et al.* (1990); Jiang and Jordensen (1996); Dolgov (2000)
Merlangius merlangus	Whiting	North Sea	Daan (1989); ICES (1988); (1997); Dahl and Kirkegaard (1986); Cranmer (1986); Jones (1954); Creutzberg and Duineveld (1986); Hertling (1938); Hislop *et al* (1991); Greenstreet *et al* (1998); Pedersen (1999, 2000); Temming *et al* (2004); Robb *et al* (1994); Temming and Mergardt (2002)
		Southern Bight	Bromley (1995)
		Baltic Sea	Hertling (1928)
		Kattegat	Pihl (1994)
		Celtic Sea	Pinnegar *et al* (2003)
Micromesistius poutassou	Blue Whiting	Mediterranean Sea	Macpherson (1978)
		Northern Spain shelf	Olaso and Rodríguez-Marín (1995); Velasco *et al.* (1996); Olaso (1998); Preciado *et al.* (2003)
		Atlantic French shelf	Sorbe (1980)
		Portuguese shelf	Cabral and Murta (1995)
		Norwegian Sea	Dumke (1983); Plekhanova (1990); Plekhanova and Soboleva (1981)
		Barents Sea	Zilanov (1984); Belikov *et al.* (2004)
Pollachius virens	Saithe	North Sea	Daan (1989); ICES (1997)
		Celtic Sea	Pinnegar *et al* (2003)
		Barents Sea	Mironova (1956); Lukmanov *et al.* (1975); A.Dolgov (2002a,b); S.Dolgov (2002)
Trisopterus esmarkii	Norway Pout	North Sea	Albert (1991, 1994b); Raitt and Adams (1965)
Trisopterus luscus	Bib	Irish Sea	Armstrong (1982)
		Northern Spain shelf	Olaso (1990); Olaso and Rodríguez-Marín (1995); Velasco *et al.* (1996); Serrano *et al.* (2003b); Preciado *et al.* (2003)
Trisopterus minutus	Poor Cod	Irish Sea	Armstrong (1982)
		Norwegian Deep	Albert (1993)
		Northern Spain shelf	Velasco *et al.* (1996); Serrano *et al.* (2003b)
Enchelyopus cimbrius	4-Bearded Rockling	North Sea	Creutzberg and Duineveld (1986)
		Norwegian Deep	Albert (1993); Mattson (1981)
Antonogadus macrophtalmus	Bigeye rocking	Northern Spain shelf	Olaso and Rodríguez-Marín (1995); Velasco *et al.* (1996); Serrano *et al.* (2003a, b); Preciado *et al.* (2003)
Phycis blennoides	Greater forkbeard	Northern Spain shelf	Velasco *et al.* (1996)

SPECIES	COMMON NAME	AREA	SOURCE
Merluccius merluccius	Hake	Celtic Sea	Pinnegar *et al* (2003); Du Buit (1996)
		Northern Spain shelf	Pereda *et al*. (1981); González *et al*. (1985); Olaso (1990); Olaso *et al*. (1994); Velasco and Olaso (1998a); Olaso and Pereda (1986); Velasco and Olaso (2000); Riis-Vestergaard *et al* (2000); Preciado *et al*. (2003); Olaso *et al*. (2004)
		Atlantic French shelf	Guichet (1995)
		Portuguese shelf	Cabral and Murta (1995)

PLEURONECTIFORMES

SPECIES	COMMON NAME	AREA	SOURCE
Lepidorhombus boscii	4-Spot Megrim	Mediterranean Sea	Morte *et al* (1999)
		Tyrrhenian Sea	Sartor and Ranieri (1996)
		Northern Spain shelf	Olaso and Rodríguez-Marín (1995); Velasco *et al*. (1996); Rodríguez-Marín and Olaso (1993); Serrano *et al*. (2003[a]); Preciado *et al*. (2003)
Lepidorhombus whiffiagonis	Megrim	North Sea	Rae (1956, 1963)
		Celtic Sea	Pinnegar *et al* (2003); du Buit (1992)
		Scottish Waters	Bennet (1963); du Buit (1984)
		Mediterranean Sea	Morte *et al* (1999)
		Northern Spain shelf	Olaso and Rodríguez-Marín (1995); Velasco *et al*. (1996); Rodríguez-Marín and Olaso (2003); Serrano *et al*. (2003[a])
		Icelandic Waters	Steinarsson (1979)
Psetta maxima	Turbot	Dutch Wadden Sea	Braber and De Groot (1973)
		Baltic Sea	Hertling (1928)
Scophthalmus rhombus	Brill	Dutch Wadden Sea	Braber and De Groot (1973)
Arnoglossus laterna	Scaldfish	Irish Sea	Amezcua *et al* (2003)
Glyptocephalus cynoglossus	Witch	Irish Sea	Amezcua *et al* (2003)
		Norwegian Deep	Mattson (1981)
		Icelandic Waters	Steinarsson (1979)
Hippoglossoides platessoides	Long Rough Dab	North Sea	Ntiba and Harding (1993)
		Kattegat	Pihl (1994)
		Irish Sea	Amezcua *et al* (2003)
		Barents Sea	Komarova (1939); Simacheva and Glukhov (1990); Berestovsky (1995, 1996); Dolgova and Dolgov (1997); Dolgov (2000)
Limanda limanda	Common Dab	Dutch Wadden Sea	Braber and De Groot (1973)
		German Bight	Knust (1986, 1987)
		North Sea	Creutzberg and Duineveld (1986); DeClerck and Torreele (1988); Temming and Hammer (1994)
		Kattegat	Pihl (1994)
		Baltic Sea	Hertling (1928)
		Irish Sea	Amezcua *et al* (2003)

SPECIES	COMMON NAME	AREA	SOURCE
Microstomus kitt	Lemon sole	North Sea	Rae (1956, 1963)
		Scottish waters	Bennet (1956)
		Irish Sea	Amezcua *et al* (2003)
		Icelandic Waters	Steinarsson (1979)
Platichthys flesus	Flounder	Dutch Wadden Sea	De Vlas (1979)
		North Sea	Summers (1980); Maes *et al.* (2003)
Pleuronectes platessa	Plaice	North Sea	DeClerck and Buseyne (1989); Jobling (1982)
		Irish Sea	Basimi and Grove (1985); Amezcua *et al* (2003)
		Southern Bight	Braber and De Groot (1973)
		Baltic Sea	Hertling (1928)
		Kattegat	Pihl (1994)
		Dutch Wadden Sea	De Vlas (1979)
Buglossidium luteum	Solenette	Irish Sea	Amezcua *et al* (2003); Nottage and Perkins (1983)
Microchirus variegatus	Thickback Sole	Irish Sea	Amezcua *et al* (2003)
Solea solea	Dover Sole	Dutch Wadden Sea	Braber and De Groot (1973)
		Irish Sea	Amezcua *et al* (2003)
SCORPAENIFORMES			
Eutrigla gurnardus	Grey gurnard	North Sea	ICES (1997); Creutzberg and Duineveld (1986); Agger and Ursin (1976); Kaiser and Spencer (1993)
		Northern Spain shelf	Olaso and Rodríguez-Marín (1995); Velasco *et al.* (1996); Serrano *et al.* (2003a, b); Preciado *et al.* (2003)
Aspitrigla cuculus	Red gurnard	Northern Spain shelf	Olaso and Rodríguez-Marín (1995); Velasco *et al.* (1996); Serrano *et al.* (2003a, b)
Trigla lucerna	Tub gurnard	Northern Spain shelf	Olaso and Rodríguez-Marín (1995); Velasco *et al.* (1996); Serrano *et al.* (2003a, b); Preciado *et al.* (2003)
Chelidonichthys obscurus	Longfin gurnard	Northern Spain shelf	Olaso and Rodríguez-Marín (1995); Velasco *et al.* (1996); Serrano *et al.* (2003a, b)
Helicolenus dactylopterus	Blackbelly rosefish	Northern Spain shelf	Serrano *et al.* (2003a, b); Preciado *et al.* (2003)
PERCIFORMES			
Echiichthys vipera	Lesser Weever	North Sea	Creutzberg and Duineveld (1986)
Trachinus draco	Greater weever	Northern Spain shelf	Olaso *et al.* (2002)
Callionymus lyra	Common Dragonet	North Sea	Creutzberg and Duineveld (1986)
		Galway Bay	King *et al* (1994)
		Northern Spain shelf	Olaso and Rodríguez-Marín (1995); Velasco *et al.* (1996)
Callionymus maculata	Spotted Dragonet	Scottish Waters	Gibson and Ezzi (1979)
Anarhichas lupus	Wolf fish	North Sea	Liao and Lucas (2000)
		Western Greenland	Munk (2002)
		Barents Sea	Zenkevich and Brotskaya (1931); Orlova *et al.* (1989)

SPECIES	COMMON NAME	AREA	SOURCE
Ammodytes marinus	Rait's Sandeel	North Sea	Macer (1966)
		Clyde	Rankine and Morrison (1989)
Hyperoplus lanceolatus	Greater Sandeel	Clyde	Rankine and Morrison (1989)
Scomber scombrus	Atlantic mackerel	North Sea	Daan (1989); ICES (1997); Mehl and Westgaard (1983); Walsh and Rankine (1979)
		English Channel	Bullen (1912)
		Northern Spain shelf	Olaso and Rodríguez-Marín (1995); Olaso *et al*. (2004)
Trachurus trachurus	Horse mackerel	North Sea	ICES (1997)
		Northern Spain shelf	Olaso *et al*. (1999)
Lycodes vahlii	Vahl's Eelpout	Norwegian Deep	Albert (1993)
Dicentrarchus labrax	Bass	North Sea (estuary)	Maes *et al*. (2003)
Mullus surmuletus	Striped red mullet	Northern Spain shelf	Olaso and Rodríguez-Marín (1995); Velasco *et al*. (1996); Serrano *et al*. (2003a, b)
Pagellus acarne	Axillary seabream	Northern Spain shelf	Olaso and Rodríguez-Marín (1995); Velasco *et al*. (1996); Serrano *et al*. (2003a, b)

MISCELLANEOUS SPECIES

SPECIES	COMMON NAME	AREA	SOURCE
Anguilla anguilla	Common eel	North Sea (estuary)	Maes *et al*. (2003)
Conger conger	European conger	Northern Spain shelf	Olaso and Rodríguez-Marín (1995); Velasco *et al*. (1996); Serrano *et al*. (2003a, b); Preciado *et al*. (2003)
Lophius americanus	American Anglerfish	western North Atlantic	Armstrong *et al* (1996)
Lophius piscatorius	Anglerfish	Irish Sea	Crozier (1985)
		Northern Spain shelf	Pereda and Olaso (1984); Pereda and Olaso (1990); Preciado *et al.* (2003)
		Portuguese shelf	Azevedo (1996)
Lophius budegassa	Black angler fish	Northern Spain shelf	Pereda and Olaso (1984)
		Portuguese shelf	Azevedo (1996)
Zeus faber	John Dory	Mediterranean Sea	Stergiou and Fourtourni (1991)
		Portuguese shelf	Silva (1999)
		Northern Spain shelf	Velasco and Olaso (1998b); Olaso (1990(
Clupea harengus	Herring	North Sea	Last (1982, 1987); Daan *et al* (1985); Bullen (1912); Savage (1937); Hardey (1924); Maes *et al*. (2003)
		Norwegian Sea	Dalpadado *et al*. (2000); Gislason and Asstthorson (2002)
		Barents Sea	Boldovsky (1941); Manteifel (1941); Rudakova (1966)
Sprattus sprattus	Sprat	North Sea	Last (1982); Maes *et al*. (2003)
Thunnus thynnus	Bluefin tuna	Northern Spain shelf	Ortiz de Zarate and Cort (1986)
Xiphias glaudius	Swordfish	Central Eastern Atlantic	Velasco and Quintan (2000)

11   Broadscale temporal changes in the fish assemblages along the European continental shelf of the eastern Atlantic

11.1   Introduction

In previous reports (ICES, 2003b, 2004b), similar temporal patterns in some size-based metrics were observed in both the Celtic Sea and Portuguese waters, which may indicate that similar processes are operating over the continental shelf of north-western Europe. Hence, studies examining the temporal and spatial patterns in fish communities and associated community and size-based metrics over this broad geographic scale may facilitate our understanding of broad scale processes that may affect potential EcoQOs.

Several IBTS and other fishing surveys operate in these southern and western areas, though there is not yet complete standardisation of the IBTS surveys in this region (ICES, 2003a, 2004a). These surveys operate from North-west Scotland (VI a), Irish Sea (VII a), west of Ireland (VII b) and Porcupine Bank, western English Channel (VII e), Bristol Channel (VII f), Celtic Sea (VIIg-j), Bay of Biscay (VIII a,b), Cantabrian Sea and northern Galicia (VIII c), and off southern Galicia, Portugal and southern Spain (IX a).

There have been several studies of the fish and fish communities in many of these area, especially in the Irish and Celtic Seas (Rogers, 1994; Warnes and Jones, 1995; Rogers *et al.*, 1998, 1999a, 1999b; Rogers and Ellis 2000; Ellis *et al.*, 2000, 2002a; Trenkel *et al.*, 2004), Bay of Biscay (e.g., Blanchard, 2001; Blanchard *et al.*, 2002; Souissi *et al.*, 2002) and Iberian waters (Sánchez, 1993; Sánchez *et al.*, 2002; Sánchez and Serrano, 2003; Farina *et al.*, 1997; Gomes *et al.*, 2001). There are, however, fewer studies encompassing the overall area. The recent SESITS project has evaluated the surveys operating from 36–52ºN, incorporating surveys from Portugal, Spain and France (Sánchez, 1999).

11.2   Studies in the Bay of Biscay

The groundfish species assemblages on the continental shelf of the Bay of Biscay are spatially organized mainly according to depth, latitude, longitude and substrate type (Sánchez, 1993; Souissi *et al.*, 2001; Sánchez and Serrano, 2003; Poulard *et al.*, 2003). The relative stability of the demersal fish communities throughout the last decade contrasts with the strong variability of some mesoscale hydrodynamic features (e.g., upwellings, lower salinity water lenses and cold pools) encountered on the French continental shelf (Puillat *et al.*, 2003) and in the Cantabrian Sea (Gil *et al.*, 2002). The narrow continental shelf in the Cantabrian Sea produces strong environmental gradients over a short distance. Using Canonical Correspondence Analysis (CCA) on the 1990s decadal surveys database, depth showed as the most influential and stable factor determining the assemblages observed and five main groups were described: coastal, inner-shelf, middle-shelf, outer-shelf and shelf-edge (Sánchez, 1993; Sánchez and Serrano, 2003). The most discriminatory were the coastal group and the deepest strata groups (outer-shelf and shelf-edge), situated at the extremes of the environmental gradients analysed (Figure 11.1). On the other hand, in the intermediate parts of these gradients, the inner-shelf and middle-shelf groups displayed reduced dispersion, with a position closer to the centroid of the biplot. This centroid was occupied by ubiquitous species that inhabit a wide range of environmental conditions, and included many of the dominant demersal predators (e.g., hake *Merluccius merluccius*, anglerfish *Lophius* spp. and conger eel *Conger conger*) that live in the area. Although the spatial patterns of fish communities remains stable, decadal changes in the structure of these communities have been described and related to climate change and fishing effects (Blanchard *et al.*, 2002). Within these communities in particular, the relative abundances of species that are at their latitudinal limit of geographical distribution exhibit changes

concomitant with a mean temperature increase of 1.5°C: the Lusitanean and sub-tropical species show increasing trends in abundance, whereas boreal ones have reportedly decreased (Poulard *et al*., 2003). Fishing on these communities alters species interactions, leading to increasing variability of the total production of the groundfish communities (Blanchard, 2001).
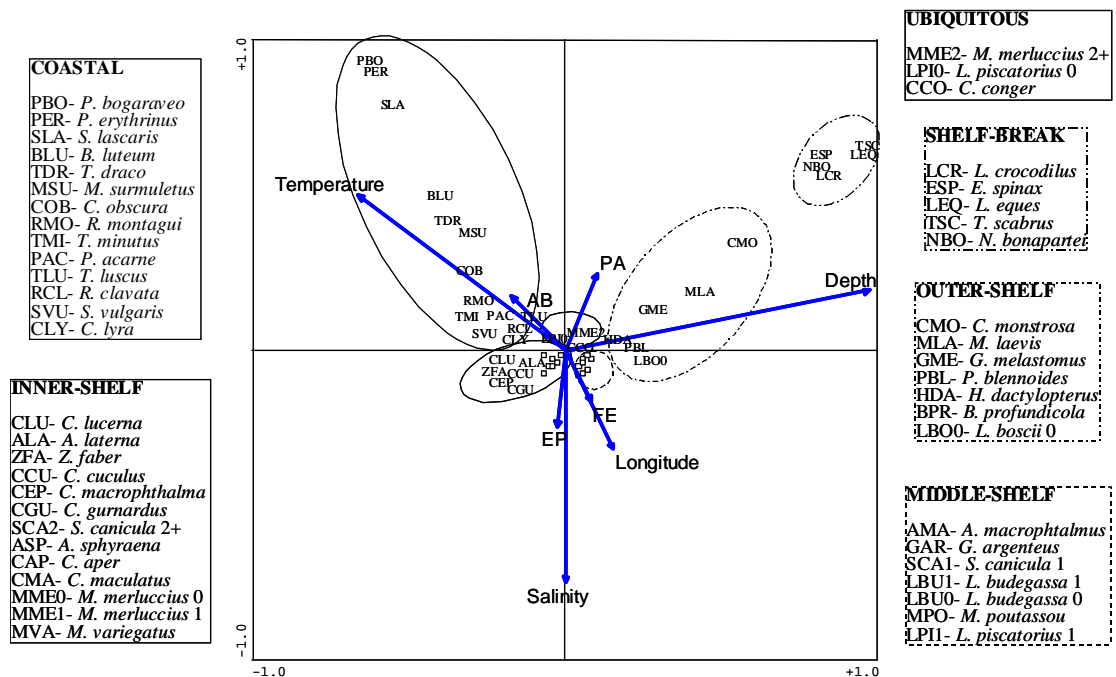
Figure 11.1: Canonical Correspondence Analysis (CCA) biplot of species *vs.* variables of the Cantabrian Sea from data of the 1993 bottom trawl survey. Environmental variables analysed are depth, bottom temperature, bottom salinity, longitude and geographical sectors (FE: Finisterre-Estaca; EP: Estaca-Peñas; PA: Peñas-Ajo and AB: Ajo-Bidasoa). Adapted from Sánchez and Serrano, 2003.

Fisheries management in this ICES area considers the boundary between the French and Spanish continental shelves (Cap Breton canyon) as a delimiter of the northern and southern stocks of some species of commercial interest (e.g., hake, megrims *Lepidorhombus* spp. and anglerfish), some of which are considered over-fished (ICES, 2005). Information on the fish communities inhabiting both sides of the Bay of Biscay has been historically obtained independently by French and Spanish bottom trawl surveys. Significant differences in the methodology and sampling gears in these surveys (ICES, 1999) hampered analyses to characterize the fish assemblages of both sides of the Bay of Biscay shelf.

The SESITS project (Southwestern European Shelf International Trawl Surveys), in which teams from France, Spain and Portugal were involved, and which was co-financed by the European Commission, focused on the standardisation of the methodology used in the surveys by overlapping working areas to obtain conversion factors (ICES 1999; Sánchez, 1999). In 1997 and 1998, the Spanish survey was extended to the French shelf at the level of the Gironde estuary. Information obtained using the same methodology throughout a wide area of the Bay of Biscay has thus been made available to study possible differences on fish populations. The study has been carried out to elucidate the importance of the Cap Breton canyon as a biogeographical boundary for fish communities and commercial target species. The study included several phases: hydrographic characteristics of the internal Bay of Biscay area, abundance indices of species of commercial interest, species assemblages and aggregation patterns of hake recruits.

The hydrographic structures around the shelf-slope in the Cantabrian Sea are the result of the regime of seasonal winds and dimensions of the continental shelf. The persistence of winds with an Eastern component in summer gives rise to dragging surface waters (Eckman effect) from the coast out to the open sea causing upwellings. Owing to the narrowness of the shelf, vertical movements of sub-surface waters are confined to areas with mesoscale dimensions of the order of 40 km. Over the French shelf the regime of winds with a northern component which would originate upwelling phenomena are not persistent enough, since at these latitudes of the northern hemisphere the winds are more characteristically of an E - W component. The great width of the French shelf leads to tidal currents being predominant in the mixing phenomenon of deep and surface waters. By its nature, this process gives rise to much greater structures (approx. 100 km) than those appearing in the Cantabrian Sea shelf area.

During the 1997 survey, an anticyclonic circulation covering the majority of the southern French continental shelf and slope resulted in a southward current over the internal shelf and northward along the slope. On the eastern Cantabrian shelf, the southward current changed its direction towards the west, but flowed along the outer side of the shelf. Offshore from Santander there was a cyclonic eddy. Studies of salinity and temperature shows the existence of some fresh water due to the continental flow (mainly from the Gironde estuary) on the French shelf. Over the Cantabrian Sea shelf area, cold water originates from the summer upwellings of sub-surface waters. The southward fluxes from the French internal shelf do not invade the Spanish shelf; instead they circulate towards the west bordering the cold and saline waters. Only a small part of this flow seems to be confined to the beginning of the shelf in the Cantabrian Sea, at around 2°W longitude.

In summary, the differences in oceanographic conditions between the two areas are not due to the Cap Breton canyon, but rather to the different width and orientation of the shelves with respect to the regime of winds and the influence of continental flows.

The biomass and abundance indices from 1997 survey (Table 11.1) do not show remarkable differences between the two areas. There was a high dominance in both areas, with 90% of biomass accounted for by 12 species and more than 50% by two species (horse mackerel *Trachurus trachurus* and blue whiting *Micromesistius poutassou*). Total fish biomass was similar in both areas (182 kg/haul). In the Cantabrian Sea total abundance was higher (No./haul) for species as a whole due to the influence of silvery pout *Gadiculus argenteus*. Species richness was the same in both shelf areas (Table 11.1). Confidence intervals to 95% for the main commercial species for the log abundance are shown in Figure 11.2, where it can be observed the high degree of overlap between the two areas for the species considered.

Table 11.1. Biomass index (kg/haul), abundance index (nº/haul) and percentage over total biomass of the 25 main species of fish on both sides of Bay of Biscay (1997 survey data).

NORTH CAP BRETON (FRENCH SHELF)				SOUTH CAP BRETON (SPANISH SHELF)			
SPECIE	KG/HAUL	Nº/HAUL	%KG	SPECIE	KG/HAUL	Nº/HAUL	%KG
Trachurus trachurus	91.27	1942.0	49.9	*Micromesistius poutassou*	81.21	1706.7	44.5
Micromesistius poutassou	30.89	568.4	16.9	*Trachurus trachurus*	14.03	142.6	7.7
Trisopterus luscus	11.51	92.8	6.3	*Squalus acanthias*	9.98	2.3	5.5
Galeus melastomus	6.84	82.6	3.7	*Gadiculus argenteus*	9.44	1842.8	5.2
Merluccius merluccius	4.30	125.1	2.3	*Merluccius merluccius*	8.71	384.6	4.8
Trisopterus minutus	3.21	115.0	1.8	*Scyliorhinus canicula*	8.40	34.4	4.6
Argentina sphyraena	3.07	157.4	1.7	*Pagellus acarne*	6.88	29.9	3.8
Scyliorhinus canicula	2.79	10.6	1.5	*Lophius piscatorius*	5.56	1.4	3.0
Lepidorhombus whiffigonis	2.65	25.7	1.4	*Argentina sphyraena*	4.73	213.5	2.6
Lophius piscatorius	2.37	0.8	1.3	*Trisopterus luscus*	4.53	63.9	2.5
Callionymus lyra	2.16	36.4	1.2	*Chelidonichthys gurnardus*	4.17	144.3	2.3
Chelidonichthys gurnardus	1.68	28.9	0.9	*Galeus melastomus*	3.33	34.6	1.8
Scomber scombrus	1.44	10.5	0.8	*Lepidorhombus whiffiagonis*	3.20	38.0	1.8
Lophius budegassa	1.43	1.4	0.8	*Raja clavata*	2.16	1.6	1.2
Trachinus draco	1.37	18.5	0.8	*Lophius budegassa*	1.48	1.7	0.8
Aspitrigla cuculus	1.28	15.0	0.7	*Callionymus lyra*	1.34	25.5	0.7
Pagellus acarne	1.23	3.8	0.7	*Trisopterus minutus*	1.31	33.3	0.7
Lepidorhombus boscii	1.21	21.2	0.7	*Phycis blennoides*	1.12	5.9	0.6
Dicologoglossa cuneata	1.03	60.5	0.6	*Trachinus draco*	1.11	15.1	0.6
Raja clavata	0.79	0.3	0.4	*Conger conger*	1.09	2.3	0.6
Solea solea	0.73	10.4	0.4	*Arnoglossus laterna*	1.03	115.8	0.6
Capros aper	0.72	69.9	0.4	*Cepola rubescens*	1.01	23.0	0.6
Molva molva	0.67	0.1	0.4	*Aspitrigla cuculus*	0.97	12.8	0.5
Helicolenus dactylopterus	0.66	2.1	0.4	*Lepidorhombus boscii*	0.83	28.8	0.5
Cepola rubescens	0.66	17.1	0.4	*Raja montagui*	0.66	0.9	0.4
Total (25)	175.94	3416.5	**96.3**	Total (25)	178.25	4905.0	**97.7**
Others fish (63)	6.83	388.0	3.7	Other fish (63)	4.11	127.3	2.3
TOTAL	**182.77**	**3804.5**	100	**TOTAL**	**182.36**	**5032.2**	100

Table 11.2: Species characteristics of fish assemblages in the Bay of Biscay (Cap Breton area).

G1: Coastal coarse sand	G3: Middle shelf (70-120 m)	G4: Outer shelf (100-160m)	G5: Shelf edge (170-350 m)
Average similarity: 69.02	**Average similarity: 66.18**	**Average similarity: 67.71**	**Average similarity: 69.34**
Trachurus trachurus	*Argentina sphyraena*	*Merluccius merluccius*	*Micromesistius poutassou*
Trachinus draco	*Trachurus trachurus*	*Arnoglossus laterna*	*Gadiculus argenteus*
Pagellus acarne	*Trisopterus minutus*	*Argentina sphyraena*	*Lepidorhombus boscii*
Merluccius merluccius	*Arnoglossus laterna*	*Trachurus trachurus*	*Argentina sphyraena*
Boops boops	*Microchirus variegatus*	*Capros aper*	*Lepidorhombus whiffiagonis*
Echiichthys vipera	*Callionymus lyra*	*Microchirus variegatus*	*Trachurus trachurus*
Arnoglossus laterna	*Merluccius merluccius*	*Eutrigla gurnardus*	*Scyliorhinus canicula*
Trisopterus luscus	*Trisopterus luscus*	*Callionymus maculatus*	*Merluccius merluccius*
Mullus surmuletus	*Aspitrigla cuculus*	*Scyliorhinus canicula*	*Galeus melastomus*
Microchirus variegatus	*Capros aper*	*Micromesistius poutassou*	*Microchirus variegatus*
Buglossidium luteum	*Micromesistius poutassou*	*Lepidorhombus boscii*	*Phycis blennoides*
Solea vulgaris	*Eutrigla gurnardus*	*Cepola rubescens*	*Conger conger*
Aspitrigla obscura	*Scyliorhinus canicula*	*Lophius budegassa*	*Lophius budegassa*
Callionymus lyra	*Callionymus maculatus*	*Lepidorhombus whiffiagonis*	*Callionymus maculatus*
Dicologoglossa cuneata	*Zeus faber*	*Gadiculus argenteus*	*Lophius piscatorius*
	Cepola rubescens	*Lophius piscatorius*	*Capros aper*
G2: Estuarine fine sand	*Raja clavata*	*Zeus faber*	*Arnoglossus laterna*
Average similarity: 80.59	*Lophius piscatorius*	*Antonogadus macrophthalmus*	*Eutrigla gurnardus*
Pomatoschistus pictus	*Pomatoschistus pictus*	*Lesuerigobius friesii*	*Trigla lyra*
Dicologoglossa cuneata	*Lophius budegassa*	*Callionymus lyra*	*Antonogadus macrophthalmus*
Trisopterus luscus	*Conger conger*	*Aspitrigla cuculus*	*Pagellus acarne*
Solea vulgaris	*Lepidorhombus whiffiagonis*	*Raja clavata*	*Cepola rubescens*
Buglossidium luteum	*Antonogadus macrophthalmus*	*Solea vulgaris*	*Aspitrigla cuculus*
Merlangus merlangus	*Trachinus draco*	*Conger conger*	
Arnoglossus laterna	*Aspitrigla obscura*	*Boops boops*	
Spondyliosoma cantharus		*Raja montagui*	
Sprattus sprattus		*Trisopterus luscus*	
Callionymus lyra		*Trachinus draco*	
Argyrosomus regius		*Trigla lyra*	
Mullus surmuletus		*Deltentosteus quadrimaculatus*	
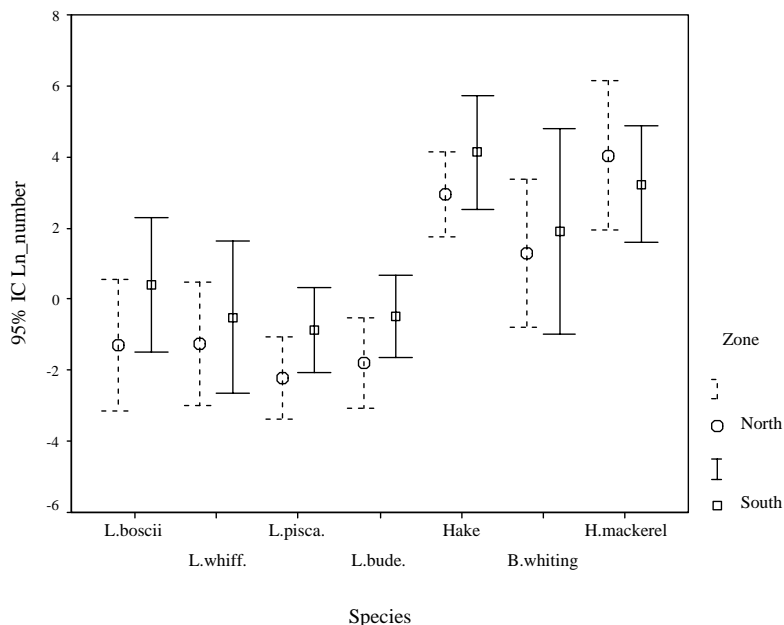
Figure 11.2: The 95% confidence limits of mean abundance (ln of number) by main commercial species caught to the North and South of Cap Breton canyon.

The hierarchical cluster analysis shows a strong bathymetric component in the area (Figure 11.3). In general terms, the main differences (similarity less than 60%) appear in five main sample groups. Groups 4 (outer shelf) and 5 (shelf edge) are independent of the influence of Cap Breton canyon since they appear throughout the area of study (Figure 11.4). On the other hand, group 2 was only found on the French shelf, off the Gironde estuary. The most characteristic species in this group (Table 11.2) included wedge sole *Dicologoglosa cuneata* and sprat *Sprattus sprattus* in shallow waters, and whiting *Merlangius merlangus* and ling *Molva molva* further offshore. These four species were not present in the Cantabrian Sea.

Canonical Correspondence Analysis (CCA) was used to assess the effect of environmental variables on fish assemblages. The abiotic variables used in the analysis were depth, bottom temperature and salinity, temperature and salinity at 50 m, and proximity to Cap Breton and the Gironde. The set of these variables explained 41.9% of the variance of species data and 70.1% of the variance of species-environment relationship. The most discriminatory environmental variables were depth, and bottom salinity and temperature (Table 11.3). The Cap Breton canyon was less discriminatory than the influence of the Gironde.
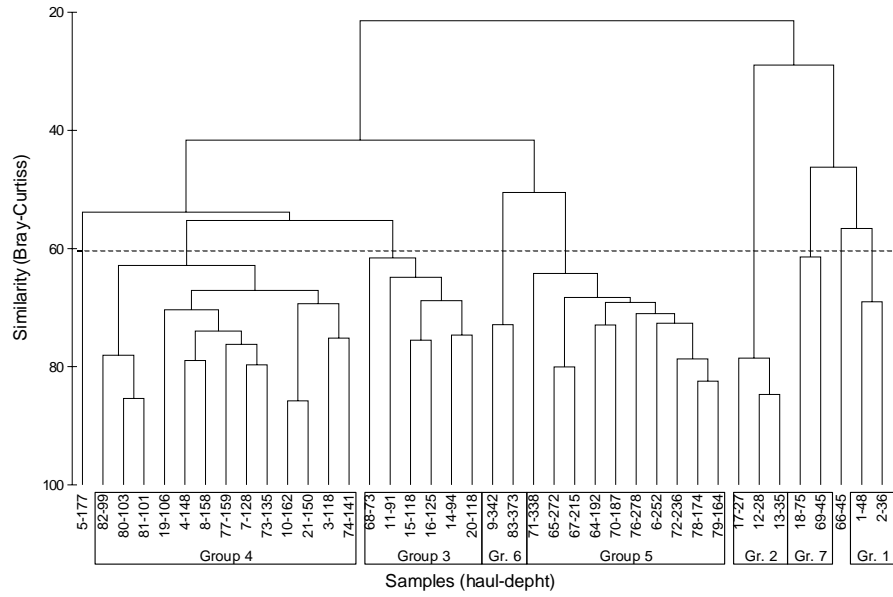
Figure 11.3: Bray-Curtis hierarchical cluster analysis of fish communities.
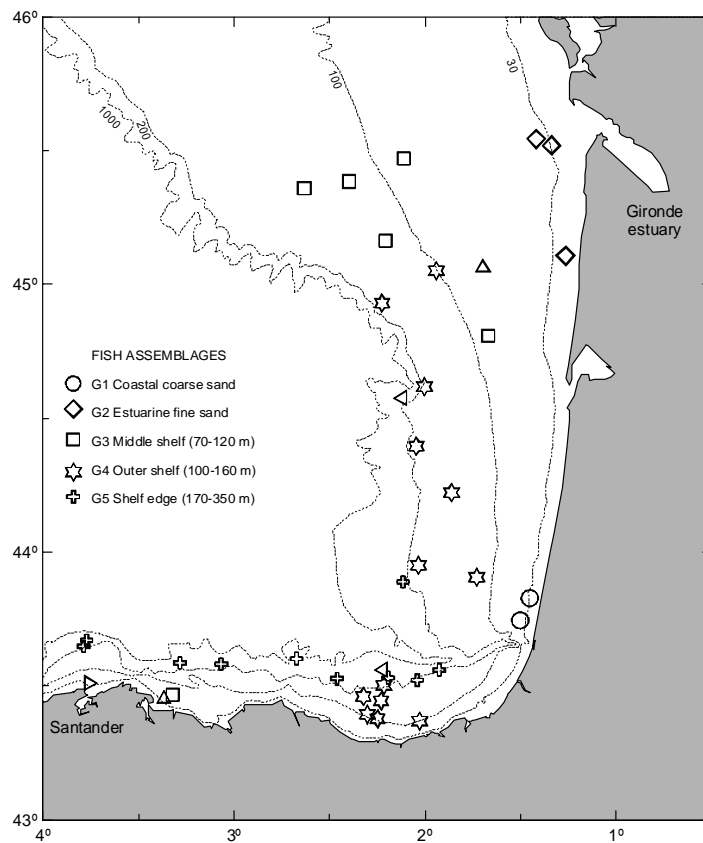
Figure 11.4: Distribution of fish assemblages from Bray-Curtis hierarchical cluster analysis of 1997 survey data in the Bay of Biscay (Cap Breton area).

Table 11.3: Interset correlations between environmental variables and the first and second canonical axis. Eigenvalues and percentage of variance of species data and of species-environmental relation explained by the CCA plane.

ENVIRONMENTAL VARIABLE	AXIS 1	AXIS 2
Depth	-0.7491	0.5779
Bottom salinity	-0.7426	-0.5373
Bottom temperature	0.9115	-0.1860
Salinity at 50 m	-0.8312	-0.1902
Latitude	0.4468	0.0846
Temperature at 50 m	0.3906	0.1337
Gironde effect	0.4769	-0.0181
Longitude	0.3242	0.0913
Cap Breton canyon	0.3212	-0.0339
Eigenvalues	0.558	0.314
% Var species data	26.8	41.9
% Var spp.-environmental relation	44.8	70.1

Depth and the particular hydrographic features of the Bay of Biscay are more relevant to the distribution patterns and abundance of species than the Cap Breton canyon itself. In general, the same species composition and population structures were found in both areas. As an exception, some differences have been found on the shelf off the Gironde estuary. The Gironde estuary seems to be the southern limit of distribution for cold water species, such as herring *Clupea harengus*, saithe *Pollachius virens*, Norway pout *Trisopterus esmarki*, dab *Limanda limanda*, sprat and whiting.

In muddy bottoms between 100 and 200 m, an important concentration of hake recruits appeared in the area of study, with two aggregations of more than 1000 individuals/hour situated on either side of the Cap Breton canyon (Figure 11.5). The highest recruit densities over the French shelf (Vasière des Landes) are located approximately in the anticyclonic core, and to a much lesser extent where there is a transport of waters towards the south. The denser concentration of hake recruits appearing in the Cantabrian Sea occurs around the area of influence of fluxes from the French shelf. Both nurseries seem to respond to the same recruitment process and their origins are likely to be from spawners situated in the French shelf (Northern Stock). Recruits from the eastern Cantabrian Sea only appear in those years in which the poleward current has not been very intense. In this case, low salinity waters with their origin in the French shelf appear. It should be noted that the recruitment index of the Southern Stock used in the calibration of VPA is obtained including the data from this nursery.
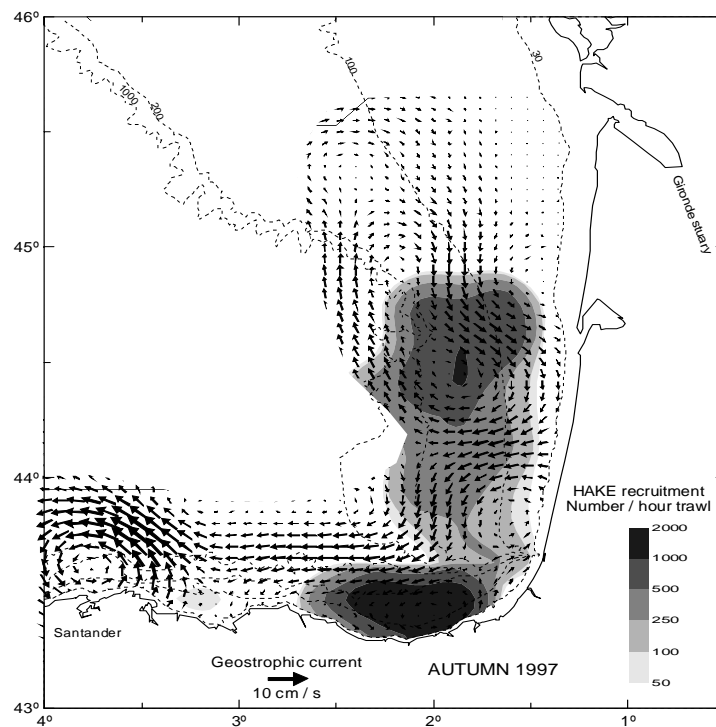
Figure 11.5: Geostrophic current (cm/s) and hake recruitment (number/hour) in the Cap Breton area during the 1997 survey (October).

11.3 Celtic Sea and Irish Sea

Beam trawl surveys are useful for identifying demersal assemblages, as they are more efficient for the sampling of many benthic and small demersal fishes, and epibenthic invertebrates. The fish and epibenthic invertebrate assemblages of the Irish Sea and western English Channel have been described in several reports (Ellis *et al*., 2000, 2002a; Ellis and Rogers, 2004), primarily using data from 4m beam trawl surveys and 2m beam trawl studies in the Celtic Sea (Ellis *et al*., 2002b). Flatfish, including dab *Limanda limanda*, plaice *Pleuronectes platessa* and sole *Solea solea* dominate catches in inshore stations in the eastern and western Irish Sea, and Cardigan and Carmarthen Bays, with dab and solenette *Buglossidium luteum* especially characteristic. Further offshore, on coarse grounds, thickback sole *Microchirus variegatus* and red gurnard *Aspitrigla cuculus* become increasingly dominant. The muddy *Nephrops* grounds in the western Irish Sea and off Cumbria are typified by flatfish such as witch *Glyptocephalus cynoglossus* and long-rough dab *Hippoglossoides platessoides*.

Cluster analyses (using presence-absence) of fish catches in the Celtic Sea, from the CEFAS groundfish survey (see Warnes and Jones 1995; Pinnegar *et al*., 2002 and ICES, 2003b, 2004b for further survey information) indicate that there are distinct fish assemblages in the western English Channel (60–90m deep), Celtic Sea (65–155m), waters off Brittany (100–200m), along the edge of the continental shelf (130–285m) and in deep water (>330m) (Figure 11.6). The main fish species occurring in these communities are listed in Table 11.4, and the spatial patterns in these fish assemblages are in close agreement with the observed patterns in epibenthic fauna (Ellis *et al*., 2002b).

Table 11.4: Species characteristic of fish assemblages in the Celtic Sea.

CELTIC SEA AVERAGE SIMILARITY: 75.09	SHELF EDGE AVERAGE SIMILARITY: 78.71	WESTERN CHANNEL AVERAGE SIMILARITY: 68.97
Trachurus trachurus	*Trachurus trachurus*	*Trachurus trachurus*
Trisopterus minutus	*Trisopterus minutus*	*Trisopterus luscus*
Scomber scombrus	*Zeus faber*	*Trisopterus minutus*
Scyliorhinus canicula	*Scomber scombrus*	*Sardina pilchardus*
Lophius piscatorius	*Scyliorhinus canicula*	*Scomber scombrus*
Melanogrammus aeglefinus	*Lophius budegassa*	*Scyliorhinus canicula*
Merlangius merlangus	*Lophius piscatorius*	*Sprattus sprattus*
Merluccius merluccius	*Melanogrammus aeglefinus*	*Merlangius merlangus*
Microstomus kitt	*Merluccius merluccius*	*Microchirus variegatus*
Molva molva	*Microchirus variegatus*	*Mullus surmuletus*
Eutrigla gurnardus	*Micromesistius poutassou*	*Engraulis encrasicolus*
Gadus morhua	*Lepidorhombus boscii*	*Callionymus lyra*
Argentina spp.	*Lepidorhombus whiffiagonis*	*Callionymus* spp.
Aspitrigla cuculus	*Callionymus* spp.	*Aspitrigla cuculus*
Zeus faber	*Capros aper*	*Echiichthys vipera*
Lepidorhombus whiffiagonis	*Argentina spp.*	*Molva molva*
Trisopterus esmarki	*Leucoraja naevus*	*Capros aper*
Callionymus spp.	*Helicolenus dactylopterus*	*Spondyliosoma cantharus*
Micromesistius poutassou	*Eutrigla gurnardus*	*Microstomus kitt*
Pleuronectes platessa	*Callionymus maculatus*	*Gadus morhua*
Microchirus variegatus	*Aspitrigla cuculus*	*Zeus faber*
Leucoraja naevus	*Merlangius merlangus*	*Merluccius merluccius*
Maurolicus mulleri	*Arnoglossus laterna*	*Melanogrammus aeglefinus*
Capros aper	*Pollachius virens*	*Trisopterus esmarki*
Limanda limanda	*Arnoglossus imperialis*	Ammodytidae (indet.)
Squalus acanthias	*Microstomus kitt*	*Scophthalmus rhombus*
Trigla lucerna	*Gadiculus argenteus*	*Pollachius pollachius*
Callionymus lyra	*Hippoglossoides platessoides*	*Raja montagui*
Sprattus sprattus	*Gadus morhua*	
Gobiidae (indet.)	*Callionymus lyra*	
Hippoglossoides platessoides	*Phycis blennioides*	
Glyptocephalus cynoglossus		

Deep-water Average similarity: 63.90	Brittany Average similarity: 72.37
Trachurus trachurus	*Trachurus trachurus*
Scomber scombrus	*Trisopterus minutus*
Phycis blennioides	*Zeus faber*
Malacocephalus laevis	*Sardina pilchardus*
Merluccius merluccius	*Scomber scombrus*
Micromesistius poutassou	*Scyliorhinus canicula*
Helicolenus dactylopterus	*Merluccius merluccius*
Gadiculus argenteus	*Micromesistius poutassou*
Argentina spp.	*Molva molva*
Capros aper	*Lepidorhombus whiffiagonis*
Lepidorhombus whiffiagonis	*Leucoraja naevus*
Lophius piscatorius	*Capros aper*
Microchirus variegatus	*Argentina* spp.
Etmopterus spinax	*Arnoglossus imperialis*
Molva dypterygia	*Aspitrigla cuculus*
Lepidorhombus boscii	*Microstomus kitt*
Myctophidae (indet.)	*Galeorhinus galeus*
Synaphobranchus kaupi	*Lophius piscatorius*
Conger conger	*Mullus surmuletus*
	Conger conger
	Mustelus asterias
	Engraulis encrasicolus
	Callionymus spp.
	Squalus acanthias
	Pollachius pollachius
	Microchirus variegatus
	Lophius budegassa

Note: In the first three-column table, the Celtic Sea column continues below row 32 with:
Conger conger, *Sardina pilchardus*, *Engraulis encrasicolus*, *Clupea harengus*, *Leucoraja fullonica*, *Raja clavata*, *Lophius budegassa*, *Callionymus maculatus*, *Pollachius pollachius*, *Arnoglossus imperialis*
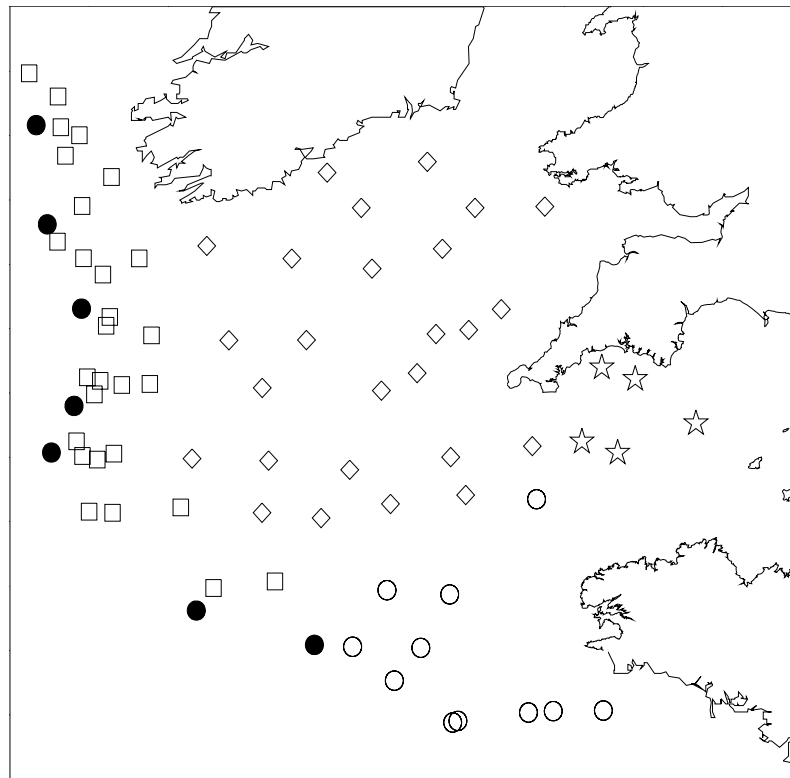
Figure 11.6: Distribution of fish assemblages in the Celtic Sea (closed circles = deep water, open squares = shelf edge, open diamonds = Celtic Sea, stars = western English Channel and open circles = Brittany).

11.4 Future studies

The SESITS report (Sánchez, 1999) provided a detailed list of those species caught in IFREMER, IEO and IPIMAR surveys, and described the spatial and temporal patterns in the distribution and relative abundance of commercial species and the hydrography of the study area. Given that the Marine Institute and CEFAS have begun fourth quarter groundfish surveys in the Irish and Celtic Seas, and the IEO also begain Porcupine groundfish surveys in 2001, the combined surveys cover an extensive latitudinal range, from southern Spain to Scotland (36–60ºN, Figure 11.7). Hence, it may now be possible to expand the spatial analyses of fish communities along the Atlantic sea board of northwestern Europe. A preliminary list of fish species caught by FRS and CEFAS (surveys in 2004) and the Marine Institute (2003) are summarized in Table 11.5, with Table 11.6 listing those fish species caught in more southerly surveys. WGFE considered that these combined surveys will provide a unique data set and hope to conduct integrated studies in conjunction with IBTSWG in future years.
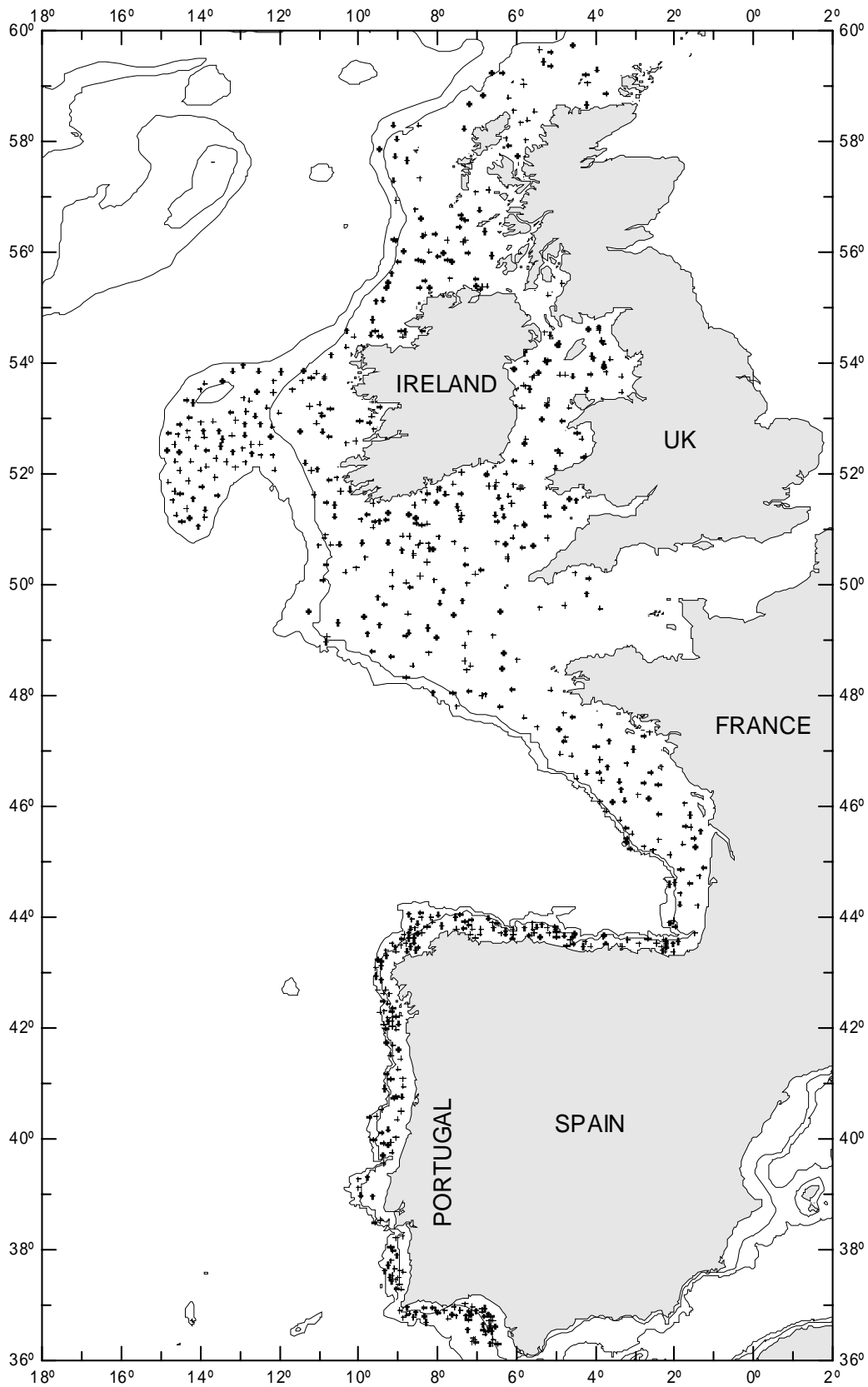
Figure 11.7: Distribution of surveys along the Atlantic seaboard of Europe.

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Table 11.5: List of fish species recorded in groundfish surveys around the British Isles during CE-FAS and FRS surveys (2004) and Marine Institute surveys (2003).

Species	CEFAS (2004)	FRS (2004)	MI (2003)
Acantholabrus palloni			✓
Agonus cataphractus	✓		✓
Ammodytes marinus			✓
Ammodytes spp			✓
Ammodytes tobianus	✓		
Ammodytidae	✓		✓
Anguilla anguilla	✓		✓
Argentina silus		✓	✓
Argentina sphyraena		✓	✓
Argentinidae	✓		✓
Arnoglossus imperialis	✓	✓	✓
Arnoglossus laterna	✓	✓	✓
Arnoglossus thori			✓
Aspitrigla cuculus	✓	✓	✓
Balistes capriscus	✓		
Belone belone	✓		
Blennius ocellaris		✓	✓
Blennius gattorugine			✓
Boops boops			✓
Buglossidium luteum	✓	✓	✓
Callionymus lyra	✓	✓	✓
Callionymus maculatus	✓	✓	✓
Callionymus reticulatus	✓		✓
Capros aper	✓	✓	✓
Cepola rubescens	✓		✓
Chimaera monstrosa		✓	
Ciliata mustela	✓		✓
Clupea harengus	✓	✓	✓
Conger conger	✓	✓	✓
Ctenolabrus rupestris	✓		✓
Cyclopterus lumpus	✓	✓	
Dasyatis pastinaca	✓		
Dicentrarchus labrax	✓		✓
Dipturus batis	✓	✓	✓
Dipturus oxyrinchus			✓
Echiodon drummondi		✓	✓
Engraulis encrasicolus	✓	✓	✓
Entelurus aequoreus		✓	✓
Eutrigla gurnardus	✓	✓	✓
Enchelyopus cimbrius	✓	✓	
Lesueurigobius friesii	✓	✓	
Gadiculus argenteus	✓	✓	✓
Gadus morhua	✓	✓	✓
Gaidropsarus spp			✓
Gaidropsarus vulgaris		✓	✓
Galeorhinus galeus	✓		✓
Galeus melastomus		✓	✓

Species	CEFAS (2004)	FRS (2004)	MI (2003)
Glyptocephalus cynoglossus	✓	✓	✓
Gobiidae	✓		
Gobius niger			✓
Gobius paganellus	✓		
Gobius spp			✓
Gymnammodytes semisquamatus	✓		
Helicolenus dactylopterus	✓	✓	✓
Hippoglossoides platessoides	✓	✓	✓
Caelorinchus caelorhinchus		✓	
Hyperoplus immaculatus	✓	✓	✓
Hyperoplus lanceolatus	✓	✓	✓
Labrus mixtus		✓	✓
Lepidorhombus boscii		✓	✓
Lepidorhombus whiffiagonis	✓	✓	✓
Leucoraja fullonica			✓
Leucoraja naevus	✓	✓	✓
Limanda limanda	✓	✓	✓
Liparis liparis	✓		✓
Liparis montagui	✓		✓
Lophius budegassa	✓	✓	✓
Lophius piscatorius	✓	✓	✓
Lumpetus lampretaeformis		✓	
Macrorhamphosus scolopax			✓
Macrouridae			✓
Maurolicus muelleri	✓	✓	✓
Melanogrammus aeglefinus	✓	✓	✓
Merlangius merlangus	✓	✓	✓
Merluccius merluccius	✓	✓	✓
Microchirus variegatus	✓	✓	✓
Micromesistius poutassou	✓	✓	✓
Microstomus kitt	✓	✓	✓
Molva molva	✓	✓	✓
Mullus barbatus			✓
Mullus surmuletus	✓	✓	✓
Mustelus asterias	✓		✓
Mustelus mustelus	✓		✓
Myctophidae			✓
Myoxocephalus scorpius	✓	✓	✓
Pagellus bogaraveo	✓		
Pegusa lascaris	✓		
Petromyzon marinus		✓	
Pholis gunnellus			✓
Phrynorhombus norvegius	✓	✓	✓
Phycis blennoides	✓	✓	✓
Platichthys flesus	✓		
Pleuronectes platessa	✓	✓	✓
Pollachius pollachius	✓	✓	✓
Pollachius virens		✓	✓
Pomatoschistus microps			✓

Species	CEFAS (2004)	FRS (2004)	MI (2003)
Raja brachyura	✓	✓	✓
Raja clavata	✓	✓	✓
Raja microocellata	✓		✓
Raja montagui	✓	✓	✓
Raja undulata			✓
Raniceps raninus			✓
Lampetra fluviatilis	✓		
Sardina (clupea) pilchardus	✓		✓
Scomber scombrus	✓	✓	✓
Scophthalmus maximus	✓	✓	✓
Scophthalmus rhombus	✓	✓	✓
Scorpaenidae			✓
Scyliorhinus canicula	✓	✓	✓
Scyliorhinus stellaris	✓		✓
Sebastes marinus		✓	
Solea solea	✓	✓	✓
Spondyliosoma cantharus	✓		
Sprattus sprattus	✓	✓	✓
Squalus acanthias	✓	✓	✓
Synganthus acus		✓	
Syngnthidae			✓
Taurulus bubalis			✓
Taurulus lilljeborgi	✓		
Gasterosteus aculeateus		✓	
Torpedo nobiliana			✓
Echiichthys vipera	✓	✓	✓
Trachinus draco	✓		✓
Trachurus trachurus	✓	✓	✓
Trigla lucerna	✓	✓	✓
Trigloporus lastoviza	✓	✓	✓
Trisopterus esmarki	✓	✓	✓
Trisopterus luscus	✓	✓	✓
Trisopterus minutus	✓	✓	✓
Chirolophis ascanii	✓		
Zeugopterus punctatus			✓
Zeus faber	✓	✓	✓

Table 11.6: List of fish species recorded in groundfish surveys in the Celtic Sea, Bay of Biscay and off the Atlantic coasts of the Iberian Peninsula (from Sánchez, 1999).

FAMILIES	*SCIENTIFIC NAMES*	*NODC CODES*	CELTIC SEA B. BISCAY	NORTH OF SPAIN	PORTUGAL	GULF OF CÁDIZ
FISHES						
AGONIDAE	*Agonus cataphractus*	*8831080803*	X			
ALEPOCEPHALIDAE	*Alepocephalus bairdii*	*8760010305*	X	X		
ALEPOCEPHALIDAE	*Alepocephalus rostratus*	*8760010302*	X	X		
ALEPOCEPHALIDAE	*Xenodermichthys copei*	*8760011201*	X	X	X	
AMMODYTIDAE	*Ammodytes marinus*	*8845010106*	X			
AMMODYTIDAE	*Ammodytes tobianus*	*8845010105*	X	X	X	
AMMODYTIDAE	*Gymnammodytes semisquamatus*	*8845010201*	X		X	
AMMODYTIDAE	*Hyperoplus immaculatus*	*8845010302*	X			
AMMODYTIDAE	*Hyperoplus lanceolatus*	*8845010301*	X	X	X	
ANGUILLIDAE	*Anguilla anguilla*	*8741010102*	X	X	X	
APOGONIDAE	*Epigonus denticulatus*	*8835180406*			X	X
APOGONIDAE	*Epigonus telescopus*	*8835180403*	X		X	
ARGENTINIDAE	*Argentina silus*	*8756010203*	X	X		
ARGENTINIDAE	*Argentina sphyraena*	*8756010209*	X	X	X	X
ATHERINIDAE	*Atherina presbyter*	*8805021003*	X	X	X	
BALISTIDAE	*Balistes carolinensis*	*8860020205*	X	X	X	
BATRACHOIDIDAE	*Halobatrachus didactylus*	*8783010403*				X
BELONIDAE	*Belone belone*	*8803020502*	X	X	X	
BERYCIDAE	*Beryx decadactylus*	*8850020101*	X	X	X	X
BERYCIDAE	*Beryx splendens*	*8810050101*	X	X	X	X
BLENNIIDAE	*Blennius ocellaris*	*8810050102*	X	X	X	X
BLENNIIDAE	*Parablennius gattorugine*	*8842010110*	X			
BOTHIDAE	*Arnoglossus imperialis*	*8857031703*	X	X	X	X
BOTHIDAE	*Arnoglossus laterna*	*8857031702*	X	X	X	X
BOTHIDAE	*Arnoglossus rueppelli*	*8857031705*			X	X
BOTHIDAE	*Arnoglossus thori*	*8857031706*	X	X	X	X
BRAMIDAE	*Brama brama*	*8835710102*			X	
CALLIONYMIDAE	*Callionymus lyra*	*8846010106*	X	X	X	
CALLIONYMIDAE	*Callionymus maculatus*	*8846010107*	X	X		X
CALLIONYMIDAE	*Callionymus reticulatus*	*8846010120*	X	X	X	X
CALLIONYMIDAE	*Synchiropus phaeton*	*8846010122*			X	X
CAPROIDAE	*Antigonia capros*	*8811060101*			X	
CAPROIDAE	*Capros aper*	*8811060301*	X	X	X	X
CARANGIDAE	*Decapterus rhonchus*	*8835280358*			X	X
CARANGIDAE	*Naucrates ductor*	*8835281501*			X	
CARANGIDAE	*Trachurus mediterraneus*	*8835280105*	X	X	X	X
CARANGIDAE	*Trachurus picturatus*	*8835280106*	X	X	X	X
CARANGIDAE	*Trachurus trachurus*	*8835280103*	X	X	X	X
CARAPIDAE	*Echiodon dentatus*	*8792020201*		X		
CARAPIDAE	*Echiodon drummondi*	*8792020202*	X			
CARCHARHINIDAE	*Prionace glauca*	*8708020601*		X		
CENTRACANTHIDAE	*Spicara flexuosa*	*8835530301*			X	X
CENTRACANTHIDAE	*Spicara maena*	*8835530302*				X
CENTROLOPHIDAE	*Schedophilus medusophagus*	*8851010302*		X		
CEPOLIDAE	*Cepola macrophthalma*	*8835700101*	X	X	X	X
CETORHINIDAE	*Cetorhinus maximus*	*8707120101*		X		
CHAULODONTIDAE	*Chauliodus sloani*	*8759060103*		X	X	X
CHAUNACIDAE	*Chaunax pictus*	*8787030101*		X	X	

FAMILIES	*SCIENTIFIC NAMES*	*NODC CODES*	CELTIC SEA B. BISCAY	NORTH OF SPAIN	PORTUGAL	GULF OF CÁDIZ
CHIMAERIDAE	*Chimaera monstrosa*	*8716020202*	X	X	X	X
CHLOROPHTHALMIDAE	*Chlorophthalmus agassizi*	*8762040101*		X	X	
CITHARIDAE	*Citharus linguatula*	*8857010201*		X	X	X
CLUPEIDAE	*Alosa alosa*	*8747010107*	X	X	X	X
CLUPEIDAE	*Alosa fallax*	*8747010109*	X	X	X	
CLUPEIDAE	*Clupea harengus*	*8747010201*	X			
CLUPEIDAE	*Sardina pilchardus*	*8747012201*	X	X	X	X
CLUPEIDAE	*Sardinella aurita*	*8747011001*			X	X
CLUPEIDAE	*Sprattus sprattus*	*8747011701*	X		X	X
CONGRIDAE	*Bathyuroconger vicinus*	*8741122001*			X	
CONGRIDAE	*Conger conger*	*8741120111*	X	X	X	X
CONGRIDAE	*Gnathophis mystax*	*8741120405*			X	
COTTIDAE	*Myoxocephalus scorpius*	*8831022207*	X			X
COTTIDAE	*Taurulus bubalis*	*8831024601*	X			
CYNOGLOSSIDAE	*Cynoglossus* sp.	*8858020200*			X	
CYNOGLOSSIDAE	*Symphurus ligulatus*	*8858020121*				X
CYNOGLOSSIDAE	*Symphurus nigrescens*	*8858020603*			X	X
DASYATIDAE	*Dasyatis pastinaca*	*8713050111*	X	X		
DASYATIDAE	*Dasyatis violacea*	*8713050102*			X	
DERICHTHYIDAE	*Derichthys serpentinus*	*8741180101*	X			
DIRETMIDAE	*Diretmoides parini*	*8810010201*			X	
DIRETMIDAE	*Diretmus argenteus*	*8810010101*			X	
DIRETMIDAE	*Diretmus* sp.	*8810010100*			X	
ECHENEIDIDAE	*Remora remora*	*8835270103*	X			
ENGRAULIDAE	*Engraulis encrasicolus*	*8747020104*	X	X	X	X
GADIDAE	*Ciliata mustela*	*8791032401*	X	X	X	
GADIDAE	*Enchelyopus cimbrius*	*8791031501*	X	X		
GADIDAE	*Gadiculus argenteus argenteus*	*8791032101*	X	X	X	X
GADIDAE	*Gadus morhua*	*8791030402*	X	X		
GADIDAE	*Gaidropsarus biscayensis*	*8791032602*			X	X
GADIDAE	*Gaidropsarus macrophthalmus*	*8791032601*	X	X		
GADIDAE	*Gaidropsarus mediterraneus*	*8791032002*	X	X	X	
GADIDAE	*Gaidropsarus vulgaris*	*8791032001*	X	X	X	
GADIDAE	*Melanogrammus aeglefinus*	*8791031301*	X			
GADIDAE	*Merlangius merlangus*	*8791031801*	X	X	X	
GADIDAE	*Micromesistius poutassou*	*8791032201*	X	X	X	X
GADIDAE	*Molva dipterygia macrophthalma*		X	X		
GADIDAE	*Molva molva*	*8791031901*	X	X	X	
GADIDAE	*Molva* spp.	*8791031900*			X	
GADIDAE	*Phycis blennoides*	*8791031602*	X	X	X	X
GADIDAE	*Phycis phycis*	*8791031603*		X	X	
GADIDAE	*Phycis* spp.	*8791031600*	X			
GADIDAE	*Pollachius pollachius*	*8791030902*	X	X	X	
GADIDAE	*Pollachius virens*	*8791030901*	X	X		
GADIDAE	*Raniceps raninus*	*8791032301*	X	X		
GADIDAE	*Trisopterus esmarki*	*8791031703*	X			
GADIDAE	*Trisopterus luscus*	*8791031702*	X	X	X	X
GADIDAE	*Trisopterus minutus*	*8791031701*	X	X	X	
GASTEROSTEIDAE	*Spinachia spinachia*	*8818010501*	X			
GEMPYLIDAE	*Nesiarchus nasutus*	*8850010701*		X	X	
GEMPYLIDAE	*Ruvettus pretiosus*	*8850010401*			X	
GOBIESOCIDAE	*Apletodon dentatus*	*8784010801*	X			

FAMILIES	SCIENTIFIC NAMES	NODC CODES	CELTIC SEA B. BISCAY	NORTH OF SPAIN	PORTUGAL	GULF OF CÁDIZ
GOBIIDAE	*Aphia minuta*	*8847016601*		X		
GOBIIDAE	*Aphia minuta mediterranea*		X			X
GOBIIDAE	*Crystallogobius linearis*	*8847014901*	X	X		X
GOBIIDAE	*Crystallogobius* sp.	*8847014900*				X
GOBIIDAE	*Deltentosteus quadrimaculatus*	*8847019001*		X		X
GOBIIDAE	*Gobiidae* spp.	*8847010000*			X	
GOBIIDAE	*Gobius niger*	*8847011316*	X	X		
GOBIIDAE	*Gobius paganellus*	*8847011320*	X			
GOBIIDAE	*Lesuerigobius friesii*	*8847016702*	X	X	X	X
GOBIIDAE	*Lesueurigobius sanzoi*	*8847016704*			X	X
GOBIIDAE	*Pomatoschistus microps*	*8847015103*		X		
GOBIIDAE	*Pomatoschistus minutus*	*8847015101*	X	X	X	X
GOBIIDAE	*Pomatoschistus norvegicus*	*8847015104*		X		
GOBIIDAE	*Pomatoschistus pictus*	*8847015102*	X	X		
GONOSTOMATIDAE	*Gonostoma bathyphilum*	*8759010402*			X	
GONOSTOMATIDAE	*Gonostoma denudatum*	*8759010404*			X	X
HAEMULIDAE	*Brachydeuterus auritus*	*8835430901*				
HAEMULIDAE	*Parapristipoma octolineatum*	*8835401402*			X	
HAEMULIDAE	*Pomadasys incisus*	*8835400515*				X
HEXANCHIDAE	*Hexanchus griseus*	*8705020101*	X	X	X	
LABRIDAE	*Acantholabrus palloni*	*8839013701*	X	X	X	
LABRIDAE	*Centrolabrus exoletus*	*8839013401*	X	X		
LABRIDAE	*Coris julis*	*8839012306*		X		
LABRIDAE	*Ctenolabrus rupestris*	*8839013501*	X	X	X	
LABRIDAE	*Labrus bergylta*	*8839013603*	X	X	X	
LABRIDAE	*Labrus bimaculatus*	*8839013604*	X	X	X	
LABRIDAE	*Symphodus bailloni*	*8839013308*	X	X	X	
LABRIDAE	*Symphodus melops*	*8839013301*	X	X		
LAMNIDAE	*Isurus oxyrhinchus*	*8707040501*		X		
LAMNIDAE	*Lamna nasus*	*8707040302*	X			
LIPARIDIDAE	*Paraliparis membranaceus*	*8831091117*		X		
LOPHIIDAE	*Lophius budegassa*	*8786010104*	X	X	X	X
LOPHIIDAE	*Lophius piscatorius*	*8786010103*	X	X	X	X
LUMPENIDAE	*Lumpenus lampretaeformis*	*8842120905*	X			
MACRORAMPHOSIDAE	*Macroramphosus scolopax*	*8819030101*	X	X		X
MACRORAMPHOSIDAE	*Macroramphosus* spp.	*8819030100*			X	
MACROURIDAE	*Caelorhynchus caelorhynchus*	*8794010405*	X	X	X	X
MACROURIDAE	*Hymenocephalus italicus*	*8794011101*		X	X	X
MACROURIDAE	*Malacocephalus laevis*	*8794010601*	X	X	X	X
MACROURIDAE	*Nezumia aequalis*	*8794010801*	X	X		X
MACROURIDAE	*Nezumia sclerorhynchus*	*8794010805*		X	X	X
MACROURIDAE	*Trachyrinchus trachyrinchus*	*8794011501*	X	X	X	
MELANONIDAE	*Melanonus zugmayeri*	*8791031401*			X	
MERLUCCIIDAE	*Merluccius merluccius*	*8791040105*	X	X	X	X
MERLUCCIIDAE	*Merluccius senegalensis*	*8791040111*			X	
MOLIDAE	*Mola mola*	*8861040101*	X	X	X	
MORIDAE	*Gadella maraldi*	*8791010801*		X	X	X
MORIDAE	*Halargyreus johnsonii*	*8791010601*	X	X		
MORIDAE	*Laemonema latifrons*	*8791010203*			X	
MORIDAE	*Lepidion eques*	*8791010501*	X	X		
MORIDAE	*Lepidion* sp.	*8791010500*			X	
MORIDAE	*Mora moro*	*8791010401*	X	X	X	

FAMILIES	*SCIENTIFIC NAMES*	*NODC CODES*	CELTIC SEA B. BISCAY	NORTH OF SPAIN	PORTUGAL	GULF OF CÁDIZ
MORONIDAE	*Dicentrarchus labrax*	*8835750101*	X	X	X	
MORONIDAE	*Dicentrarchus punctatus*	*8835750102*	X	X	X	
MUGILIDAE	*Chelon labrosus*	*8836010704*	X	X		
MUGILIDAE	*Liza aurata*	*8836010902*	X	X	X	X
MUGILIDAE	*Liza ramada*	*8836010901*	X	X	X	X
MUGILIDAE	*Mugil cephalus*	*8836010101*		X	X	
MULLIDAE	*Mullus barbatus barbatus*	*8835450203*			X	X
MULLIDAE	*Mullus surmuletus*	*8835450202*	X	X	X	X
MYCTOPHIDAE	*Cerastocopelus maderensis*	*8762140102*				X
MYCTOPHIDAE	*Diaphus dumerillii*	*8762140206*		X		
MYCTOPHIDAE	*Diaphus holti*	*8762140211*				X
MYCTOPHIDAE	*Diaphus metopoclampus*	*8762140214*			X	
MYCTOPHIDAE	*Diaphus* sp.	*8762140200*		X		
MYCTOPHIDAE	*Electrona rissoi*	*8762141801*		X		X
MYCTOPHIDAE	*Hygophum benoiti*	*8762141103*				X
MYCTOPHIDAE	*Lampanyctus ater*	*8762140305*	X			
MYCTOPHIDAE	*Lampanyctus crocodilus*	*8762140317*		X		
MYCTOPHIDAE	*Lampanyctus macdonaldi*	*8762140315*	X			
MYCTOPHIDAE	*Myctophidae*	*8762140000*			X	
MYCTOPHIDAE	*Myctophum punctatum*	*8762141504*	X	X		X
MYCTOPHIDAE	*Notoscopelus kroeyerii*	*8762140405*	X	X		
MYCTOPHIDAE	*Symbolophorus veranyi*	*8762140603*	X			X
MYLIOBATIDAE	*Myliobatis aquila*	*8713070204*	X	X	X	
MYLIOBATIDAE	*Pteromylaeus bovinus*	*8713070401*				X
MYXINIDAE	*Myxine glutinosa*	*8606010201*	X			
NEMICHTHYIDAE	*Nemichthys scolopaceus*	*8741210202*		X	X	
NETTASTOMATIDAE	*Facciolella oxyrhyncha*	*8741100102*			X	X
NETTASTOMATIDAE	*Nettastoma melanurum*	*8741100201*			X	
NOMEIDAE	*Cubiceps gracilis*	*8851020203*		X	X	
NOTACANTHIDAE	*Nocacanthus bonapartei*	*8743030302*	X	X		
NOTACANTHIDAE	*Notacanthus chemnitzii*	*8743030301*			X	
OPHICHTHIDAE	*Echelus myrus*	*8741250101*			X	
OPHIDIIDAE	*Benthocometes robustus*	*8803020500*			X	
OPHIDIIDAE	*Ophidion barbatum*	*8792010607*		X		
OXYNOTIDAE	*Oxynotus centrina*	*8710010702*			X	
PARALEPIDIDAE	*Notolepis rissoi*	*8762070201*	X	X		
PARALEPIDIDAE	*Paralepis coregonoides*	*8762070402*		X		
PERISTEDIDAE	*Peristedion cataphractum*	*8826020316*			X	X
PETROMYZONIDAE	*Petromyzon marinus*	*8603010301*	X	X		
PHOLIDAE	*Pholis gunnellus*	*8842130209*	X		X	
PHOSICHTHYIDAE	*Polymetme corythaeola*	*8759010701*	X	X	X	
PHOSICHTHYIDAE	*Vinciguerria poweriae*	*8759010803*			X	
PLATYTROCTI	*Holtbyrnia macrops*	*8756040101*		X		
PLEURONECTIDAE	*Glyptocephalus cynoglossus*	*8857040502*	X	X		
PLEURONECTIDAE	*Hippoglossoides platessoides limandoides*			X		
PLEURONECTIDAE	*Limanda limanda*	*8857040904*	X			
PLEURONECTIDAE	*Microstomus kit*	*8857041202*	X	X		
PLEURONECTIDAE	*Platichthys flesus flesus*	*885704140201*	X	X	X	
PLEURONECTIDAE	*Pleuronectes platessa*	*8857041502*	X		X	
POMATOMIDAE	*Pomatomus saltatrix*	*8835250101*			X	
RAJIDAE	*Breviraja* sp.	*8713040300*				X
RAJIDAE	*Raja(Rostroraja) alba*	*8713040151*			X	

FAMILIES	*SCIENTIFIC NAMES*	*NODC CODES*	CELTIC SEA B. BISCAY	NORTH OF SPAIN	PORTUGAL	GULF OF CÁDIZ
RAJIDAE	*Raja asterias*	*8713040137*		X		X
RAJIDAE	*Raja (Dipturus) batis*	*8713040143*	X			X
RAJIDAE	*Raja brachyura*	*8713040138*	X	X	X	
RAJIDAE	*Raja (Leucoraja) circularis*	*8713040147*	X	X	X	X
RAJIDAE	*Raja clavata*	*8713040159*	X	X	X	X
RAJIDAE	*Raja (Leucoraja) fullonica*	*8713040146*	X			
RAJIDAE	*Raja microocellata*	*8713040140*	X	X	X	
RAJIDAE	*Raja miraletus*	*8713040136*			X	
RAJIDAE	*Raja montagui*	*8713040141*	X	X	X	
RAJIDAE	*Raja (Leucoraja) naevus*	*8713040148*	X	X	X	X
RAJIDAE	*Raja (Dipturus) nidarosiensis*	*8713040144*	X			
RAJIDAE	*Raja (Dipturus) oxyrhinchus*	*8713040145*	X		X	X
RAJIDAE	*Raja* spp.	*8713040100*			X	
RAJIDAE	*Raja undulata*	*8713040158*	X	X	X	
SALMONIDAE	*Salmo salar*	*8755010305*	X	X		
SCIAENIDAE	*Argyrosomus regius*	*8835442701*	X	X		
SCIAENIDAE	*Umbrina canariensis*	*8835441107*	X			X
SCOMBERESOCIDAE	*Scomberesox saurus*	*8803030201*	X	X	X	
SCOMBRIDAE	*Auxis rochei*	*8850030701*			X	
SCOMBRIDAE	*Sarda sarda*	*8850030202*	X			
SCOMBRIDAE	*Scomber japonicus*	*8850030301*	X	X	X	X
SCOMBRIDAE	*Scomber scombrus*	*8850030302*	X	X	X	X
SCOPHTHALMIDAE	*Lepidorhombus boscii*	*8857032301*	X	X	X	X
SCOPHTHALMIDAE	*Lepidorhombus whiffiagonis*	*8857032302*	X	X	X	X
SCOPHTHALMIDAE	*Phrynorhombus norvegicus*	*8857032201*		X		
SCOPHTHALMIDAE	*Phrynorhombus regius*	*8857032202*	X	X		
SCOPHTHALMIDAE	*Psetta maxima maxima*	*8857030402*	X	X	X	
SCOPHTHALMIDAE	*Scophthalmus rhombus*	*8857030403*	X	X	X	
SCOPHTHALMIDAE	*Zeugopterus punctatus*	*8857032101*	X			
SCORPAENIDAE	*Helicolenus dactylopterus*	*8826010301*	X	X	X	X
SCORPAENIDAE	*Pontinus kuhlii*	*8826010509*			X	
SCORPAENIDAE	*Scorpaena loppei*	*8826010625*	X	X		
SCORPAENIDAE	*Scorpaena notata*	*8826010627*	X	X	X	X
SCORPAENIDAE	*Scorpaena porcus*	*8826010629*	X	X		
SCORPAENIDAE	*Scorpaena scrofa*	*8826010628*	X	X	X	
SCORPAENIDAE	*Scorpaena* spp.	*8826010600*			X	
SCORPAENIDAE	Scorpaenidae	*8826010000*			X	
SCORPAENIDAE	*Trachyscorpia cristulata echinata*		X	X	X	
SCYLIORHINIDAE	*Galeus melastomus*	*8708010203*	X	X	X	X
SCYLIORHINIDAE	*Galeus* spp.	*8708010200*			X	
SCYLIORHINIDAE	*Scyliorhinus canicula*	*8708010306*	X	X	X	X
SCYLIORHINIDAE	*Scyliorhinus stellaris*	*8708010307*	X	X		X
SERRANIDAE	*Anthias anthias*	*8835020723*			X	X
SERRANIDAE	*Callanthias ruber*	*8835025401*		X	X	X
SERRANIDAE	*Polyprion americanus*	*8835022801*	X		X	
SERRANIDAE	*Serranus cabrilla*	*8835022316*		X	X	X
SERRANIDAE	*Serranus hepatus*	*8835022314*			X	X
SERRIVOMERIDAE	*Serrivomer beani*	*8741200102*			X	
SETARCHIDAE	*Setarches guentheri*	*8826011001*			X	
SOLEIDAE	*Bathysolea profundicola*	*8858011001*	X	X	X	
SOLEIDAE	*Buglossidium luteum*	*8858010801*	X	X	X	
SOLEIDAE	*Dicologoglossa cuneata*	*8858012101*	X	X	X	X

FAMILIES	SCIENTIFIC NAMES	NODC CODES	CELTIC SEA B. BISCAY	NORTH OF SPAIN	PORTUGAL	GULF OF CÁDIZ
SOLEIDAE	*Microchirus azevia*	8858010902		X	X	X
SOLEIDAE	*Microchirus boscanion*	8858010904			X	
SOLEIDAE	*Microchirus ocellatus*	8858010901			X	
SOLEIDAE	*Microchirus variegatus*	8858010903	X	X	X	X
SOLEIDAE	*Solea lascaris*	8858010610	X	X	X	
SOLEIDAE	*Solea senegalensis*	8858010614		X	X	X
SOLEIDAE	*Solea vulgaris*	8858010601	X	X	X	X
SPARIDAE	*Boops boops*	8842010104	X	X	X	X
SPARIDAE	*Dentex dentex*	8835431005			X	
SPARIDAE	*Dentex gibbosus*	8835431003				X
SPARIDAE	*Dentex macrophthalmus*	8835431002			X	
SPARIDAE	*Dentex maroccanus*	8835431001			X	
SPARIDAE	*Dentex* spp.	8835431000			X	
SPARIDAE	*Diplodus annularis*	8835430404				X
SPARIDAE	*Diplodus bellottii*	8835430407			X	X
SPARIDAE	*Diplodus cervinus cervinus*	8835430405	X	X	X	
SPARIDAE	*Diplodus puntazzo*	8835431701		X	X	
SPARIDAE	*Diplodus sargus*	8835430403	X	X		
SPARIDAE	*Diplodus sargus cadenati*				X	
SPARIDAE	*Diplodus vulgaris*	8835430406	X	X	X	X
SPARIDAE	*Lithognathus mormyrus*	8835431601	X	X		X
SPARIDAE	*Pagellus acarne*	8835430802	X	X	X	X
SPARIDAE	*Pagellus bellotii*	8835430803				X
SPARIDAE	*Pagellus bogaraveo*	8835430801	X	X	X	X
SPARIDAE	*Pagellus erythrinus*	8835430804	X	X	X	X
SPARIDAE	*Pagrus auriga*	8835430603				X
SPARIDAE	*Pagrus pagrus*	8835430601	X	X	X	
SPARIDAE	*Sarpa salpa*	8835431801	X	X	X	
SPARIDAE	*Sparus aurata*	8835431101	X	X	X	
SPARIDAE	*Spondyliosoma cantharus*	8835431201	X	X	X	
SPHYRAENIDAE	*Sphyraena sphyraena*	8837010118				X
SQUALIDAE	*Centrophorus granulosus*	8710010301			X	X
SQUALIDAE	*Centrophorus squamosus*	8710010302	X	X		X
SQUALIDAE	*Centroscymnus coelolepis*	8710011201	X	X		
SQUALIDAE	*Centroscymnus crepidater*	8710011202	X			
SQUALIDAE	*Dalatias licha*	8710010401	X	X	X	
SQUALIDAE	*Deania calceus*	8710011401	X	X	X	X
SQUALIDAE	*Etmopterus pusillus*	8710010504			X	
SQUALIDAE	*Etmopterus spinax*	8710010510	X	X	X	X
SQUALIDAE	*Scymnodon ringens*	8710011601	X	X	X	
SQUALIDAE	*Squalus acanthias*	8710010201	X	X		
SQUALIDAE	*Squalus blainvillei*	8710010202			X	
SQUATINIDAE	*Squatina squatina*	8711010103	X			
STERNOPTYCHIDAE	*Argyropelecus aculeatus*	8759020102			X	
STERNOPTYCHIDAE	*Argyropelecus gigas*	8759020105		X		
STERNOPTYCHIDAE	*Argyropelecus hemigymnus*	8759020106	X	X	X	
STERNOPTYCHIDAE	*Argyropelecus olfersi*	8759020107	X	X		
STERNOPTYCHIDAE	*Maurolicus muelleri*	8759010501	X	X	X	
STICHAEIDAE	*Chirolophis ascanii*	8842120505	X			
STOMIIDAE	*Stomias boa*	8759070205		X	X	
STOMIIDAE	*Stomias boa ferox*	8759070202	X			
STROMATEIDAE	*Stromateus fiatola*	8851030501			X	

FAMILIES	SCIENTIFIC NAMES	NODC CODES	CELTIC SEA B. BISCAY	NORTH OF SPAIN	PORTUGAL	GULF OF CÁDIZ
SYNAPHOBRANCHIDA	Synaphobranchus kaupi	8741150104	X	X		
SYNGNATHIDAE	Hippocampus hippocampus	8820020209	X	X		
SYNGNATHIDAE	Nerophis ophidion	8820022202	X			
SYNGNATHIDAE	Syngnathus acus	8820020120	X			
SYNGNATHIDAE	Syngnathus rostellatus	8820020119	X			
SYNGNATHIDAE	Syngnathus spp.	8820020100	X			
SYNGNATHIDAE	Syngnathus typhle	8820020123	X	X		
TETRAODONTIDAE	Ephippion guttiferum	8861011401			X	
TETRAODONTIDAE	Sphoeroides pachygaster	8861010209			X	
TETRAODONTIDAE	Tetraodontidae	8861010000			X	
TORPEDINIDAE	Torpedo marmorata	8713030105	X	X	X	X
TORPEDINIDAE	Torpedo nobiliana	8713030102	X			X
TRACHICHTHYIDAE	Hoplostethus atlanticus	8810020201		X	X	
TRACHICHTHYIDAE	Hoplostethus mediterraneus	8810020202	X	X	X	X
TRACHINIDAE	Echiichtys vipera	8840060101	X	X	X	
TRACHINIDAE	Trachinus draco	8840060102	X	X	X	X
TRIAKIDAE	Galeorhinus galeus	8708020102	X	X	X	X
TRIAKIDAE	Mustelus asterias	8708020408	X	X		
TRIAKIDAE	Mustelus mustelus	8708020409	X	X	X	
TRICHIURIDAE	Aphanopus carbo	8850020301	X	X	X	
TRICHIURIDAE	Benthodesmus elongatus symoni	8792011201			X	X
TRICHIURIDAE	Lepidopus caudatus	8850020401	X	X	X	X
TRICHIURIDAE	Trichiurus lepturus	8850020201		X	X	
TRIGLIDAE	Aspitrigla cuculus	8826020801	X	X	X	
TRIGLIDAE	Aspitrigla obscura	8826020802	X	X	X	X
TRIGLIDAE	Eutrigla gurnardus	8826020601	X	X	X	
TRIGLIDAE	Lepidotrigla cavillone	8826020413	X	X	X	X
TRIGLIDAE	Lepidotrigla dieuzeidei	8826020414				X
TRIGLIDAE	Trigla lucerna	8826020501	X	X	X	X
TRIGLIDAE	Trigla lyra	8826020502	X	X	X	X
TRIGLIDAE	Trigla spp.	8826020500			X	
TRIGLIDAE	Trigloporus lastoviza	8826020701	X	X	X	X
URANOSCOPIDAE	Uranoscopus scaber	8840140802		X	X	X
XIPHIIDAE	Xiphias gladius	8850040101	X		X	
ZEIDAE	Cyttopsis roseus	8811030101	X	X	X	
ZEIDAE	Zenopsis conchifer	8811030202			X	X
ZEIDAE	Zeus faber	8811030301	X	X	X	X
ZOARCIDAE	Melagnostigma atlanticum	8793010902		X		

12 Review of the IBTS Manual

12.1 Introduction

Groundfish surveys provide the most appropriate data available for the examination of large-scale spatial and temporal analyses of fish communities for offshore waters, and therefore for the derivation of metrics with which to assess changes in the structure, function and diversity of fish communities. Such studies can then address issues such as the impacts of fishing and climate change on fish communities. Hence, WGFE request that IBTS recognise the fact that groundfish survey data are vital for assessing the status of non-target fish species and fish communities as a whole. This means that quality assurance procedures ought to be in place to ensure that not only are fishing protocols standardised, but that also catch sampling and sub-sampling are appropriate for community analyses. For example, when sub-sampling, the entire catch should be sorted for rarer fish species, with only a restricted selection of species combined for subsequent sub-sampling.

Taxonomic rigour in species identification has been emphasised as an important issue for surveys. It has been highlighted that the IBTS has some potential problems associated with data entry errors and more fundamental problems associated with the mis-identification of selected taxa, primarily non-commercial fish species (Daan, 2001). Additionally, there are several taxa that some member states report at either a species, generic or family level. Although these problems will have no impact on the assessment of commercial fish stocks, it does have implications on the utility of the IBTS dataset for studies on fish communities, including diversity metrics and other potential indicators. Potential problematic taxa in the North Sea and in the southern and western IBTS surveys include:

- Deep-water sharks (Squalidae)
- Smoothhounds (*Mustelus* spp.)
- Skates and rays (Rajidae)
- Shads (*Alosa* spp.)
- Argentines (*Argentina* spp.)
- Rocklings (Gadidae, Lotinae)
- Rat-tails (Macrouridae)
- Clingfishes (Gobiesocidae)
- Sticklebacks (Gasterosteidae)
- Myctophids (Myctophidae)
- Hatchet fish (Sternoptychidae)
- *Beryx* spp.
- *Hoplostethus* spp.
- Pipefish (Syngnathidae)
- Redfish (*Sebastes* spp.)
- Scorpion fish (*Scorpaena* sp.)
- Sea scorpions (Cottidae)
- Sand eels (Ammodytidae)
- Dragonets (*Callionymus* spp.)
- Wrasse (Labridae)
- Eelpouts (Zoarcidae)
- Snake blennies (Stichaeidae)
- Mullets (Mugilidae)
- Gobies (Gobidae)
- Sea breams (Sparidae)
- Horse mackerel (*Trachurus* spp.)
- Flatfish (certain sister taxa, e.g., *Bathysolea-Diclogoglossa*)

12.2 Methods of improving Quality Assurance procedures for fish identification

WGFE considered that there were several ways that the national fisheries laboratories could improve data quality for non-target fish species:

1. The development and dissemination of user-friendly keys (including photographic keys) for problematic taxa, which should improve data quality and also ensure comparability between national survey data sets.

2. The more unusual species caught on surveys should be photographed and preserved or frozen for subsequent verification, with specimens deposited in national or regional museums where appropriate.

3. Laboratories could consider establishing and maintaining a reference collection of the fish species encountered in their surveys, with particular reference to problematic taxa and rare and unusual species.

4. Software for the input of catch data could include filters to flag those fish species that are outside their normal geographic, bathymetric range or outside normal size limits. The DATRAS database has such filters.

5. Better training for sea going staff to ensure taxonomic expertise is improved and that the need for correct fish identification is highlighted as an important element of the surveys. Testing procedures to ensure quality control should be recommended.

12.3 References

Daan, N. 2001. The IBTS database: a plea for quality control. ICES CM 2001/T:03.

13   Recommendations for future work and election of new Chair

The WGFE unanimously recommends that Dr A. Daniel Duplisea, Fisheries & Oceans Canada, Mont-Joli, Canada should be invited to Chair WGFE from 1 January 2006.

Issues regarding threatened fishes, fish communities and many other aspects of fish ecology are increasingly asked of ICES. Hence, it was considered that WGFE should meet next year in Copenhagen. The following potential work areas for WGFE were suggested:

a)   establish standardised protocols for filtering survey data to ensure that subsequent statistical analyses are comparable across a range of scales; (b) liaise with other ICES Working Groups to collate a temporal series of fishing mortality rates for the main species of the assemblages to provide estimates of multispecies F at appropriate spatial scales; (c) define what a 'large fish' is; (d) evaluate how a suite of indicators change in relation to estimated trends in multispecies F; and (e) use simulation tools to evaluate the sensitivity of various EcoQO indicators to multispecies F.

The development of EcoQOs for fish communities and threatened and declining fish species are required by OSPAR. This work supports Action Point 2.2.2, and ultimately Action Points 2.2.1 and 3.2.

b)   Further work regarding the abundance-occupancy relationships should be undertaken, with special reference to fisheries and ecosystem management issues, and the underlying mechanisms that affect such relationships.

Abundance-range size relationships show clear potential links to other work covered by the group (e.g., fish habitat issues and the development of EcoQOs). This work supports Action Points 1.2.1 and 1.2.2.

c)   Continue studies on food rations and prey composition of North Sea fishes by (a) re-evaluating predation mortalities of the MSVPA prey fish populations, and examine the consequences by relevant runs of MSVPA/FOR when using food rations of MSVPA predators obtained by application of a new mechanistic gastric evacuation model rather than food rations used at present by the ICES, and (b) estimate food rations and prey compositions of grey gurnard, horse mackerel, and mackerel in the North Sea, applying new information about gastric evacuation rates.

This work supports Action Points 1.2.1, 3.5 and 4.3.

d)   Address any upcoming nature conservation issues for marine fishes.

This work supports Action Points 1.2 and 2.2.

e)   Continue the descriptions of fish habitat, to support studies on threatened, commercial and selected non-target species.

Such studies have implications to management issues and will also aid in the interpretation of abundance-range size relationships. This work supports Action Points 1.2.1 and 1.4.2.

f)   Obtain better estimates of relative catchabilities of marine fishes, on a size-specific basis when appropriate.

This work underpins many of the work areas above, therefore supporting studies in relation to Action Points 1.2, 1.11 and 1.13.

g)   Liaise with IBTS to continue studies on the broadscale spatial and temporal patterns in selected fish species and communities along the European continental shelf of the eastern North Atlantic (e.g., the area covered by parts of ICES divisions VI-IX).

A more regional approach may provide a better understanding of the spatial-temporal dynamics of fish communities in these ICES areas. This work supports Action Points 1.2.2 and 1.6.

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Annex 2: Working documents

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