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5–9 March 2007

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Participants in the 2007 Working Group on Fish Ecology Meeting, 5–9 March 2007, IFREMER Nantes, France. From left to right: Verena Trenkel, Marie-Joëlle Rochet, Benjamin Planque, Tom Blasdale, Daniel Duplisea, Helen Drewery/Fraser, Simon Greenstreet, Ronald Fricke, Anna Rindorf, Remment ter Hofstede, Lena Bergström, Jim Ellis, Anik Brind'Amour, Hugues Benoît, Pierre Petitgas, Thomas de Lang Wenneck, Nick Dulvy, Dave Kulka, Mike Frisk.

Executive Summary

Report Highlights

- An analysis of species distributional responses to environmental variables as requested by OSPAR (Section 2) showed species-specific response that could not be solely attributed to any one environmental variable. Mechanisms for responses likely differ between species and fishing effects on abundance within distributions cannot be discounted as important factors.

- A spawner biomass versus fishing mortality plot, common in a single species precautionary approach framework, was constructed using output from MSVPA for the North Sea. Though there are some major assumptions required to construct this sort of plot for a community, it can help identify targets and potentially reference points for community indicators. This framework was developed and interpretation of the proportion of large fish indicators and mean fish weight indicator were explored with this indicator in Section 6.

Executive Summary

In 2007 WGFE carried out work on some of its carry-over themes of research related to abundance-occupancy relationships (Section 3) and essential fish habitat (Section 4) and development of ecosystem quality objectives (Section 5). In addition to these work areas, WGFE responded to two specific requests from OSPAR: (1) examine the impacts of physical environmental changes on the distribution and abundance of fish in the OSPAR area (TOR a), Section 2). (2) Using North Sea fish community models, determine recovery trajectories and times to an appropriate target point for the proportion of large fish and mean fish size community indicator (TOR f), g), Section 6). The first of these terms of reference was organised jointly with the Working Group on the Ecology and Life History of Small Pelagics (WGLESP) and the work presented in Section 2 of this report must be considered a common contribution by WGFE and WGLESP.

The OSPAR request on fish distributional changes work involved the extraction or large amounts of spatial referenced fish abundance and temperature data. A large part of the report is devoted to this work owing to the fact that it is a joint section with another working group and because of the diversity of analyses that this work can support. It was found that three distinct temperature regimes in time can be identified in the OSPAR area and that the spatial changes in fish during these regimes were species specific. Pelagic species showed the greatest changes while demersal fish distributions varied less over the temperature regimes though some demersals showed increases or decreases in area with temperature regime. Fishing effects were not considered in the analysis but clearly changes in abundance owing to fishing could confound interpretation of environmentally driven distributional changes. Terms of reference for the 2008 meeting specifically includes wording to address this issue.

Exploration of the mechanisms behind distribution changes were analysed further in Section 2. Specific questions such where do individuals come from during expansion, what are their optimal habitat preferences and can this be used to explain distributional changes were addressed and specifically why and when a habitat might be selected by a species or individual. Tools were outlined that could help answer these questions and case studies developed.

Advances in understanding abundance-occupancy relationships were again made at WGFE in 2007. The Southern Gulf of St. Lawrence fish community was used as a case study in a species-by-species analysis showing that species specific responses were different. Including temperature as a surrogate for suitable range area sometimes affected the patterns observed, though a strong effect was not clear. Though results for a temperature as habitat suitability

surrogate were not strong, this analysis went beyond a purely phenomenological description of abundance-occupancy relationship into suitable habitat and habitat selection thus advanced the ideas further into exploration of mechanism.

Considerable time and effort were devoted to essential fish habitat issues at the 2007 meeting and useful progress was made. Several studies showed the dependence of fish species distributions on habitat features such as depth, temperature and benthic community composition. These distributions were usually broken down into age and/or size categories thus further refining dependence on a particular habitat as a nursery or adult area. Though the definition of nursery can be ambiguous this was often defined as the first age or peak in a length frequency distribution which a survey could detect. Unfortunately, owing to problems with sampling pre-benthic stages of demersal species, little could be said about habitat dependence for the earliest of life stages. Furthermore, deficiencies in habitat data meant that maps could not be constructed at the necessary scale of resolution to define an area of congregation for a particular species. This is unfortunate as many of the arguments related to the conception of essential fish habitat are grounded in small scale dependence of species on areas. In areas where appropriate data exist, it is likely further progress of defining essential fish habitat can be made but much of the larger scale inference will remain a more speculative exercise until appropriate data or surrogate methods become available.

The interconnectedness of the issues in the ToRs related to OSPAR fish distribution, abundance-occupancy and essential fish habitat arose out of the 2007 meeting. It became clear the when one seeks mechanism for distributional changes then issues of habitat choice and utilisation of area as a function of population size are key processes. Clearly working with these issues together in future should provide not just scientific products such as maps and distributions but could closely tie these products to mechanisms. Studies at this level therefore could give us the ability to disentangle causes of distributional changes, provide advice on issues related to marine protected areas (MPA) and predict impact of fishing and climate on efficacy of MPA management measures. WGFE has suggested terms of reference for 2008 that specifically aim to advance this synthesis of observation and mechanism.

In 2007 WGFE continued its analysis of EcoQO fish community indicators. A meta-analysis of size-based indicators over several French survey datasets revealed inconsistencies between indicators for the direction of change within systems. The most sensitive indicator appeared to the proportion of large fish biomass defined as the mean 95th percentile of length. Diversity size spectra were explored in different fishing intensity blocks using the Scottish groundfish survey series dating back to the 1920s. Different diversity indices had variety of patterns and some were relatively insensitive fishing intensity. Some size classes of fish showed sensitivity to fishing effects but diversity size spectrum slopes were not useful to characterise the community response to fish pressure.

The proportion of large fish and mean fish weight indicators (PLF) again figured prominently in WGFE's work in 2007. The specific task of determining defensible methods for target setting for these indicators and simulating the kinds of fishing scenarios it would take to reach them was undertaken. Survey time series for the North Sea were used to determine the variability of the indicators while similar analyses were made for output from the most recent accepted North Sea MSVPA run. MSVPA results also allowed the development of F vs B plots similar to the ICES single species precautionary approach stock management scheme but in this case based on multispecies F. Using these methods; the late 1960s was suggested as a reasonable target period. Single species projections for the seven MSVPA demersal species were performed under different fishing scenarios. These projections suggested a recovery time if fishing stocks at Fpa would be on the order of 10 years; however, there are many caveats in interpreting these single species projections in a multispecies context and recommendations are provided for future work of this nature if the PLF indicators are to be used further in ecosystem approach to fisheries management.

One of the main advances made in WGFE in 2007 was the emphasis on analysis and explanation of mechanism behind fish movements. This perhaps reflects a natural maturation of WGFE beyond an initial pattern exploration phase to a phase where mechanistic explanations are proposed and explored. This is a reflection of the continuity of group membership (thus of a core set of questions) plus the addition of new participants (thus an injection of new ideas) to the group each year. Continuing this membership and attendance formula combined with continued work on core issue such as essential fish habitat, abundance-occupancy and community indicators will ensure a dynamic and fruitful future for the basic and applied science products produced by WGFE.

1 Introduction

1.1 Terms of reference

2006/2/LRC03 The Working Group on Fish Ecology [WGFE] (Chair: D. Duplisea, Canada) will meet back-to-back with the Working Group on Life History and Ecology of Small Pelagic Fish [WGLESP] in Nantes, France, from 5–9 March 2007 to:

a) assess and report on changes in the distribution, population abundance and condition of fish in the OSPAR maritime area in relation to changes in hydrodynamics and sea temperature, drawing on expertise from assessment groups as appropriate. Coordinate with WGLESP Chair as WGLESP is providing a response to this request for some pelagic fishes. (Further details on the interpretation and handling of this ToR will be provide by ACE);

b) assess and report on the extent to which the changes reported in (a) can reliably be attributed to changes in hydrodynamics and sea temperature. (Further details on the interpretation and handling of this ToR will be provide by ACE);

c) EcoQOs: continue analyses of the sensitivity, response and specificity of fish community indicators using simulation approaches and supporting empirical analyses;

d) Essential fish habitat:

- i) study the functional coupling between fish and their biotic and abiotic environment to identify the characteristics of essential habitats for fish species (and life-history stages) of interest. Examine the distributions of demersal and pelagic fish in relation to habitat properties, and identify those ecological, physiological and behavioural components that may affect the distribution of fish.
- ii) Estimate the cumulative area representing (1) the core abundance of eggs, larvae and nursery areas of commercial species; (2) the survey abundance of all fish species completing their total life cycle within a particular management area as a hypothetical implementation of essential fish habitat (EFH) protection.
- iii) Explore the utility of using IBTS and other national data to identify the broadscale distribution of nursery grounds of commercial and vulnerable fish species in the ICES area.
- iv) Overlay fish distribution maps with habitat and environmental layers for available data as an exploratory exercise for developing hypotheses on mechanisms;

e) Abundance-Occupancy:

- i) 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 and to examine new techniques for analysis and compared between more species, life-history stages and areas.
- ii) look for difference in the nature of the abundance-occupancy relationship within a species but between populations in the ICES and compare with the same species in distant areas (e.g. NAFO) and attempt to relate any difference to historical ecological, environmental and/or fishery conditions.
- iii) examine how fishery catchability is likely to change in the presence or absence of abundance-occupancy relationships.

f) from current model population estimates and survey data show the historical trend in the proportion of large fish and mean fish weight (North Sea);

g) for current models of North Sea fish communities:

- i) determine future trajectories of the proportion of large fish (<30 cm) and mean fish weight under different scenarios of fishing mortality.

ii) from (g-i) determine the time to recover to reference levels in the early 1980s as determined in (f) for the selected indicators.

1.2 Participants

The following scientists attended the Working Group meeting. Full contact details are given in Annex 1:

Hugues Benoît (DFO, Canada)
Lena Bergström (Fiskeriverket, Sweden)
Tom Blasdale (JNCC, UK)
Anik Brind'Amour (IFREMER, France)
Helen Drewery/Fraser (FRS, UK)
Nick Dulvy (CEFAS, UK)
Daniel Duplisea (DFO, Canada)
Jim Ellis (CEFAS, UK)
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Remment ter Hofstede (IMARES, Netherlands)
Verena Trenkel (IFREMER, France)
Sandrine Vaz (IFREMER, France)
Thomas de Lang Wenneck (IMR, Norway)

1.2.1 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 2005, and continued ecological studies, including the development and testing of Ecological Quality Objectives (EcoQOs) for fish communities, abundance – occupancy relationships, and the relative catchability of fishes in different survey gears, evaluation of decline criteria used by various conservation organisations (ICES, 2004; ICES, 2005). In 2006 WGFE continued with these core pieces or work and developed a broad road map to steer future directions of the working group (ICES 2006). 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.

1.2.2 References

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

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

ICES. 2005. Report of the Working Group on Fish Ecology. ICES CM 2005/G:05; 220 pp.

ICES. 2006. Report of the Working Group on Fish Ecology. ICES CM 2006/G:06; 154 pp.

2 Changes in fish distribution and condition in the OSPAR area in relation to environmental variables

ToR – a) assess and report on changes in the distribution, population abundance and condition of fish in the OSPAR maritime area in relation to changes in hydrodynamics and sea temperature.

WGFE has not previously addressed the subject of changes in fish distribution, abundance and condition in relation to changing environmental conditions; however, changes in fish distribution in relation to changes in sea temperature in the North Atlantic have been addressed elsewhere in the past. Perry *et al*. (2005) examined the distributions of demersal species in North Sea trawl surveys between 1977 and 2001. 21 of the 36 species included in the study showed changes in their centers of distribution in response to changing water temperatures. Of 20 species which had their northern distributional limit in the North Sea, half extended their limits towards higher latitude or deeper water. Distributional shifts occurred more commonly in species with high rates of population growth and small body size at maturity. Beare *et al*. (2003 and 2004) documented recent migrations of a number of Atlantic species into the northern North Sea and suggested that these changes might be related to warmer temperatures.

Other studies have focused on the response of individual species to environmental change e.g. Rogers, 1985; Johannessen *et al*., 2004 documented a large scale change in the distribution of cod in Greenlandic waters in response to the large-scale warming of the North Atlantic during the 1920s and 1930s. Similar distributional shifts were noted in other parts of the range.

This chapter addresses a portion of the broad ToR, namely changes in distribution in relation to changes in sea temperature. It is a relative long chapter containing three distinct analyses: (1) Section 2.1.3.2 contains a series of maps showing changes in distributions of select pelagic and demersal species in three different temperature periods. Section 2.1.3.3 contains an analysis of the same species using a technique called quotient plots. Thus Sections 2.1.3.2 and 2.2.3.3 are duplicate analyses of the same data set using two different techniques. (2) Sections 2.2–2.5 explore mechanisms for changes in distribution which relate to environmental preferenda, migration and spawning habitat, food supply etc. (3) Finally, Section 2.6 is an analysis independent of previous sections which explores how North Sea fish distributions have changed in response to various environmental drivers.

2.1 Changes in species distributions in relation to temperature

2.1.1 Data

2.1.1.1 Temperature

Data from the Comprehensive Ocean Atmosphere Data Set (COADS) over the North and Barents Seas is used to analyse interannual variability in sea surface temperature. The data consists of monthly averages on SST over the entire North Sea for the period 1960–2005. This information was used to examine change in thermal conditions over time.

Bottom temperature reading from the North Atlantic were extracted from the ICES Oceanographic database from 1977–2006 to evaluate relationship of thermal conditions to inter-annual and seasonal variability in species distribution.

2.1.1.2 Trawl Surveys

Fish distributions mapped from the Barents Sea using data from 5 surveys. All of the surveys except one (Campelen shrimp trawl for the Barents Sea and off the west coast of Norway)

employed the GOV trawl (with some modifications in the different areas). No attempt was made to standardize among gears. However, a comparison of catch rates of GOV and Campelen gears in an area of overlap in the northern extent of the North Sea suggest that GOV rates are considerably higher. Thus, the Campelen data that extends into the North Sea was removed from the analysis and catch rates are on a different scale north of Lat. 62^{O} vs. south of that line. Thus, the Norwegian/east coast and Barents Sea are at different scale of fish density than the North Sea and areas to the west and south. However, comparison between time periods and quarters can be made within each of the two areas.

Twenty-two pelagic and demersal species were selected for temporal-spatial analyses (Table 2.1.1).

The aim of this section is to assess changes in the distribution and abundance of marine species in the OSPAR maritime area in relation to hydrodynamics and sea temperature. Twenty-two ecologically indicative pelagic and demersal species (ones expected to show responses to environmental change), including the threatened and declining species identified by OSPAR, were selected for analysis. In terms of temporal and spatial scale, we looked at 3 periods over 29 years, when the thermal conditions changed and where there was sufficient survey data. The study encompassed the Cantabrian to Barents Seas, including the North Sea where previous studies have indicated significant change over this period. We then selected a sub-set of these species to examine in more detail, life history mechanisms and habitat associations related to the changes in the spatial patterns observed. The ToR is broad in scope and it was not possible to comprehensively address all aspects in the present analyses. As such, future work should expand on these results to relate in particular changes in abundance and condition of fish.

Table 2.1.1. List of 5 pelagic and 17 demersal selected for analysis. Maximum density refers to the median number per tow in the highest category of fish density. The column "Distribution Changes" summarizes increases and decreases in range and density of fish between periods (P1=1977–1989, P2=1990–1999, P3=2000–2005) examined. NC= no change, I=increase, D=decrease. Species are placed in order of pelagic then demersal, least change to greatest change observed.

	SPECIES (LEAST TO MOST CHANGE) PELAGIC – MINIMAL CHANGE	MAXIMUM DENSITY	DISTRIBUTION CHANGES
1	*Sprattus sprattus* (sprat)	12,565	P1-P2-**NC**, P2-P3-**NC**
	Pelagic – significant change		
1	*Clupea harengus* (herring)	7,227	P1-P2-**D**, P2-P3-**D** – **Barents Sea** only
2	*Trachurus trachurus* (horse mackerel)	5,952	P1-P2-**I**, P2-P3-**NC** – **North Sea**
3	*Sardina pilchardus* (sardine)	300	P1-P2-**I**, P2-P3-**I – all areas**
4	*Engraulis encrasicolus* (anchovy)	46,850	P1-P2-**I**, P2-P3-**I – North Sea**
	Demersal – minimal change		
1	*Capros asper* (boarfish)	9,102	P1-P2-**NC**, P2-P3-**NC**
2	*Pleuronectes platessa* (American plaice)	400	P1-P2-**NC**, P2-P3-**NC**
3	*Melanogrammus aeglefinus* (haddock)	1967	P1-P2-**NC**, P2-P3-**NC**
4	*Merlangius merlangus* (whiting),	3,000	P1-P2-**NC**, P2-P3-**NC**
5	*Solea solea* (sole)	60	P1-P2-**NC**, P2-P3-**NC**
6	*Pollachius virens* (saithe)	600	P1-P2-**NC**, P2-P3-**NC**
	Demersal – significant change		
1	*Merluccius merluccius* (hake)	240	P1-P2-**I**, P2-P3-**I - Nor to Cantab Sea**
2	*Amblyraja radiata* (starry ray)	15	P1-P2-**I**, P2-P3-**I - Nor to Barents Sea**
3	*Mullus surmuletus* (striped red mullet)	11	P1-P2-**I**, P2-P3-**I - North Sea**
4	*Lophius piscatorius* (anglerfish)	6	P1-P2-**I**, P2-P3-**I - Nor to Cantab Sea**
5	*Zeus faber* (John Dory)	10	P1-P2-**I**, P2-P3-**I - Nor to Cantab Sea**
6	*Scyliorhinus canicula* (lesser spotted dogfish)	100	P1-P2-**I**, P2-P3-**I - Nor to Cantab Sea**
7	*Trisopterus luscus* (bib)	1,000	P1-2-**D**, P2-P3-**D - North Sea**
8	*Gadus morhua* (Atlantic cod)	554	P1-P2-**D**, P2-P3-**D - Nor to Bare Sea**
9	*Squalus acanthias* (spurdog)	15	P1-P2-**D**, P2-P3-**D** - Nor to Bare Sea
10	*Raja clavata* (thornback ray)	3	P1-P2-**D**, P2-P3-**D - Nor to Bare Sea**
11	*Helicolenus dactylopterus* (bluemouth redfish)	25	P1-P2-**I**, P2-P3-D - **Nor to Canta Sea**

Given possible seasonal changes in distribution, analyses were done by quarter (1, 3 and 4). Distribution was mapped for three intervals corresponding to periods of different thermal conditions.

Table 2.1.2.Overview of surveys used in analyses.

COUNTRY	SURVEY	Q	GEAR	DESIGN	SINCE	DATRAS
Den-Eng-Fra-Ger-Sco-Net-Nor-Swe	North Sea IBTS	1;3	GOV	by ICES rectangle	1965	Y
Scotland	West of Scotland (Rockall) - Deep Water Survey	3	GOV BT184	by ICES rectangle	1985	N
Scotland	Western Division Bottom Trawl Survey	1	GOV	by ICES rectangle	1981	Y
Scotland	Scottish Mackerel Recruit (SMR)	4	GOV	by ICES rectangle	1985	N
Ireland	West coast Groundfish Survey	4	rockhopper	by ICES rectangle	1990	N
Ireland	Irish Sea-Celtic Sea Groundfish Surveys	4	GOV	by ICES rectangle	1997	N
Northern Ireland	Irish Sea	1;4	rockhopper	stratified by depth and seabed-type with fixed stations	1992	N
England	Celtic Sea and Western Approaches Groundfish Survey	1	PHHT	Fixed by area and depth strata	1981	N
England	Irish Sea and Celtic Sea	4	GOV	Fixed stations in strata	2004	N
France	Celtic Sea and Bay of Biscay Groundfish Survey	4	GOV	stratified random by area and depth	1987	Y
Portugal	Groundfish Survey	3;4	NCT	Fixed	1979	N

2.1.2 Analysis

2.1.2.1 Temperature

From sea surface temperature averages, monthly anomalies were calculated and then averaged by quarter (Months 1–3, 4–6, 7–9, 10–12). Time series for each quarter was standardised (zero mean, unit variance). The dissimilarity between individual years was calculated as the Euclidean distance between each year in the space of coordinate formed by the 4 quarter standardised temperature anomalies. In this analysis 2 years are considered similar if their temperature anomalies are similar for all four quarter (this is slightly different than just looking at average annual temperature anomaly. For example two years with average annual anomalies can have very distinct seasonal patterns with e.g. year 1 having negative anomalies in winter and positive ones in summer and year 2 showing the reverse pattern).

The dissimilarity matrix was used as input to a multidimensional scaling analysis (MDS) which represent the distance between years in a 2D space. The results are represented by an MDS plot and the time series of annual temperature anomalies.

2.1.2.2 Distribution

Data from the North Atlantic ICES Oceanographic database from 1977–2006 was used to evaluate inter-annual and seasonal variability in species distribution with respect to bottom temperature. Quarterly (months 1–3, 4–6, 7–9, 10–12) surfaces depicting bottom temperature

were created using potential mapping in SPANS (Geomatica, 2006). A surface was created for each of three periods of different thermal conditions (refer to the description of the analysis quarterly annual SST temperature anomalies below) (Figure 2.1.3). Each surface was created to cover the range of temperatures comprising a classification of 1 degree intervals from -2.0 to 13+^{O}C.

The temperature surfaces described above were overlaid with the survey point data. A bottom temperature category was appended to the survey data set and this was used to examine habitat (bottom temperature) association of the 22 species listed in Table. 2.1.1.

Point survey number per tow were converted to surfaces using potential mapping in SPANS to depict quarterly distributions for the 22 selected species for each of three periods of different thermal conditions described above. Numbers per tow was categorized into 14 equal areas (low- blue to brown - high) plus are areas surveyed with no catch (depicted by grey). Refer to Kulka and Pitcher (1999) for details of the method.

2.1.2.3 Quotient Plots

The quotient plot analysis is used to examine changes in fish distribution with respect to bottom temperature. The technique is described in the GLOBEC/SPACC Workshop on Characterizing and Comparing the Spawning Habitats of Small Pelagic Fish, Report, 12–13 January 2004, Concepción, Chile. van der Lingen, C.D., L. Castro, L., Drapeau and D. Checkley.

The method calculates the ratio of mean fish abundance for a given temperature range, over the mean fish abundance for all temperatures. A ratio of one signifies ‘no preference’ for a given temperature range. A ratio above 1 means preference and a ratio below one means avoidance of a given temperature range.

2.1.3 Results

2.1.3.1 Temperature

Three distinct periods of thermal conditions were identified from the MDS plot (Fig 2.1.1). The first period (blue) from 1960 to 1988 is located at the centre of the plot. The second period (1989–1998) is a period of high variability between years, indicating a period of change, values often outside the range of observations in the earlier period. The third period (1999–2005) shows much lesser variability and is separated from the first and second periods indicating a warmer period relative to earlier times.

Based on the analysis of SST, described above, survey data were partitioned into three periods (1977–1989, 1990–1999 and 2000–2005) and quarterly (months 1–3, 4–6, 7–9, 10–12) intervals corresponding to periods of different thermal conditions.

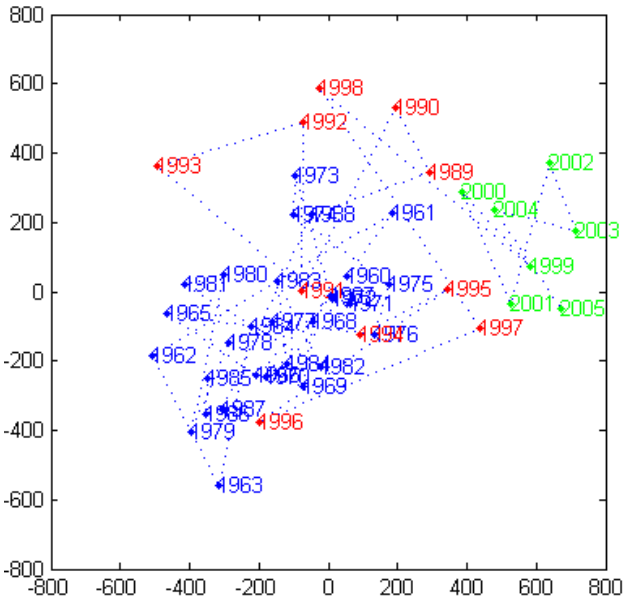


Figure 2.1.1. MDS plot calculated from the temperature averages, monthly anomalies and averaged by quarter.

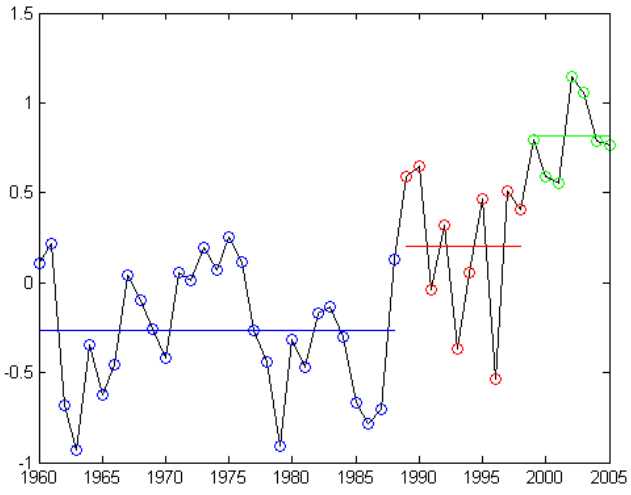


Figure 2.1.2. Annual temperature anomalies from a long term mean averaged by quarter.

2.1.3.2 Distribution

Change in distribution of the 22 species examined is described qualitatively in terms of area occupied and density.

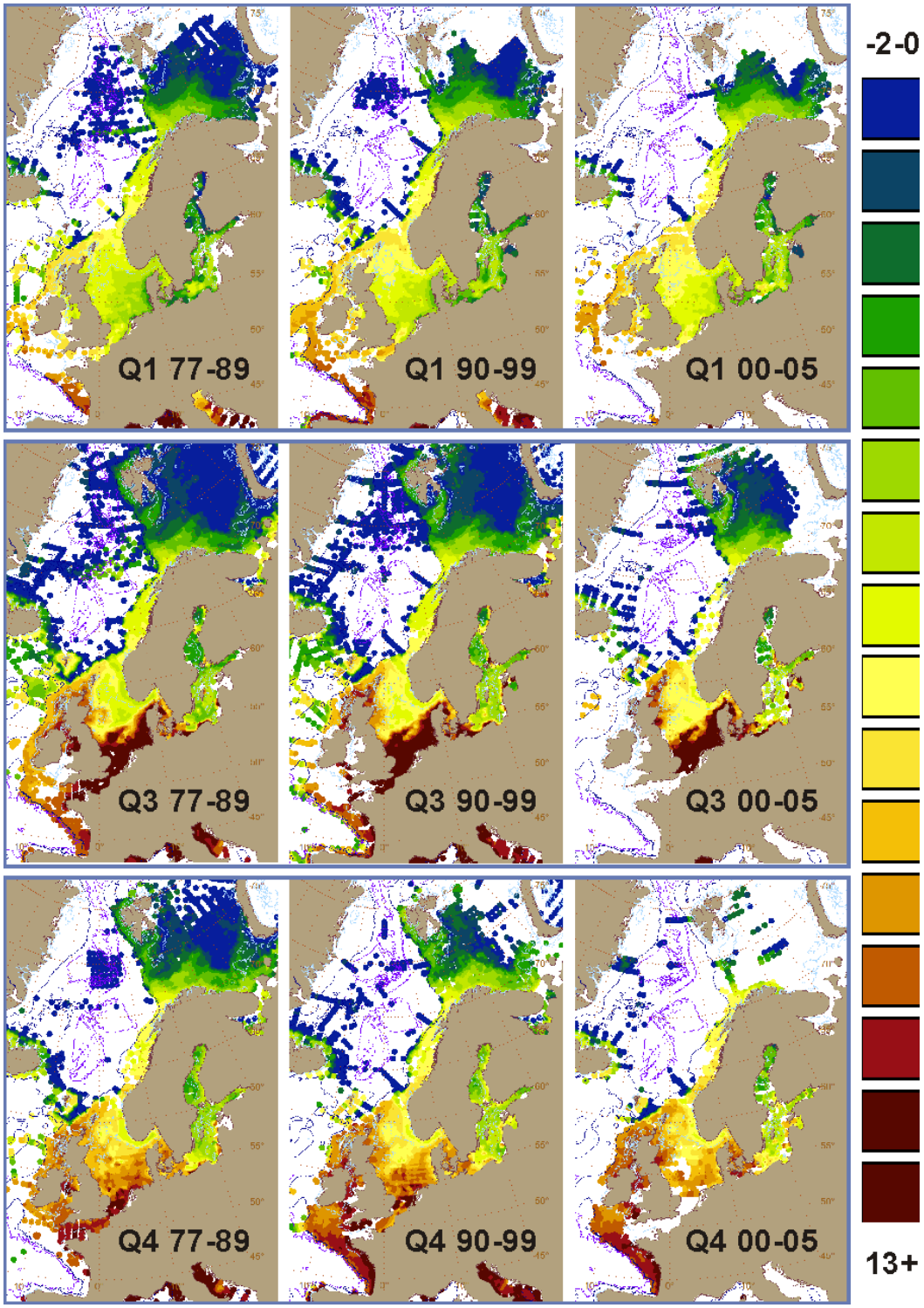


Figure 2.1.3. Quarterly bottom temperature surfaces corresponding to three time periods, 1977–1989, 1990–1999 and 2000–2005 corresponding to three periods of different thermal conditions. The scale, binned into 16 categories is consistent across quarters and periods ranging from < -2.0^{O} C to +13^{O}C by 1 degree intervals.

2.1.3.2.1 Pelagic Species

Seasonal variation in bottom temperature was lowest to the north in the Barents Sea where temperatures remained low, primarily <5^{O}C year round. Bottom temperatures in the northern 2/3rds of the Barents Sea rarely exceeded 2^{O}C (Figure 2.1.3). The degree of change among the three time periods examined was most discernable in the area of greatest seasonal change, in the North Sea, primarily the southern shallow sector. Particularly in quarter 1, an increase of about 2 degrees is observed there between 1997–1989 and 2000–2005 in the southeast, lower but still apparent in other areas. Temperatures in the Irish, Celtic and Cantabrian Seas showed relatively little variation, remaining at > 11^{O}C year round. Correspondingly, changes in distribution of species distribution over time, described below, were consistently most apparent in the North Sea, the area of greatest thermal variation.

The species distribution maps were examined qualitatively for change among 3 the periods of warming conditions. Of the 5 pelagic species examined, *Sprattus sprattus* (sprat) showed the least distributional change over time (Figure 2.1.4) Within the area surveyed, this species is only rarely observed north of the North Sea and is found only close to shore in the Cantabrian Sea. It reaches its highest density in the southern shallow portion of the North Sea. There, it became slightly more densely concentrated with time although there was no significant change in the area occupied.

Clupea harengus (herring) underwent greatest change in distribution in the Barents Sea (Figure 2.1.5). There, within the area surveyed, area occupied and degree of concentration decreased over time. Herring were largely absent there (in quarter 1) in 2000–2005.Ther was no change observed in the North Sea. Herring were not recorded in the Cantabrian Sea (quarter 4).

Trachurus trachurus (horse mackerel) was also restricted to the seas surrounding Great Britain and south into the Cantabrian Sea (Figure 2.1.6). Both extent and density increased considerably between 1977–1989 and 1990–1999 in the North Sea increasing from few low density records to moderate density covering the entire North Sea. There was little change observed in 2000–2005 in the North Sea but density increased in the Cantabrian Sea between the two latter periods. Seasonally, horse mackerel were dispersed in the North Sea in quarter 1 but concentrated in the southern North Sea in quarters 3 and 4.

Sardina pilchardus (sardine) within the area surveyed was concentrated more to the south with only occasional records in the Barents Sea and east Norway (Figure 2.1.7). With only occasional records recorded in the North Sea during 1997–1989, distribution increased to cover the western extent of the area in 1990–1999 and were highly concentrated in the south in 2000–2005 (quarter 1). In quarter 3, no sardine records were recorded in the first period but moderate concentrations were observed in the south after 1990. High density concentrations were more extensive in 2000–2005 in the Cantabrian Sea (quarter 4)

Similar to sardine, *Engraulis encrasicolus* (anchovy) distributed differently within seasons, being dispersed throughout most of the North Sea in quarter 1, more concentrated in the southern North Sea in quarters 3 and 4 (Figure 2.1.8). Anchovy were absent from the North Sea prior to 1989, appearing in low concentrations in 1990–1999 increasing in density in 2000–2005. Anchovy underwent as decline in density in the Cantabrian Sea in 2000–2005.

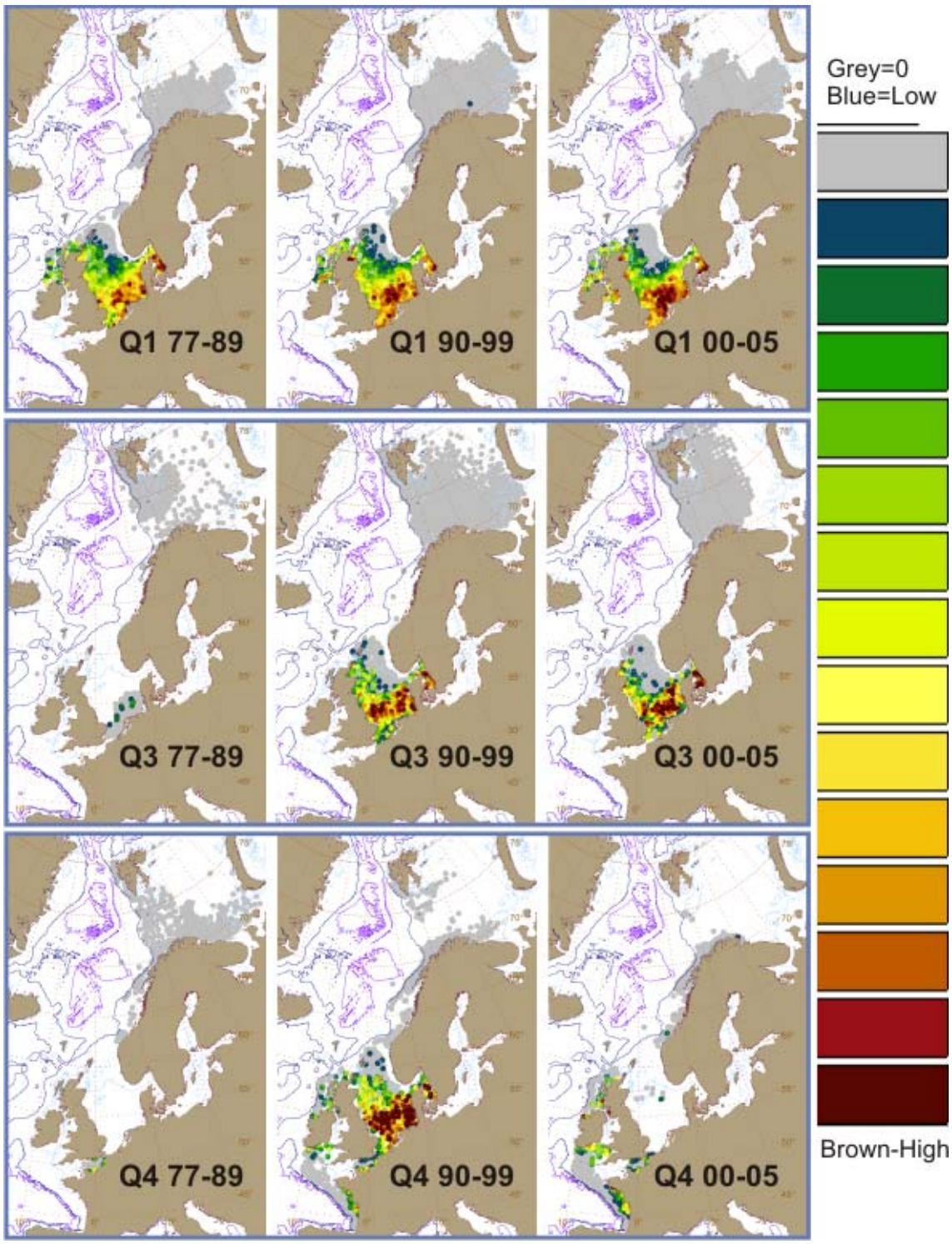


Figure 2.1.4. Distribution of pelagic species *Sprattus sprattus* (sprat) mapped for three periods of different thermal conditions for quarters 1, 3 and 4. Density classification is on a consistent scale across time periods and quarters to facilitate an evaluation of changes in distribution over time.

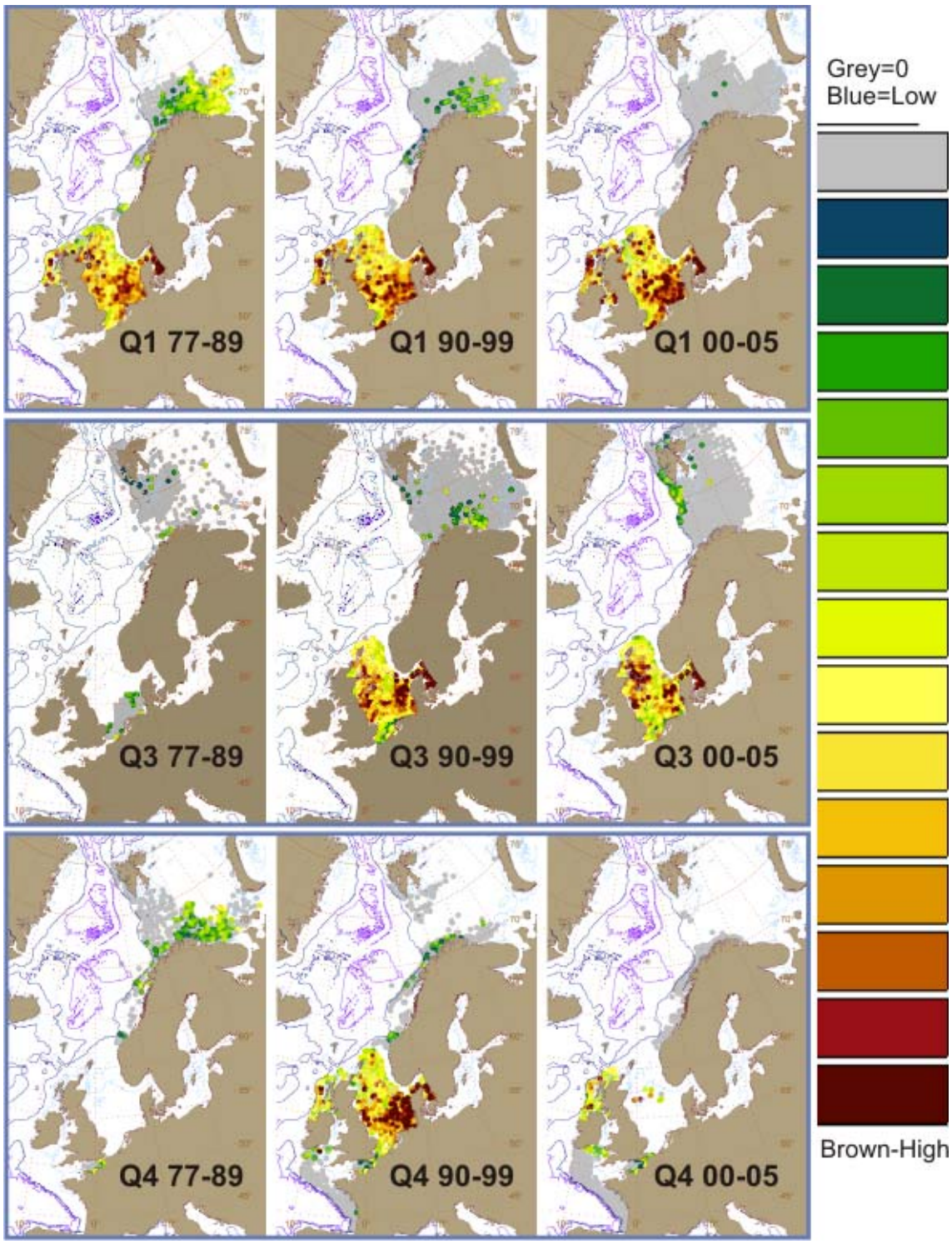


Figure 2.1.5. Distribution of pelagic species *Clupea harengus* (herring) mapped for three periods of different thermal conditions for quarters 1, 3 and 4. Density classification is on a consistent scale across time periods and quarters to facilitate an evaluation of changes in distribution over time.

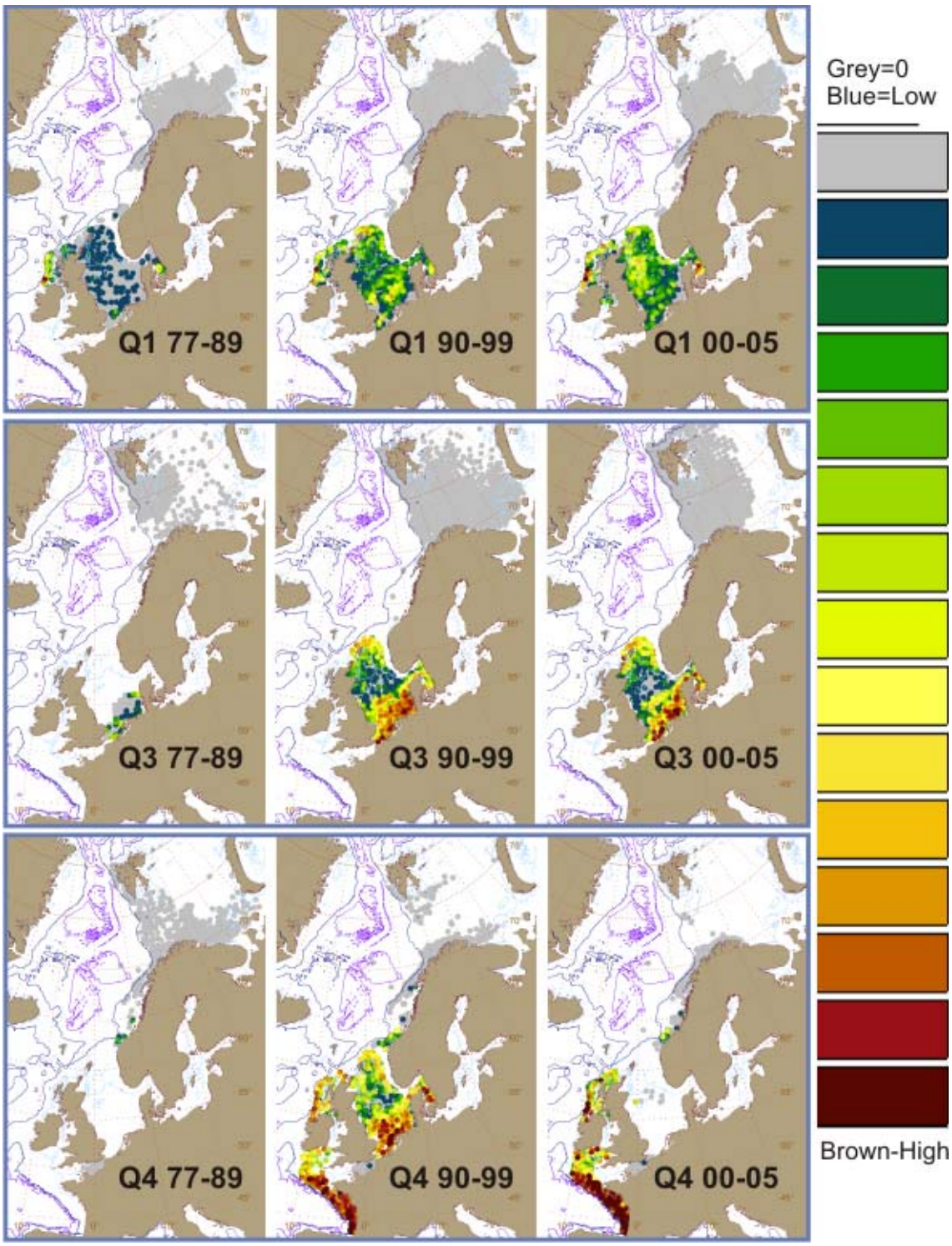


Figure 2.1.6. Distribution of *Trachurus trachurus* (horse mackerel) mapped for three periods of different thermal conditions for quarters 1, 3 and 4. Density classification is on a consistent scale across time periods and quarters to facilitate an evaluation of changes in distribution over time.

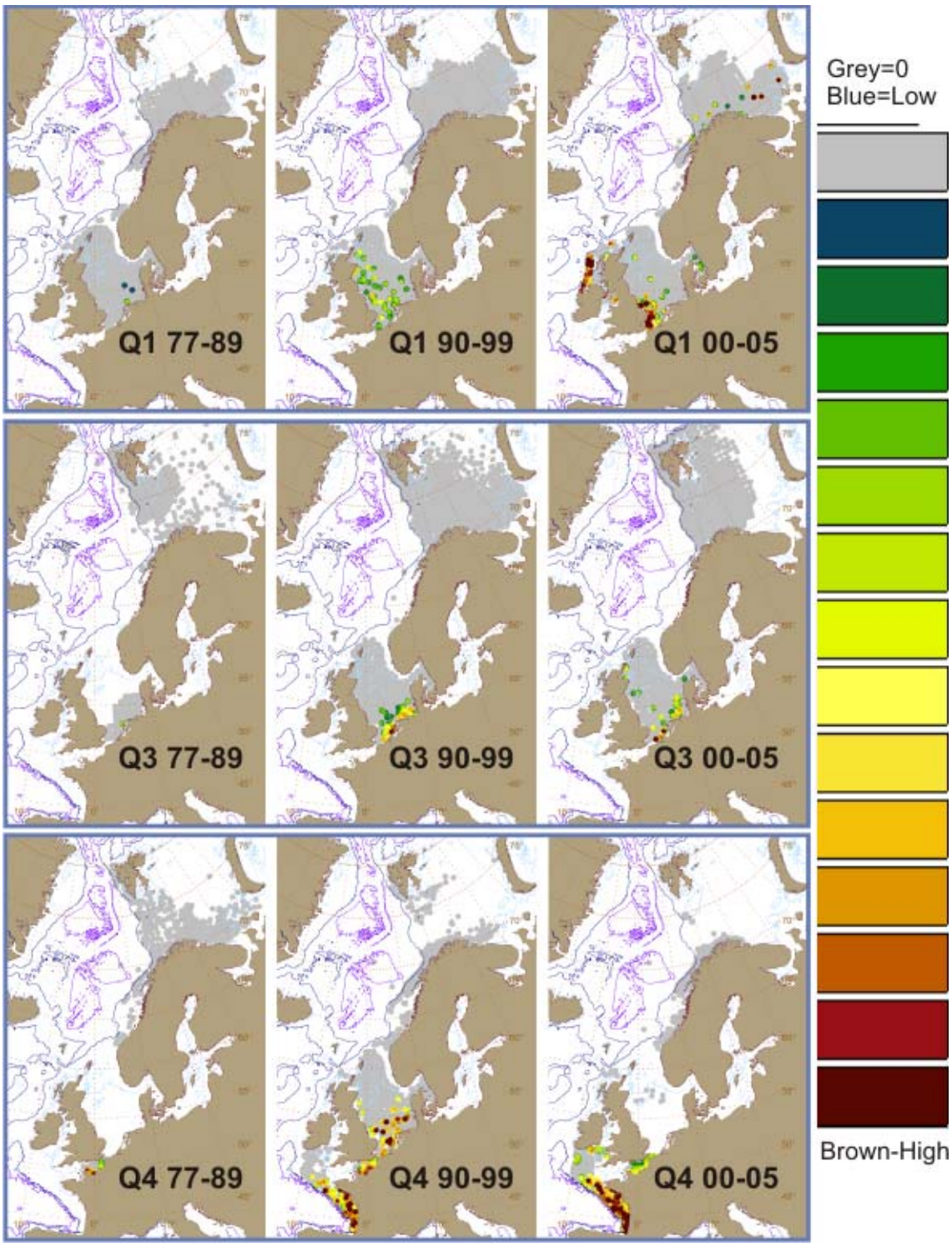


Figure 2.1.7. Distribution of *Sardina pilchardus* (sardine) mapped for three periods of different thermal conditions for quarters 1, 3 and 4. Density classification is on a consistent sale across time periods and quarters to facilitate an evaluation of changes in distribution over time.

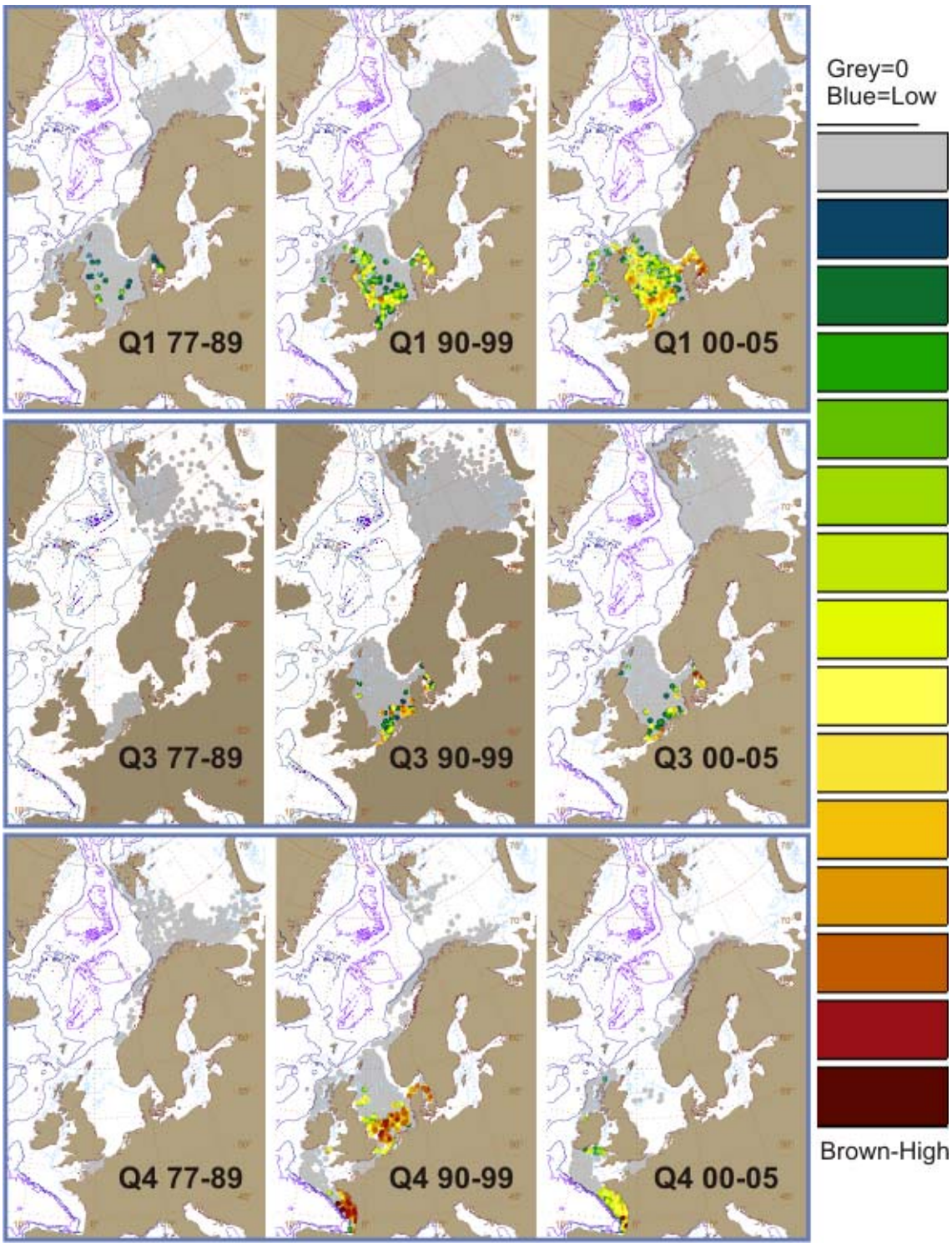


Figure 2.1.8. Distribution of pelagic species *Engraulis encrasicolus* (anchovy), mapped for three periods of different thermal conditions for quarters 1, 3 and 4. Density classification is on a consistent scale across time periods and quarters to facilitate an evaluation of changes in distribution over time.

2.1.3.2.2 Demersal Species

Six of 17 demersal species, namely *Capros asper* (boarfish), *Pleuronectes platessa* (American plaice), *Melanogrammus aeglefinus* (haddock), *Merlangius merlangus* (whiting), *Solea solea* (sole), *Pollachius virens* (saithe) underwent little or no change in their distribution (Figures 2.1.9–2.1.24). Saithe, plaice, whiting and haddock were consistently distributed in the northern half of the North Sea and along the Norwegian coast. Plaice, whiting and haddock also occurred from the Irish to Cantabrian Seas. Sole was consistently distributed in the shallow portion of the North Sea and along the western British and French Coasts. Boarfish were restricted mainly to the outer Cantabrian, Celtic and Irish Seas, with occasional records in the northern North Sea.

Six species, *Merluccius merluccius* (hake), *Amblyraja radiata* (starry ray), *Mullus surmuletus* (striped red mullet), *Lophius piscatorius* (anglerfish), *Zeus faber* (John Dory) *Scyliorhinus canicula* (lesser spotted dogfish) underwent an increase in distribution over the period examined (Figures 2.1.15–2.1.20). Hake maintained a similar area occupied but underwent an increase in density in the Cantabrian, Celtic, Irish and particularly in the northern extent of the North Sea. Starry ray increased in density in the North and Barents Sea at the core of its distribution. In quarter 1, mullet, absent in 1977–1989 in the North Sea became increasingly more densely concentrated in the western part of that area in the latter time periods. In quarter 3 and 4, it was concentrated mainly in the southern North Sea and the along the coast of France but at much lower concentrations in 1977–1989 compared to later periods. Monkfish progressively increased in density in the northern half of the North Sea as well as the Cantabrian and Celtic Seas. It may also have increased in density along the Norway coast. John Dory, distributed mainly from the Cantabrian to Irish Sea, not only increased in density in these areas over time but spread in the North Sea. Similarly, lesser spotted dogfish increased in density in the Contrarian to Irish Seas and the western extent of the North Sea. It al increased its area occupied in the North Sea.

Four species, *Trisopterus luscus* (bib), *Squalus acanthias* (spurdog), *Gadus morhua* (Atlantic cod), *Raja clavata* (thornback ray) underwent a decrease in distribution (Figures 2.1.21–2.1.24). Bib decreased both in terms of density and area occupied in the North Sea, higher concentrations confined to the southwest in 2000–2005. Changes were less apparent in the English Channel to the French coast. Cod, a widely spread species, occupying all areas except the Cantabrian Sea underwent a decline in density in all areas particularly in the North Sea. Thornback ray, quite highly concentrated in the Barents Sea in 1977–1989 was only recorded sporadically in the latter two periods. In the North Sea, although fairly widely distributed in 1977–1989 became increasingly confined to the southwest in the English Channel and the area immediately to the north. Fairly high concentrations were also observed in the Irish and Cantabrian Seas.

Helicolenus dactylopterus (bluemouth redfish) was not recorded in 1977–1989 in the North Sea but dramatically increase its range between in 1990–1999 in the northern extent. Subsequently it underwent a decrease. It also underwent an increase in density in the Cantabrian Sea between 1990–1999 and 2000–2005.

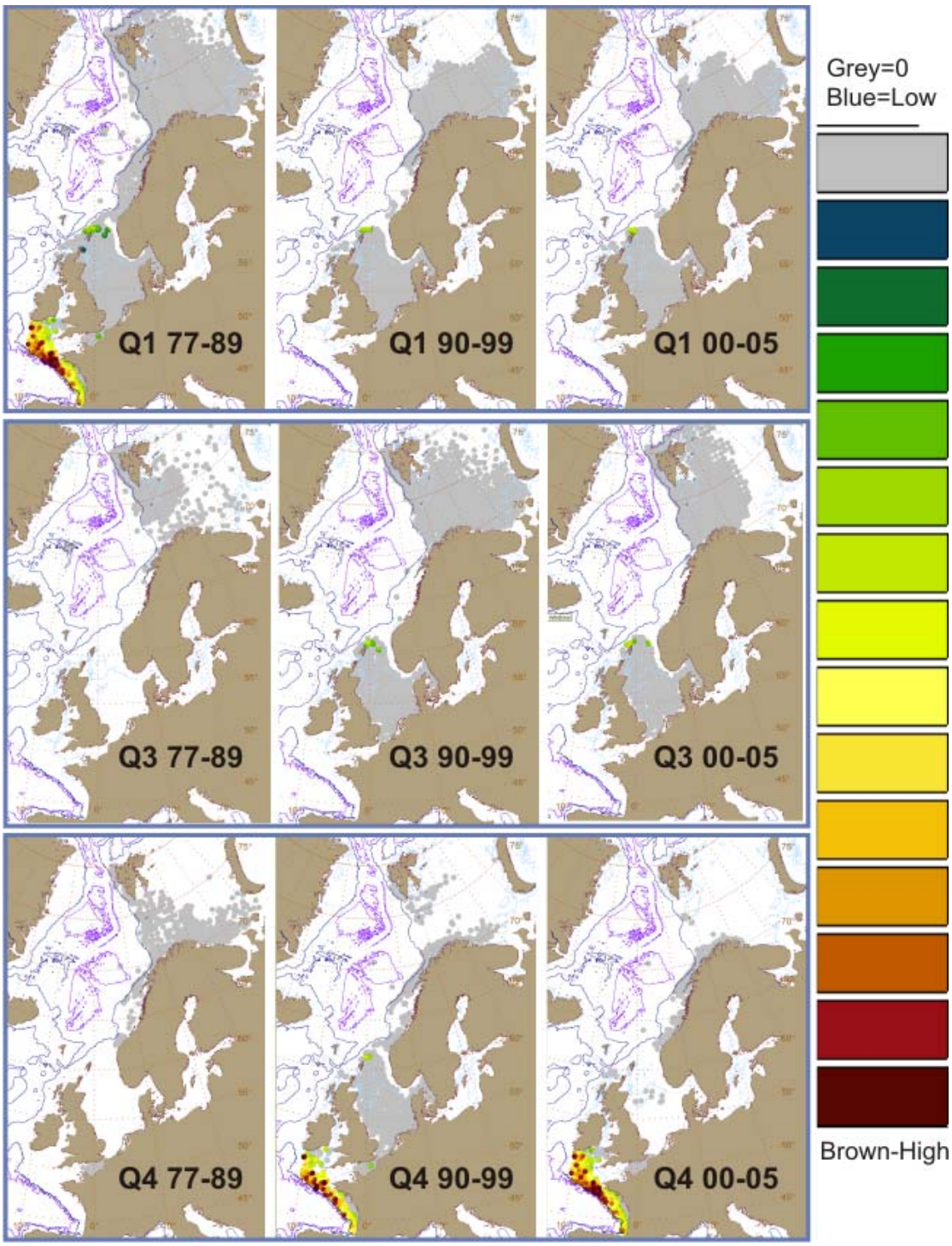


Figure 2.1.9. Distribution of *Capros asper* (boarfish) mapped for three periods of different thermal conditions for quarters 1, 3 and 4. Density classification is on a consistent scale across time periods and quarters to facilitate an evaluation of changes in distribution over time.

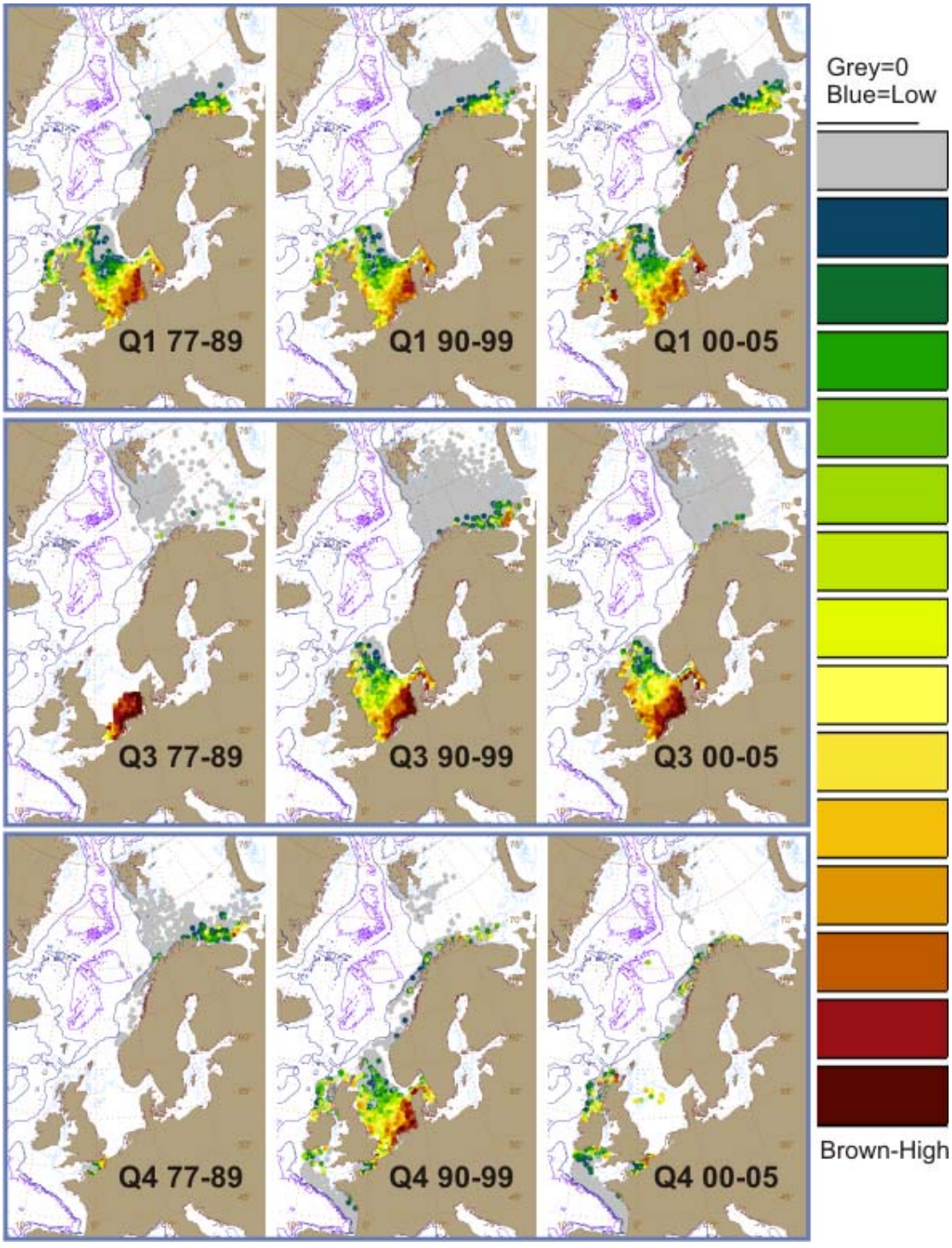


Figure 2.1.10. Distribution of *Pleuronectes platessa* (American plaice) mapped for three periods of different thermal conditions for quarters 1, 3 and 4. Density classification is on a consistent scale across time periods and quarters to facilitate an evaluation of changes in distribution over time.

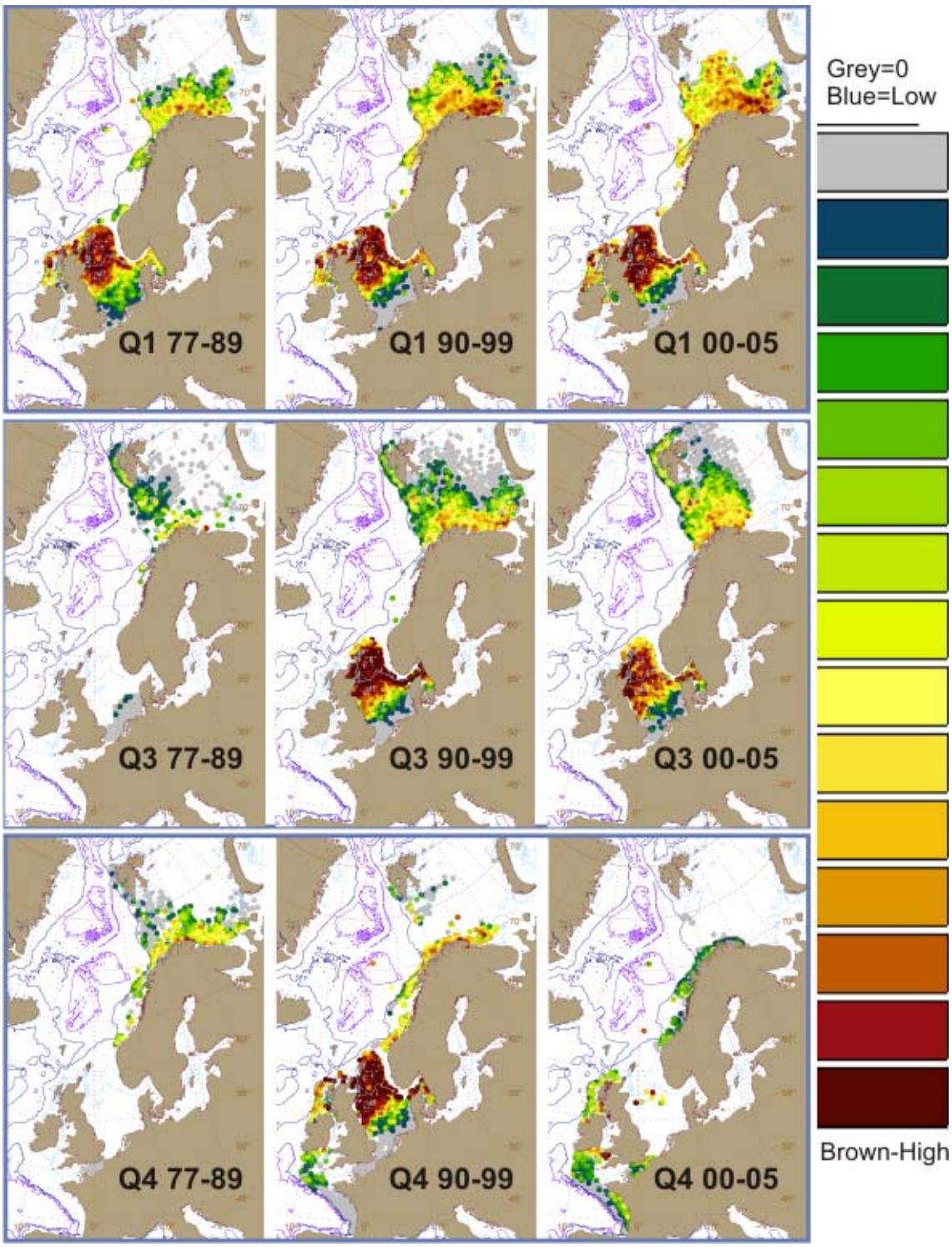


Figure 2.1.11. Distribution of *Melanogrammus aeglefinus* (haddock) mapped for three periods of different thermal conditions for quarters 1, 3 and 4. Density classification is on a consistent scale across time periods and quarters to facilitate an evaluation of changes in distribution over time.

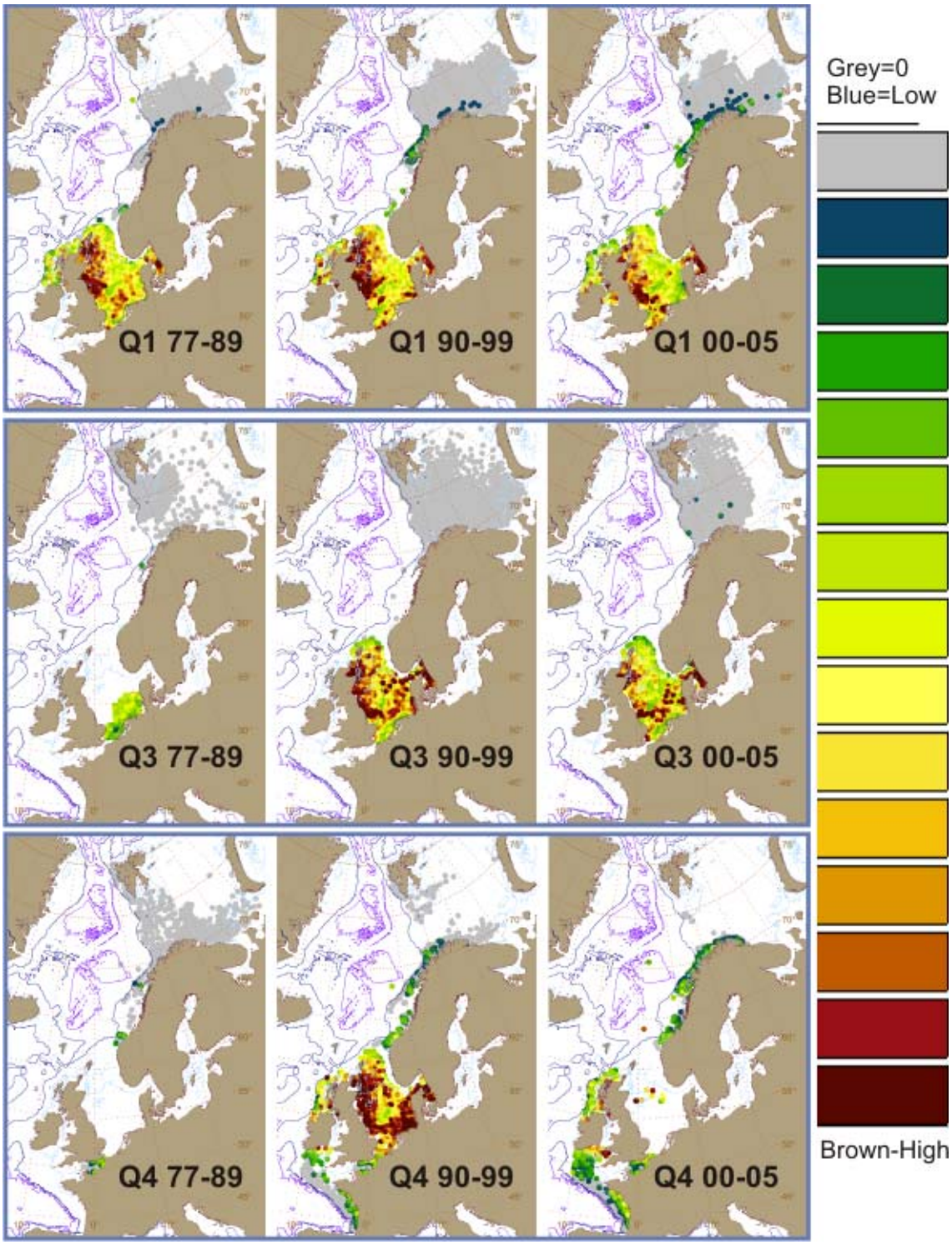


Figure 2.1.12. Distribution of *Merlangius merlangus* (whiting) mapped for three periods of different thermal conditions for quarters 1, 3 and 4. Density classification is on a consistent scale across time periods and quarters to facilitate an evaluation of changes in distribution over time.

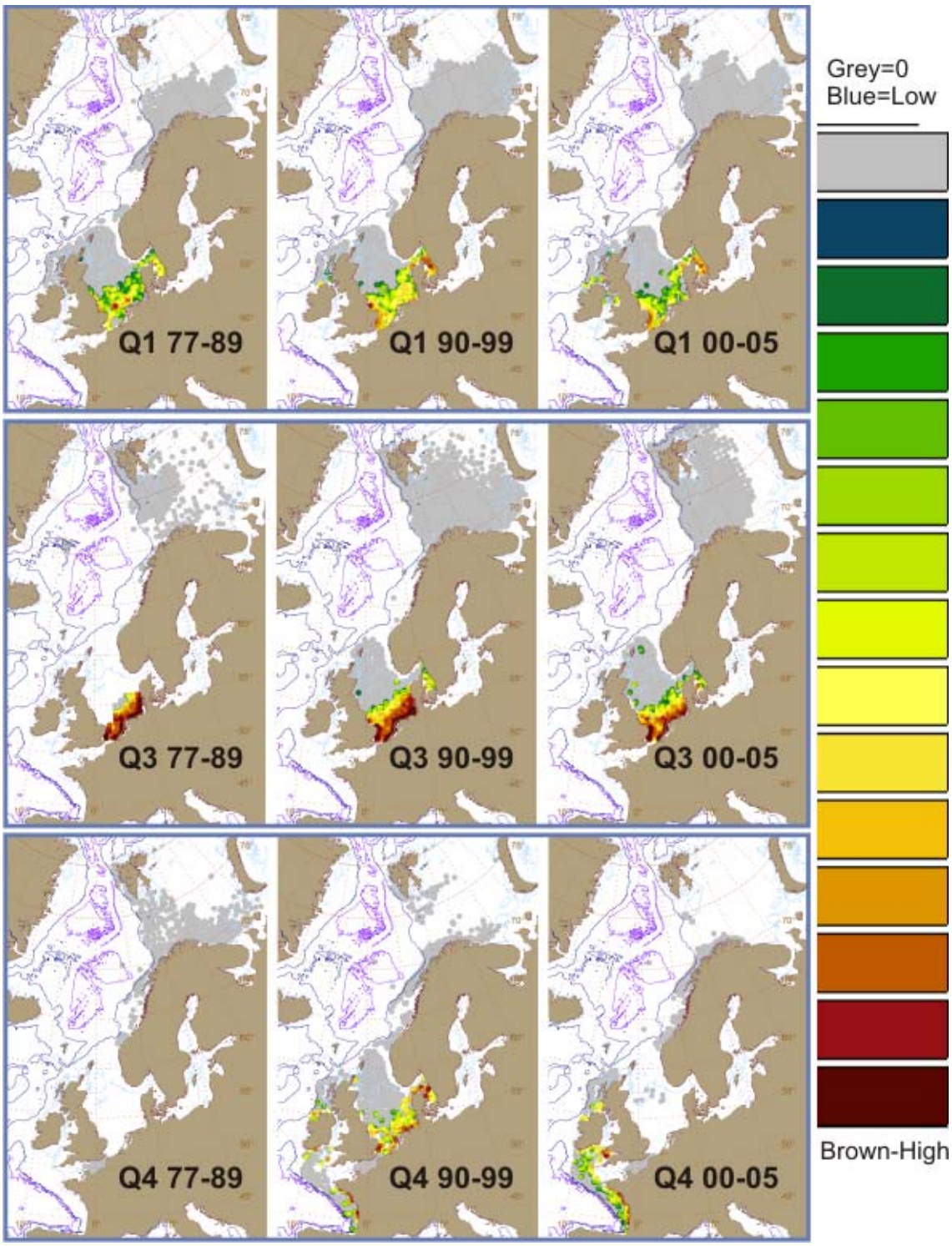


Figure 2.1.13. Distribution of *Solea solea* (sole) mapped for three periods of different thermal conditions for quarters 1, 3 and 4. Density classification is on a consistent scale across time periods and quarters to facilitate an evaluation of changes in distribution over time.

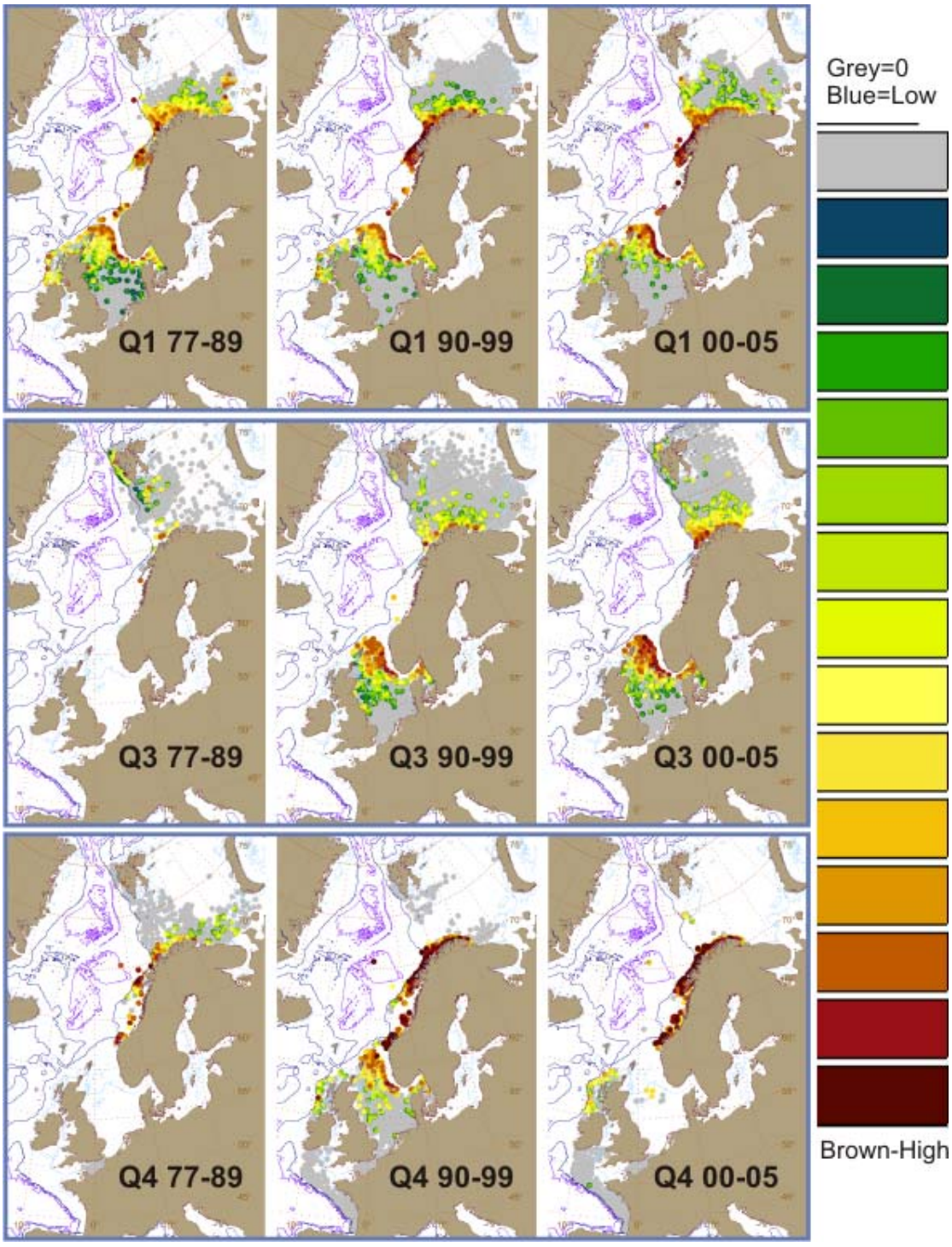


Figure 2.1.14. Distribution of *Pollachias virens* (saithe) mapped for three periods of different thermal conditions for quarters 1, 3 and 4. Density classification is on a consistent scale across time periods and quarters to facilitate an evaluation of changes in distribution over time.

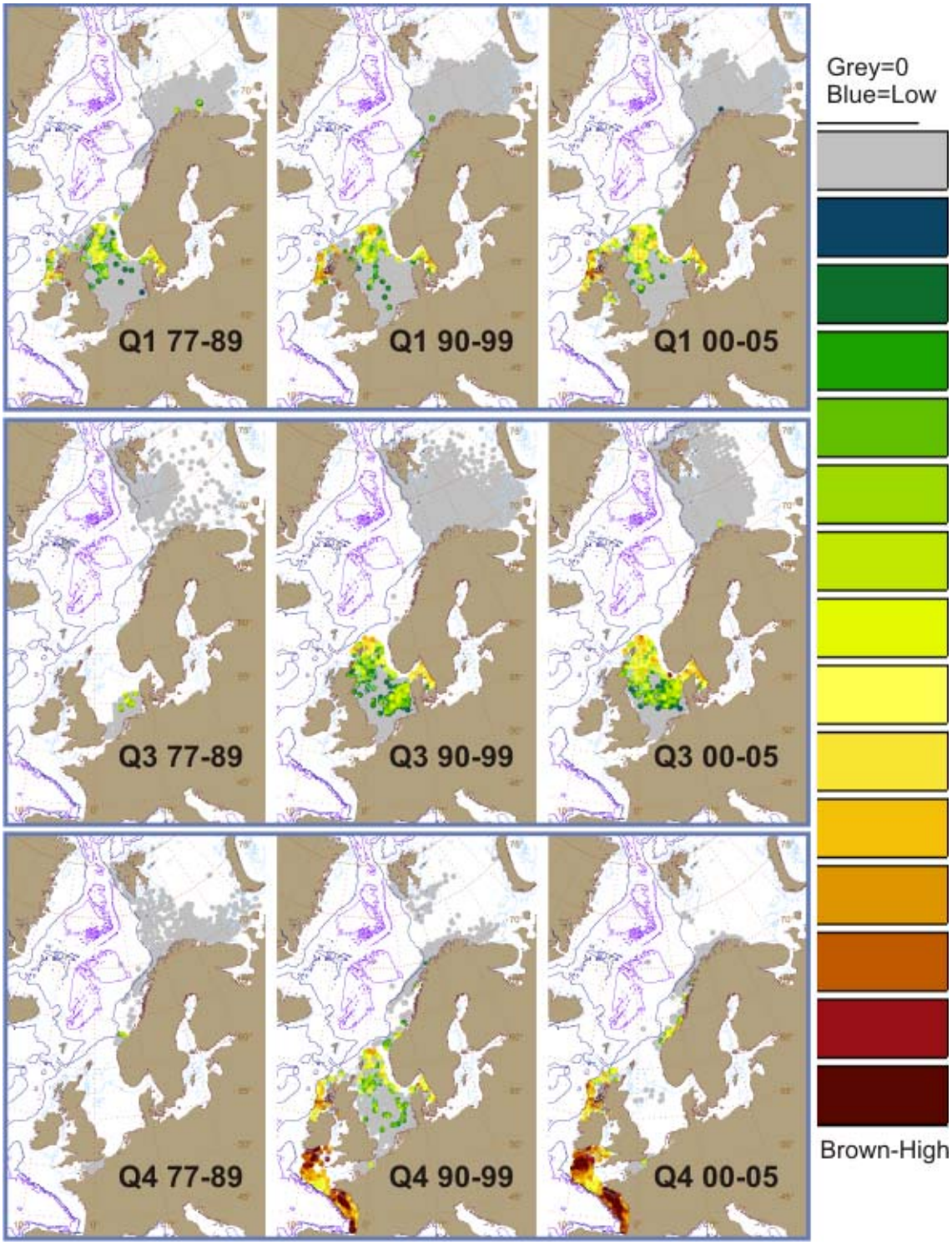


Figure 2.1.15. Distribution of *Merluccius merluccius* (hake) mapped for three periods of different thermal conditions for quarters 1, 3 and 4. Density classification is on a consistent scale across time periods and quarters to facilitate an evaluation of changes in distribution over time.

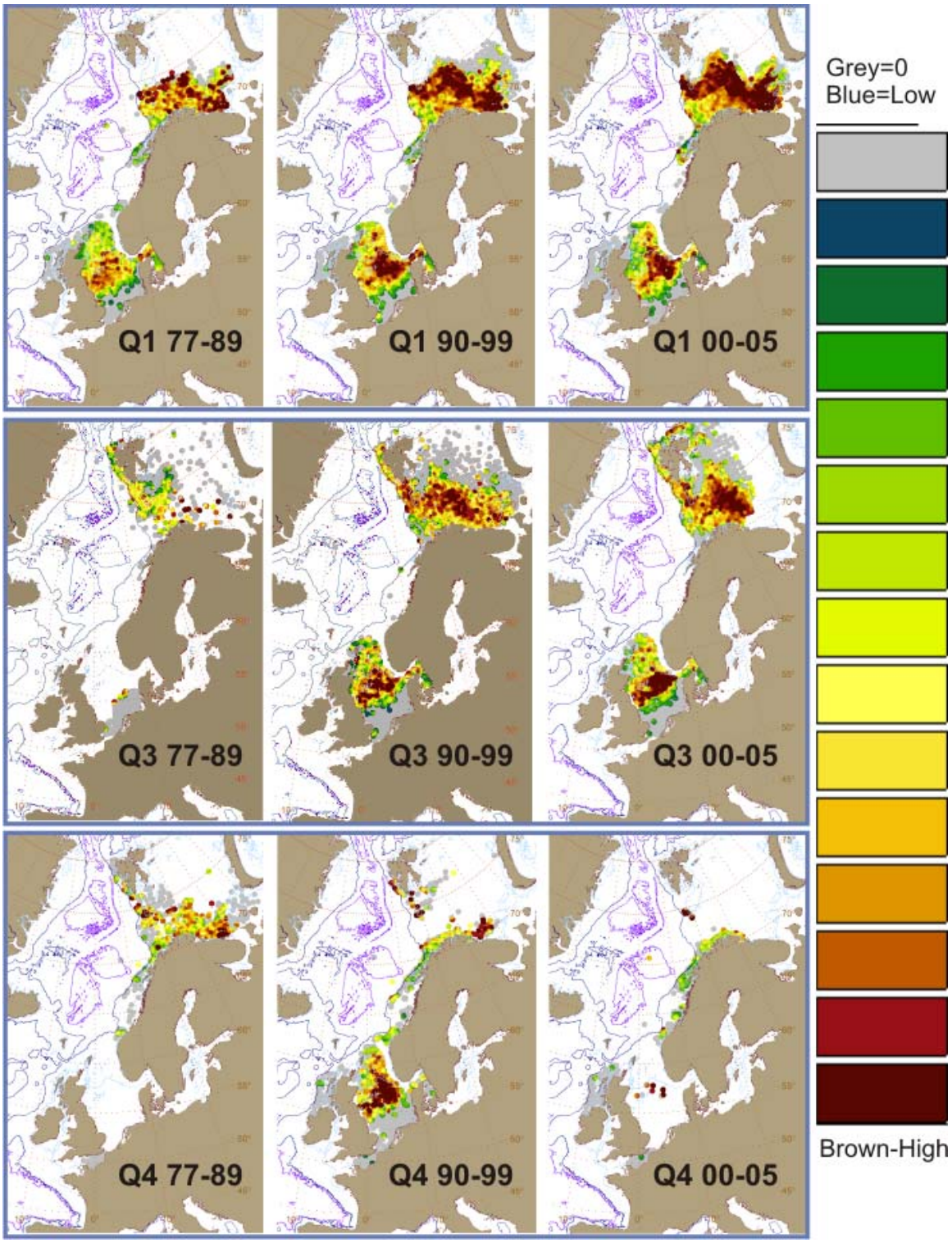


Figure 2.1.16. Distribution of *Amblyraja radiata* (starry ray) mapped for three periods of different thermal conditions for quarters 1, 3 and 4. Density classification is on a consistent scale across time periods and quarters to facilitate an evaluation of changes in distribution over time.

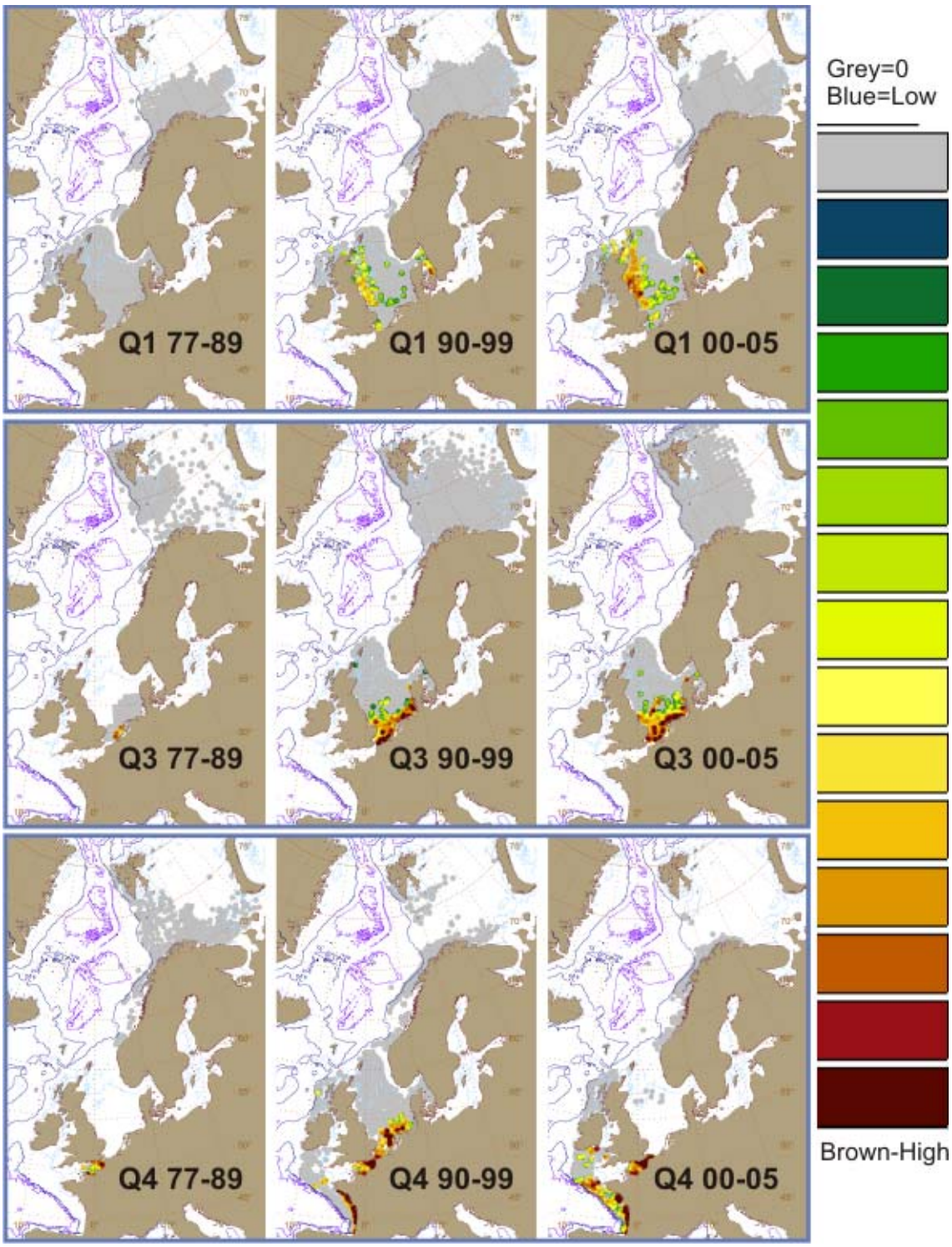


Figure 2.1.17. Distribution of *Mullus surmullus* (striped red mullet) mapped for three periods of different thermal conditions for quarters 1, 3 and 4. Density classification is on a consistent scale across time periods and quarters to facilitate an evaluation of changes in distribution over time.

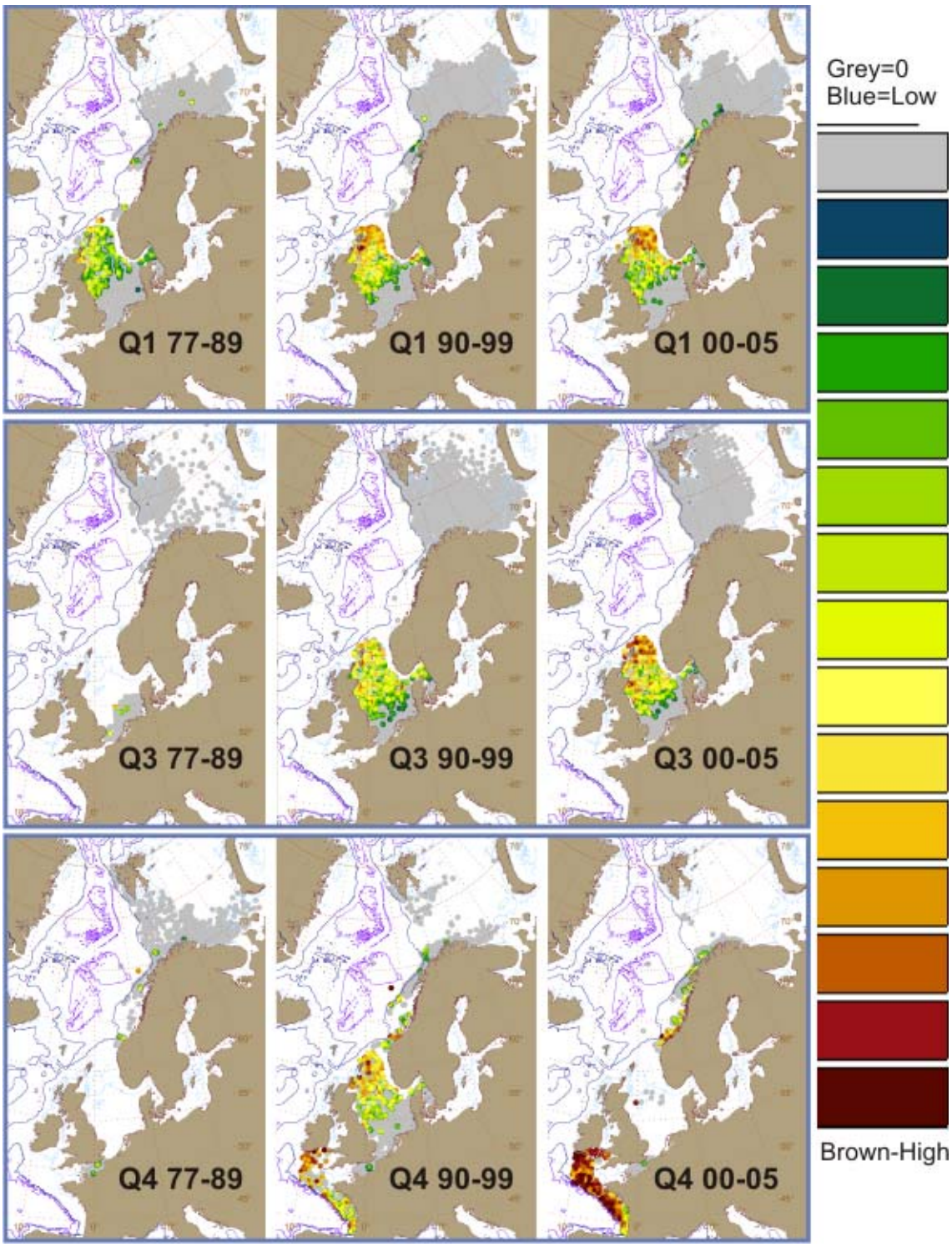


Figure 2.1.18. Distribution of *Lophius piscatorius* (monkfish) mapped for three periods of different thermal conditions for quarters 1, 3 and 4. Density classification is on a consistent scale across time periods and quarters to facilitate an evaluation of changes in distribution over time.

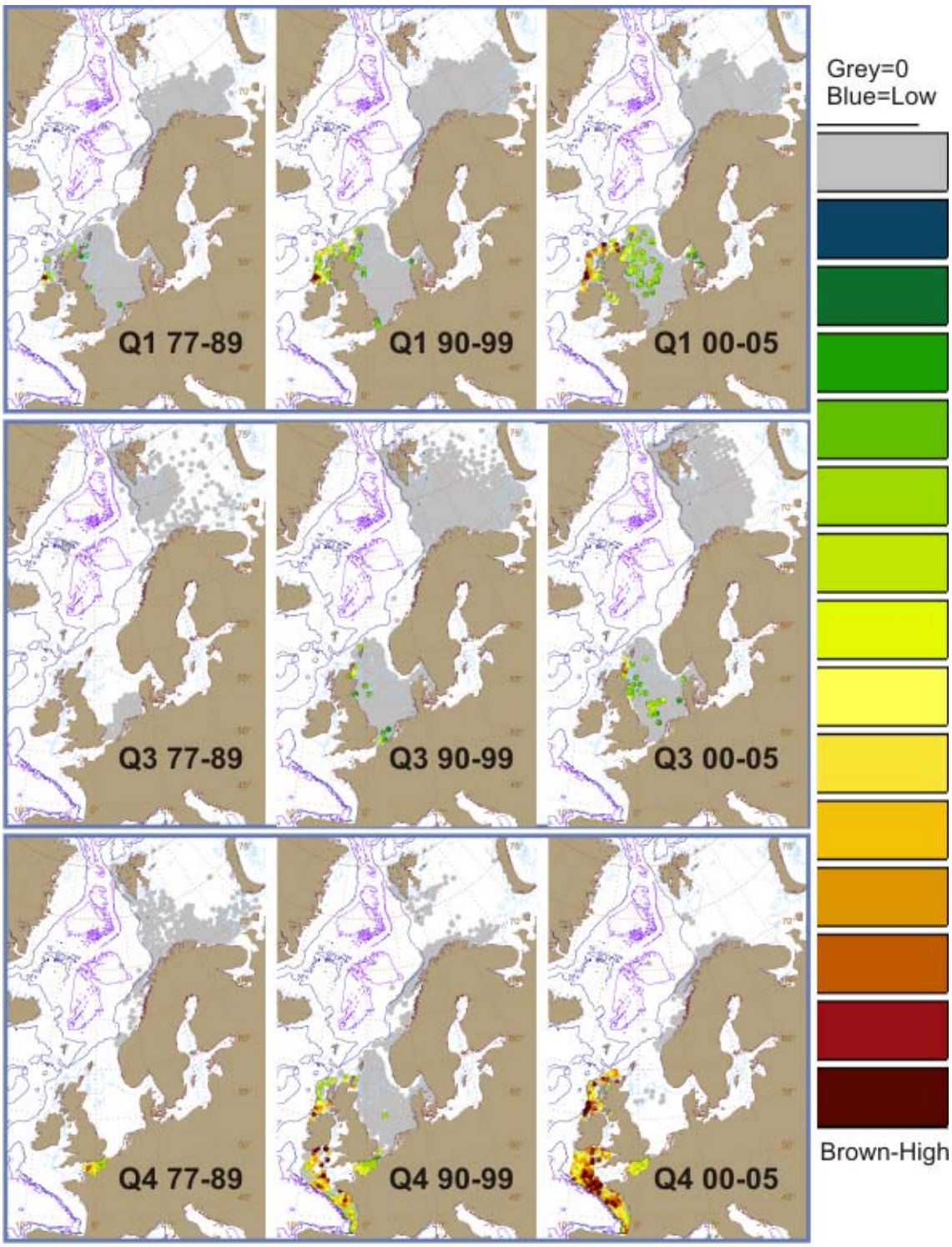


Figure 2.1.19. Distribution of *Zeus faber* (John Dory) mapped for three periods of different thermal conditions for quarters 1, 3 and 4. Density classification is on a consistent scale across time periods and quarters to facilitate an evaluation of changes in distribution over time.

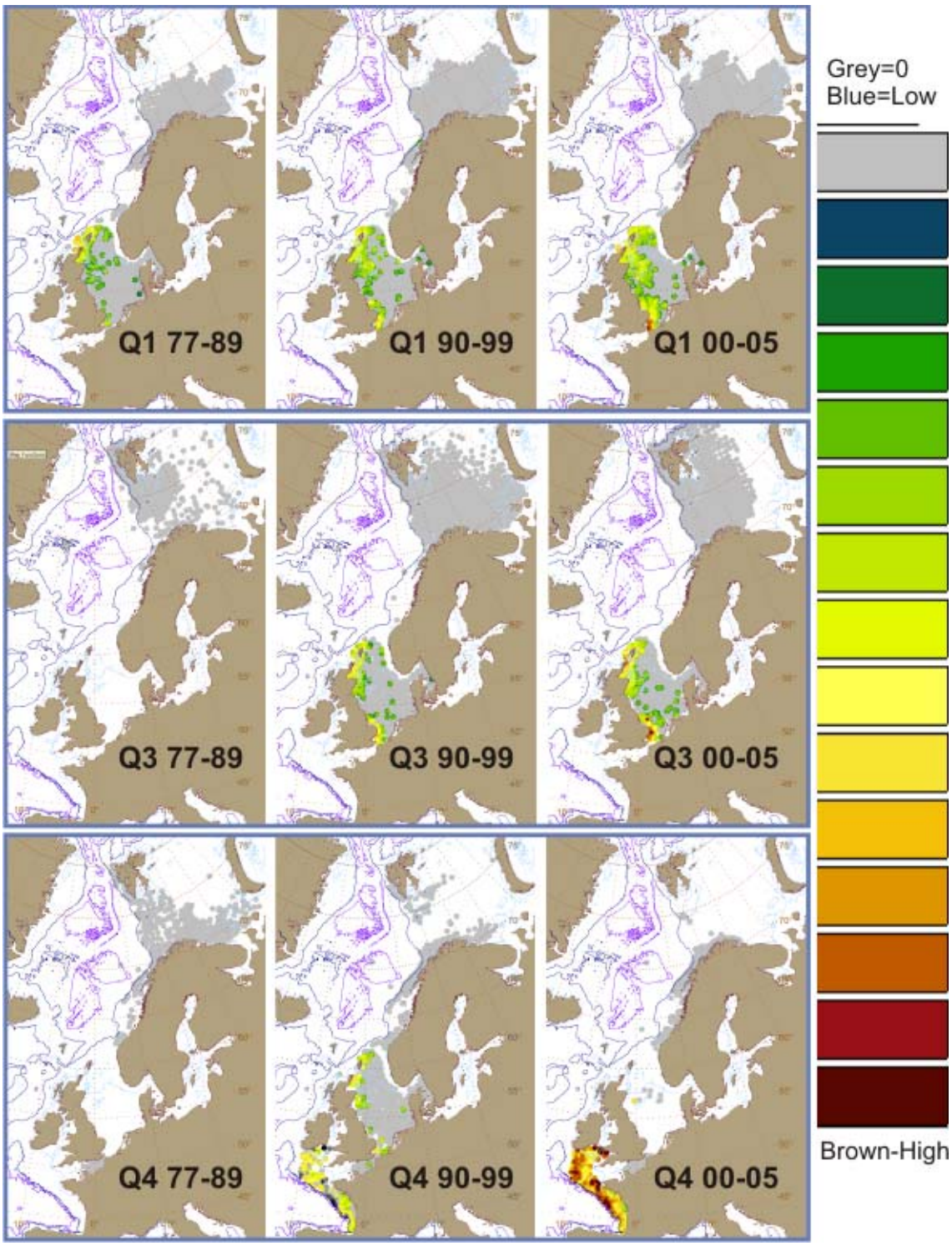


Figure 2.1.20. Distribution of *Scyliorhinus canicula* (lesser spotted dogfish) mapped for three periods of different thermal conditions for quarters 1, 3 and 4. Density classification is on a consistent scale across time periods and quarters to facilitate an evaluation of changes in distribution over time.

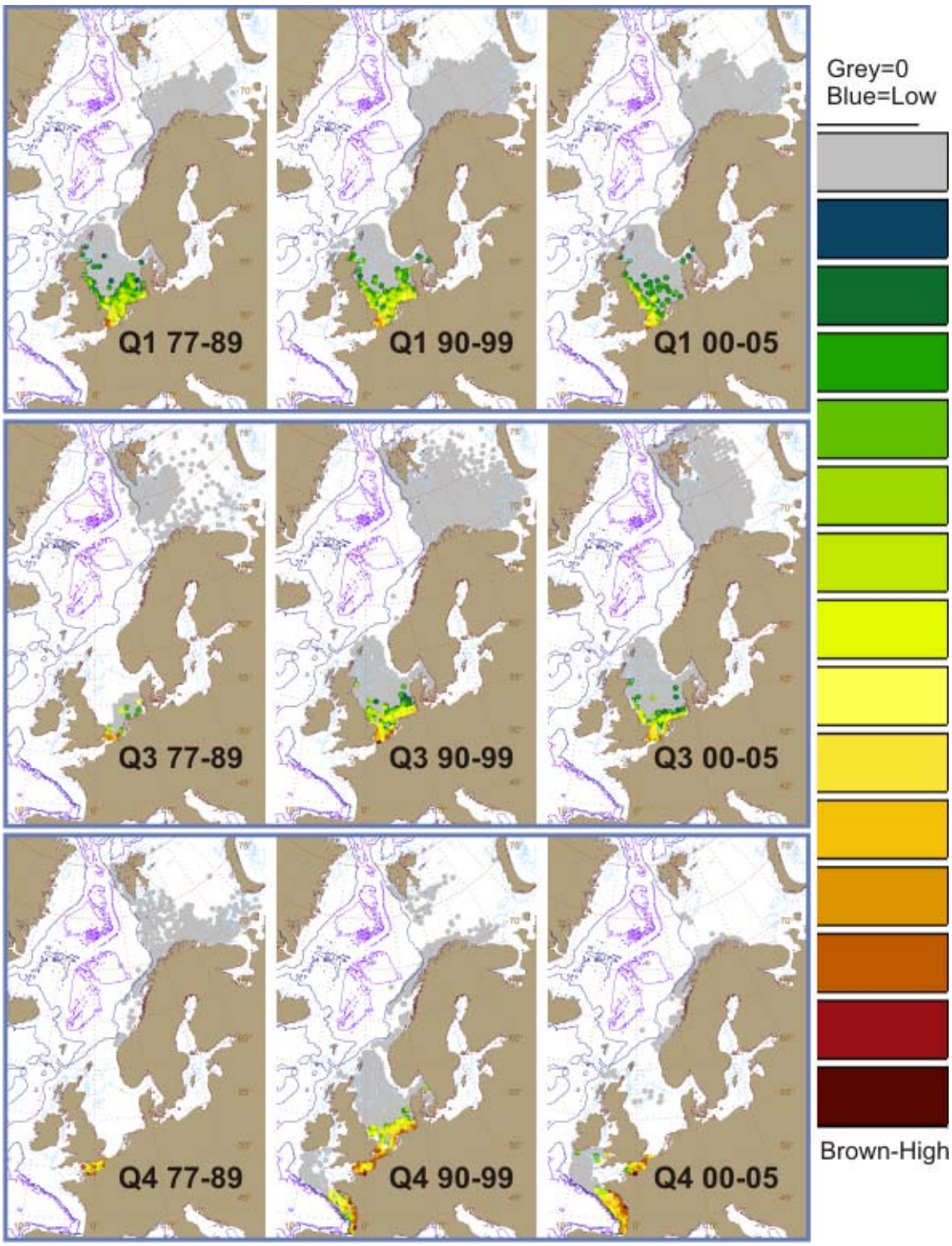


Figure 2.1.21. Distribution of *Trisopterus luscus* (bib) mapped for three periods of different thermal conditions for quarters 1, 3 and 4. Density classification is on a consistent scale across time periods and quarters to facilitate an evaluation of changes in distribution over time.

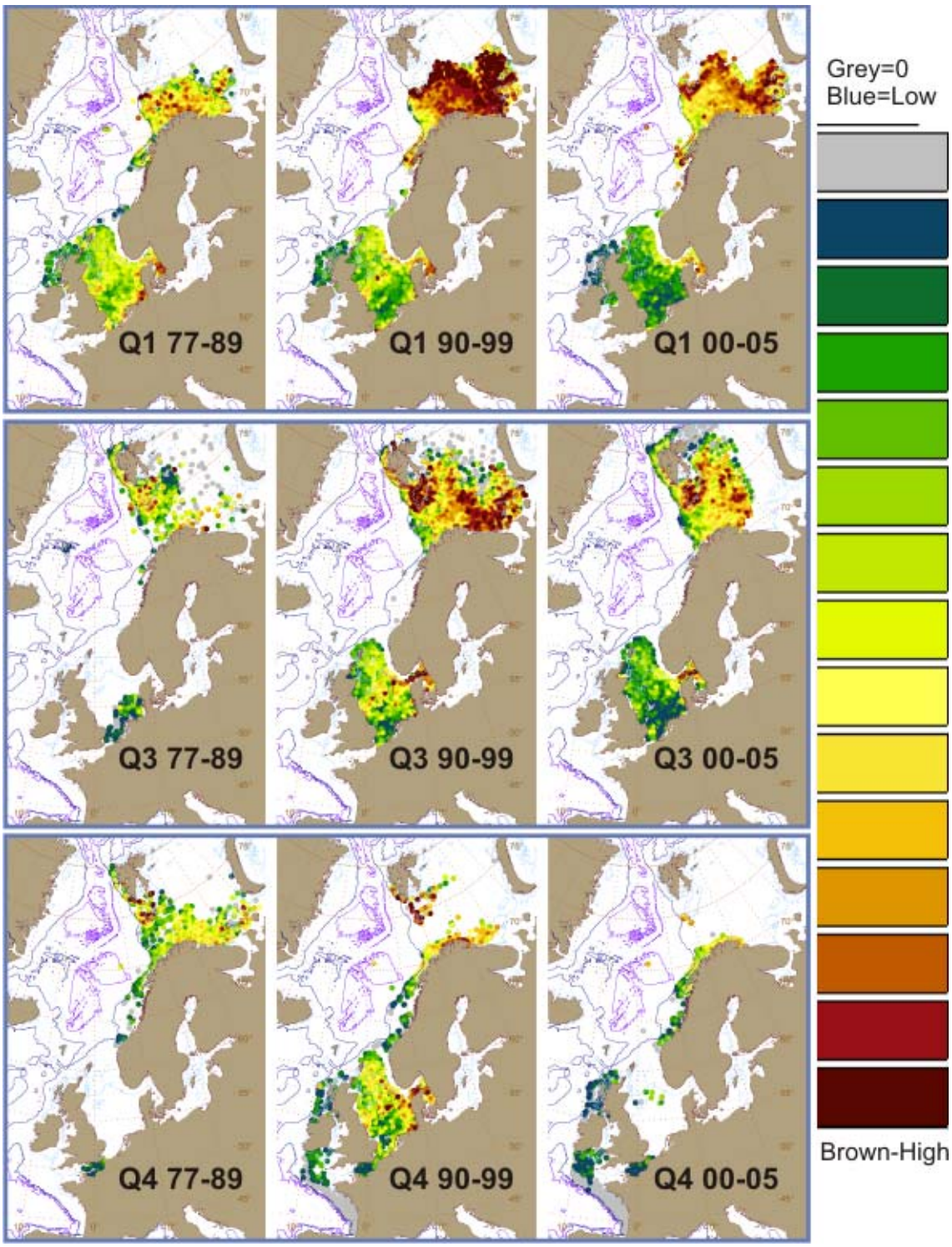


Figure 2.1.22. Distribution of *Gadus morhua* (Atlantic cod) mapped for three periods of different thermal conditions for quarters 1, 3 and 4. Density classification is on a consistent scale across time periods and quarters to facilitate an evaluation of changes in distribution over time.

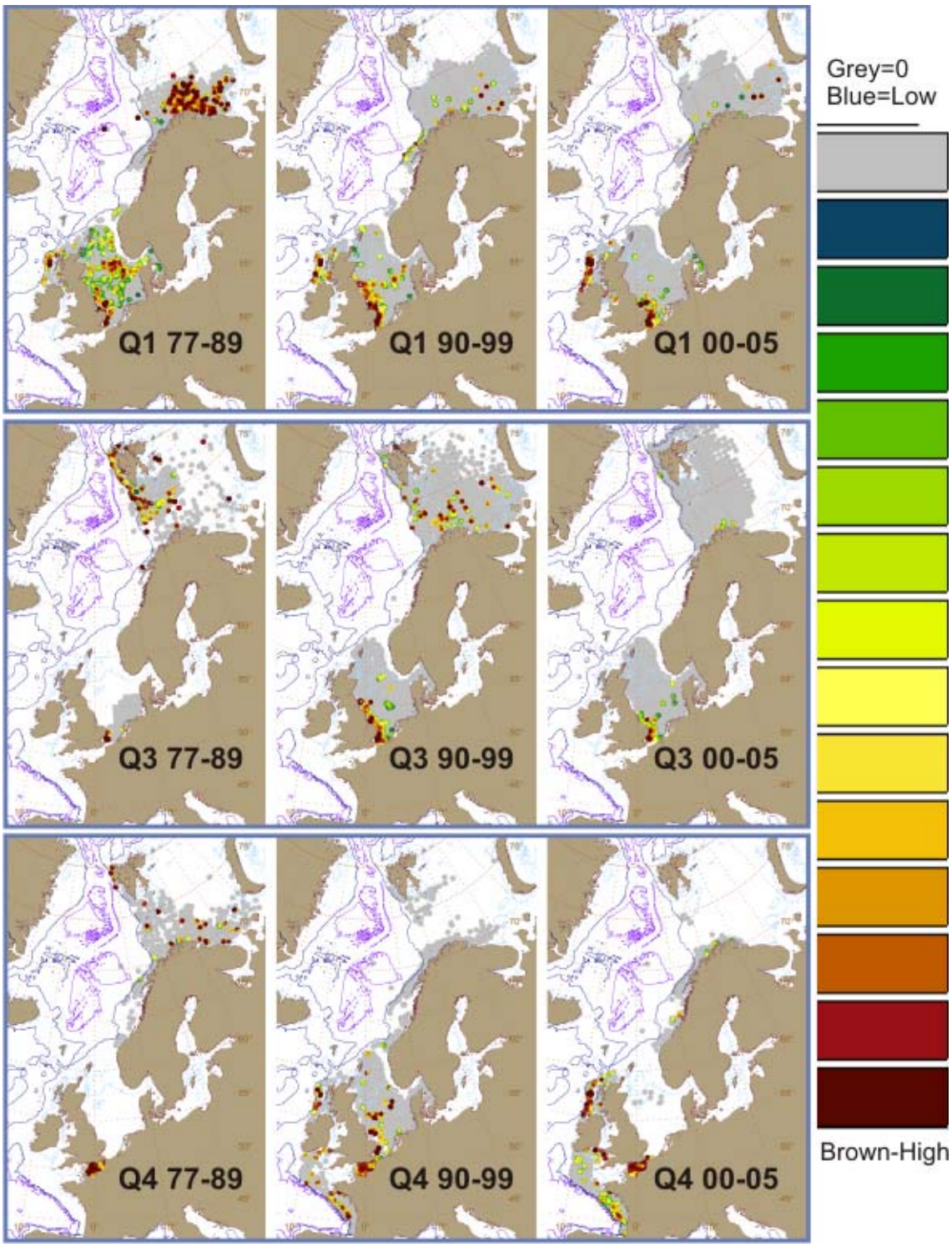


Figure 2.1.23. Distribution of *Raja clavata* (thornback ray) mapped for three periods of different thermal conditions for quarters 1, 3 and 4. Density classification is on a consistent scale across time periods and quarters to facilitate an evaluation of changes in distribution over time.

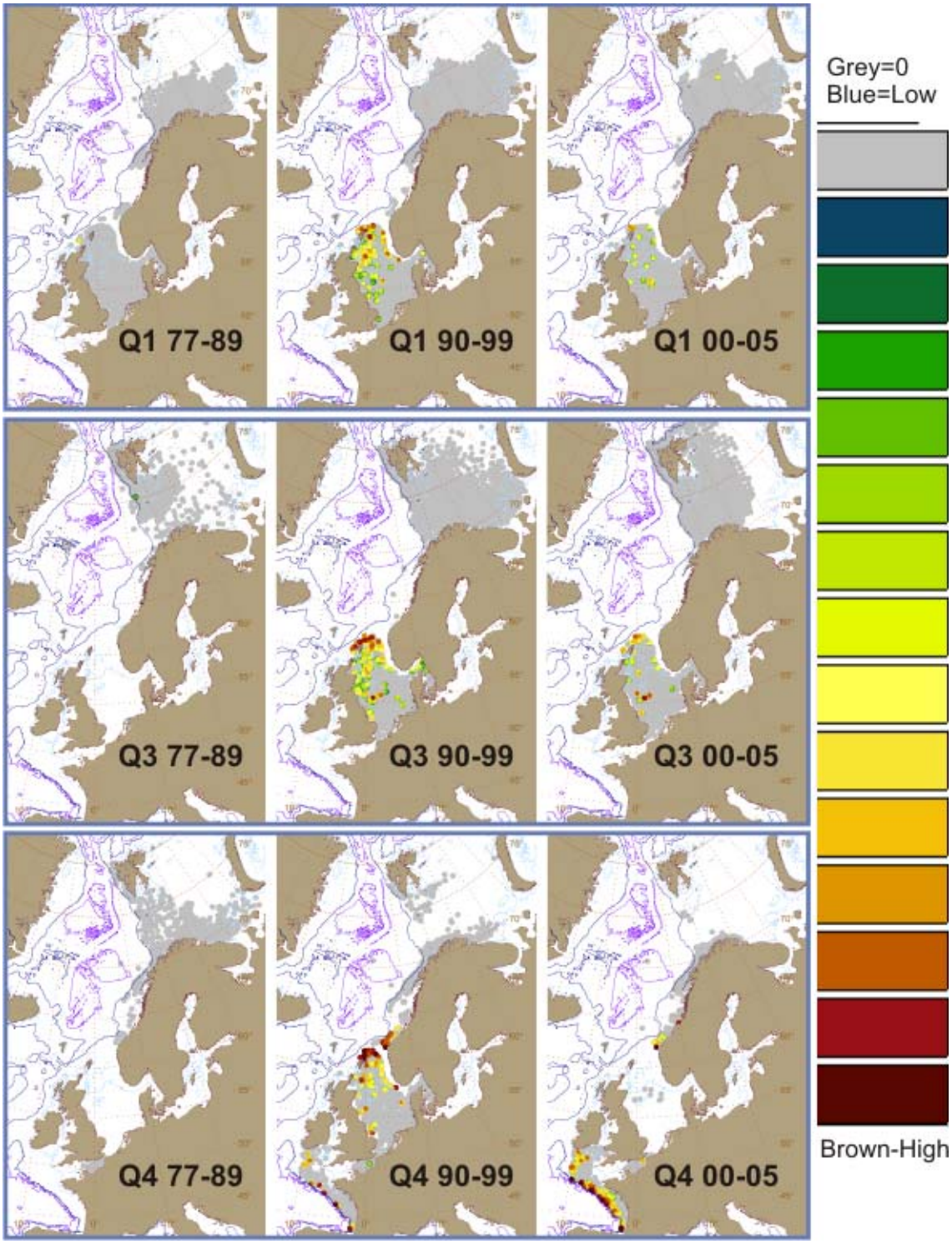


Figure 2.1.24. Distribution of *Helicolenus dactylopterus* (blue mouth redfish) mapped for three periods of different thermal conditions for quarters 1, 3 and 4. Density classification is on a consistent scale across time periods and quarters to facilitate an evaluation of changes in distribution over time.

2.1.3.3 Quotient plots

Quotient plots relating bottom temperature to distribution for the 22 species are presented by quarter (columns) and by time period (rows) for GOV (Cantabrian, Celtic, Irish and North Sea, GOV trawl) and CAM (Norwegian coast and Barents Sea, Campelen shrimp trawl) samplers.

Each graph shows the temperature preferendum for a given species in a given period and for a given quarter. The x-axis is the temperature range of the analysis and the y-axis is the quotient value (1=no preference, >1=preference, <1=avoidance). The horizontal red line marks the quotient value of one. The plain line corresponds to the analysis performed for GOV sampler and dotted line for CAM sampler.

The quotient plot figures summarise the combination between the maps of temperature (Figure 2.1.3) and that of the 22 species (Figures 2.1.4–2.1.23). They describe temperature preferenda based on abundance occurrence. Overall as seen visually on the figures (2.1.24–2.1.45), the preferred temperature seems to be more variable between seasons than across years. All species are proportionally more abundant in colder waters in Quarter 1 than in Quarter 3, thus showing a consistent seasonal difference in their preferred temperature range.

Two groups of species can be distinguished for the GOV sampler. A first group in which seasonal differences are minimal and centred around 10°C and a second group in which seasonal differences are important, lower than 10°C in Quarter 1 and higher than 15°C in Quarter 3. The first group comprises all the demersal species except plaice, sole, red mullet, bib, and thornback ray. The demersal exceptions can be classified in the second group together with all the pelagic species.

A closer look at the quotient plots across years shows that in recent years the thermal preferendum of several species is warmer than in the previous periods. This is the case for example for sole (*Solea vulgaris*) and plaice (*Pleuronectes platessa*) in the southern areas (GOV sampling) as shown in Figure 2.1.46. This change in temperature preferendum may indicate a degree of adaptation of the populations to changes in the temperature of their local environment, or simply a low dependence upon temperature conditions in comparison with other factors that are geographically stable (e.g. bathymetry, sediment types).

For some species, the northern and southern populations appear to have distinct thermal habitat. This is the case for thorny skate (*Amblyraja radiata*) for which thermal preferendum in quarter 1 is around 5°C in northern areas and around 7–8°C in the south (Figure 2.1.47). A similar picture is observed for herring in quarter 3 with thermal preferendum around 0-5°C in northern areas and around 10–15°C in the south.

2.1.3.3.1 Pelagic species

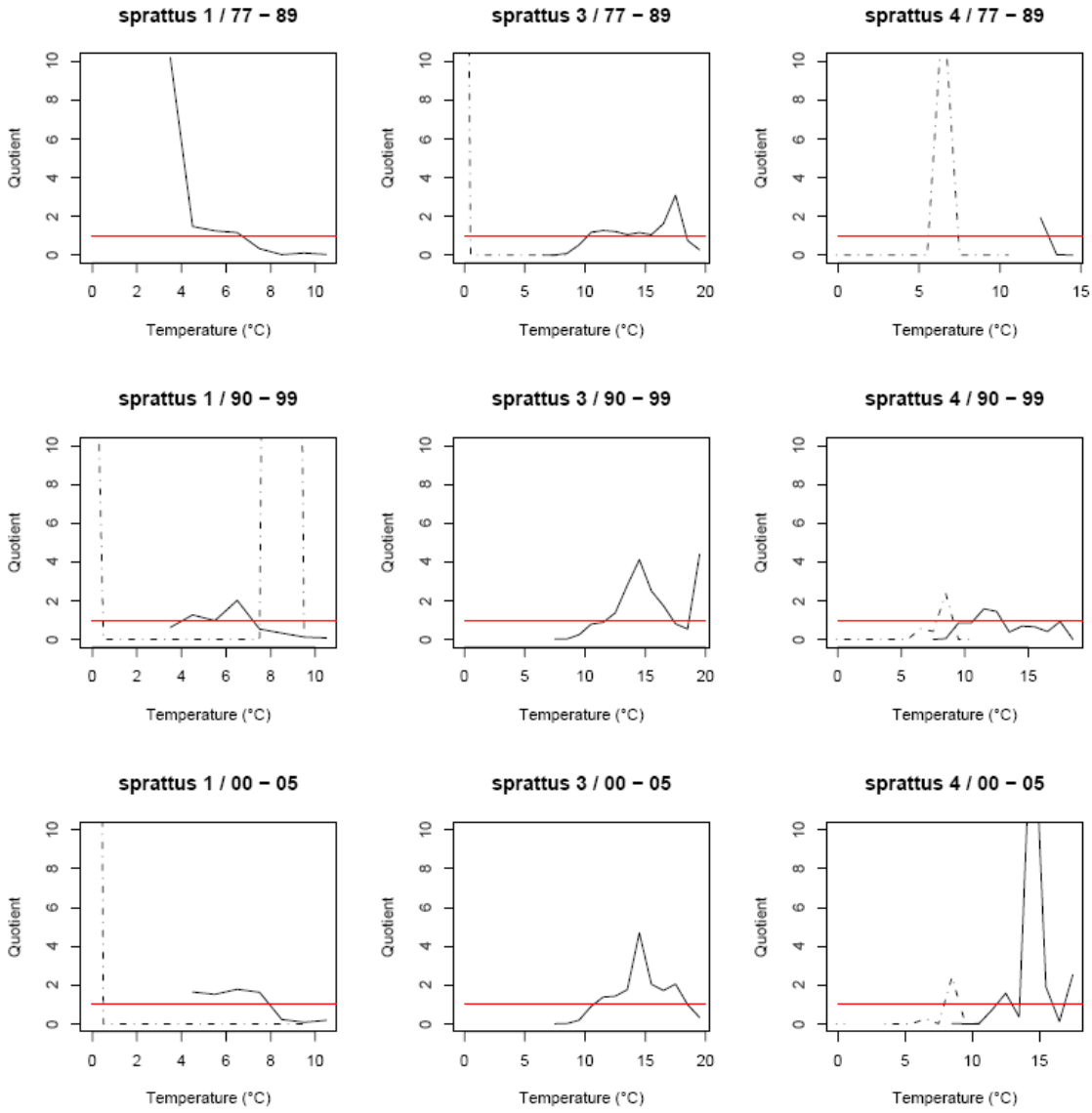


Figure 2.1.25. Quotient plots for *Sprattus sprattus* (sprat) by quarter and three periods. A ratio of one signifies 'no preference' for a given temperature range. A ratio above 1 means preference and a ratio below one means avoidance of a given temperature range.

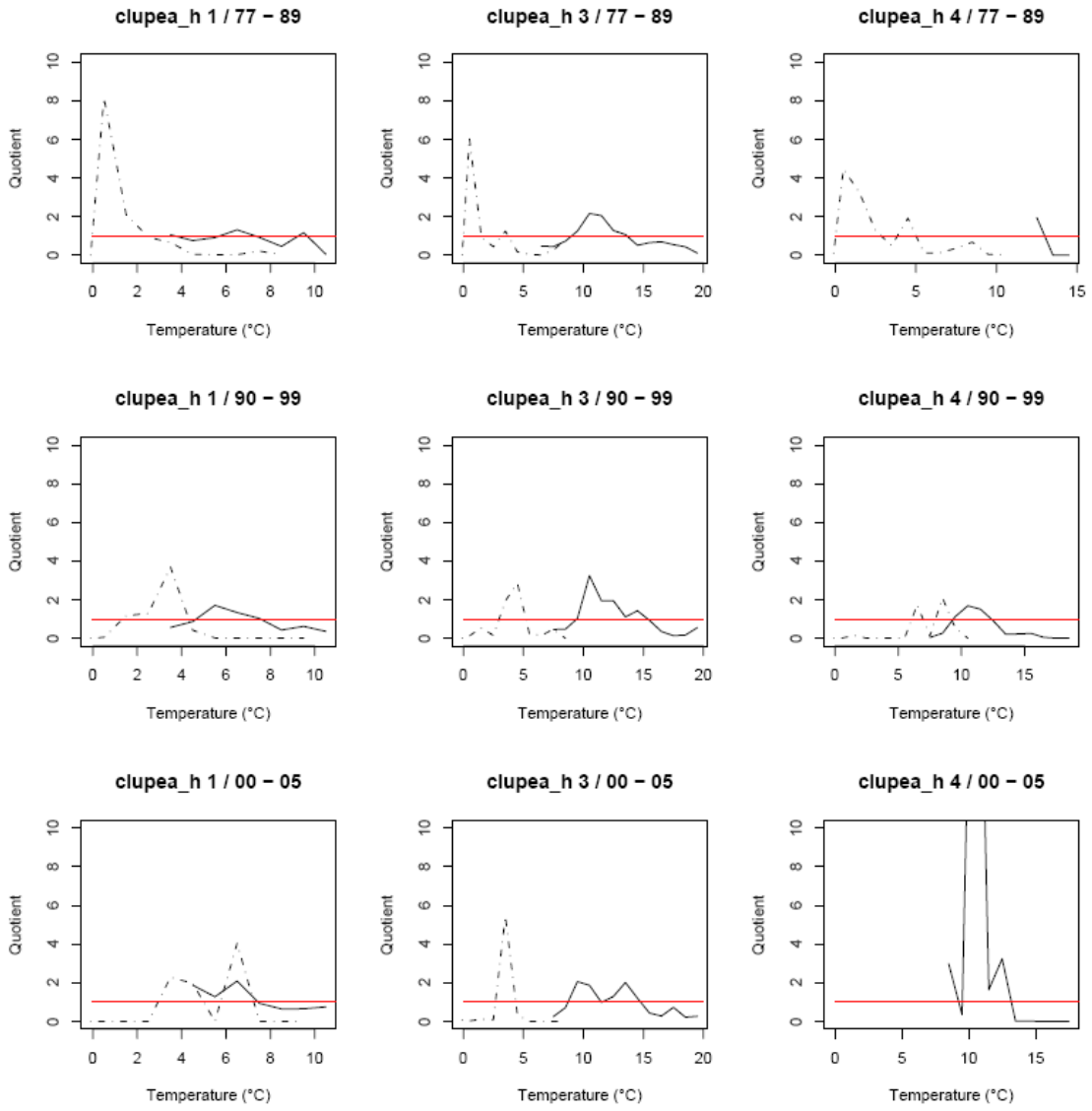


Figure 2.1.26. Quotient plots for *Clupea harengus* (herring) by quarter and three periods. A ratio of one signifies 'no preference' for a given temperature range. A ratio above 1 means preference and a ratio below one means avoidance of a given temperature range.

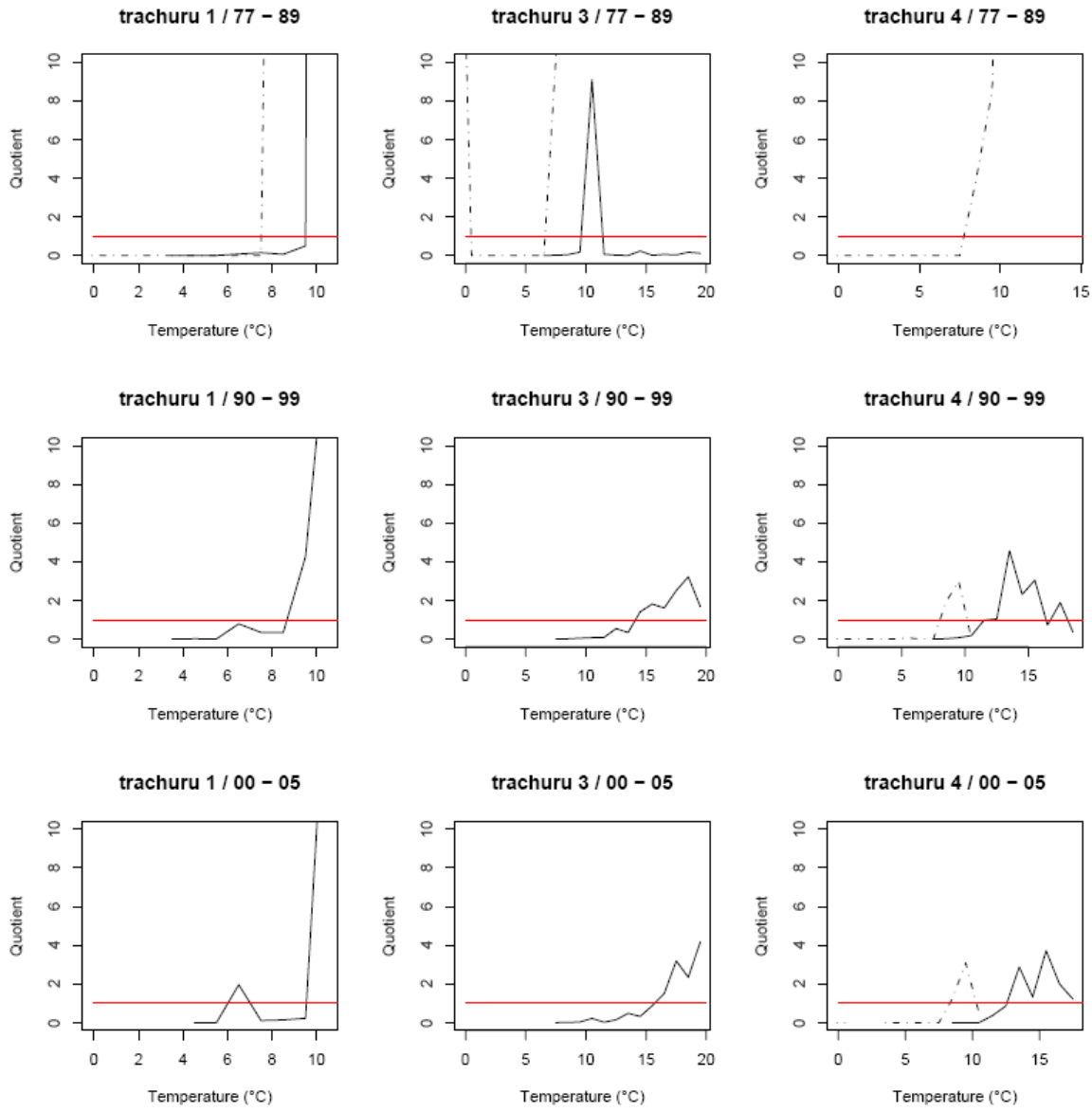


Figure 2.1.27. Quotient plots for *Trachurus trachurus* (horse mackerel) by quarter and three periods. A ratio of one signifies 'no preference' for a given temperature range. A ratio above 1 means preference and a ratio below one means avoidance of a given temperature range.

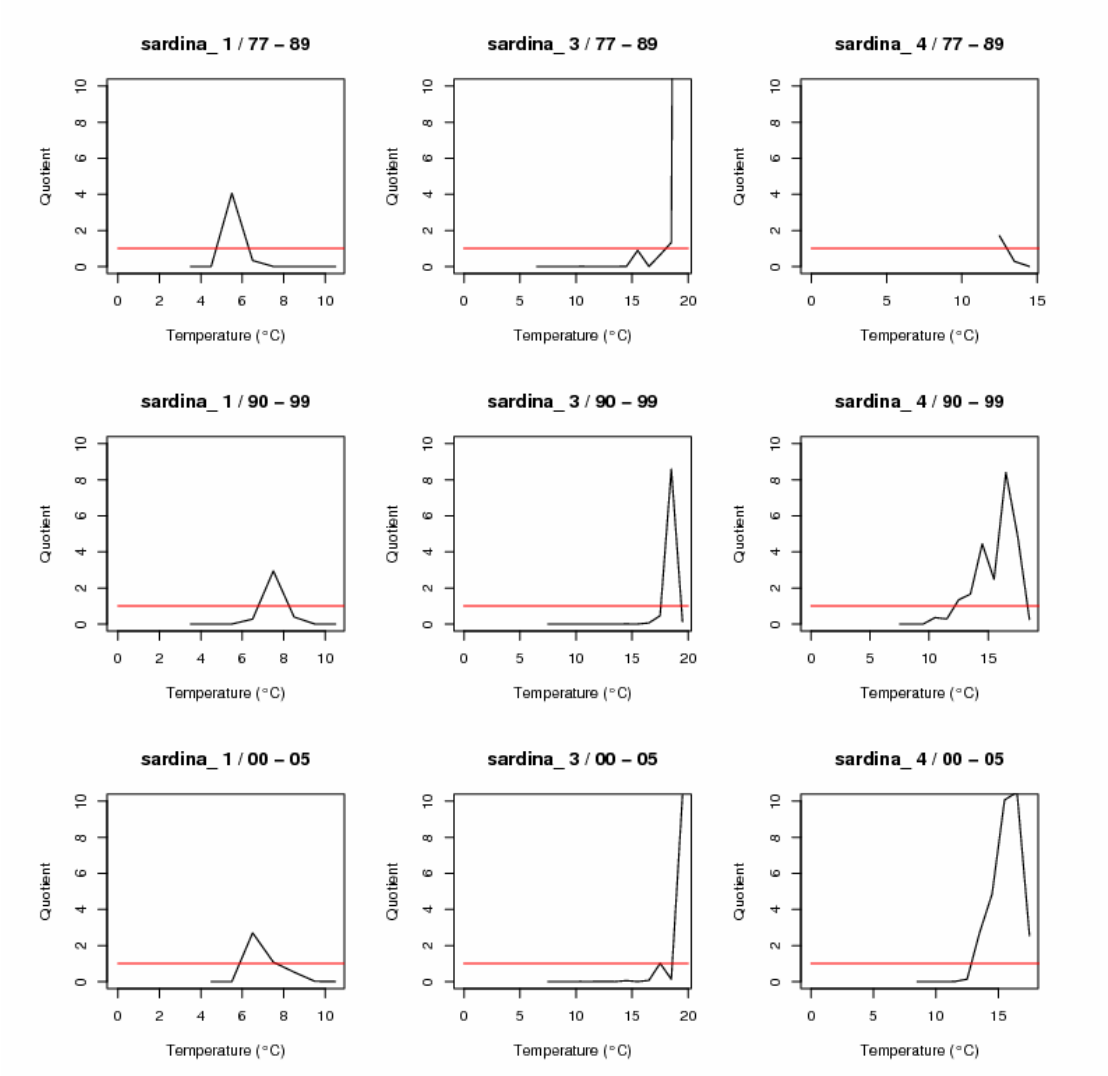


Figure 2.1.28. Quotient plots for *Sardina pilchardus* (sardine) by quarter and three periods. A ratio of one signifies 'no preference' for a given temperature range. A ratio above 1 means preference and a ratio below one means avoidance of a given temperature range.

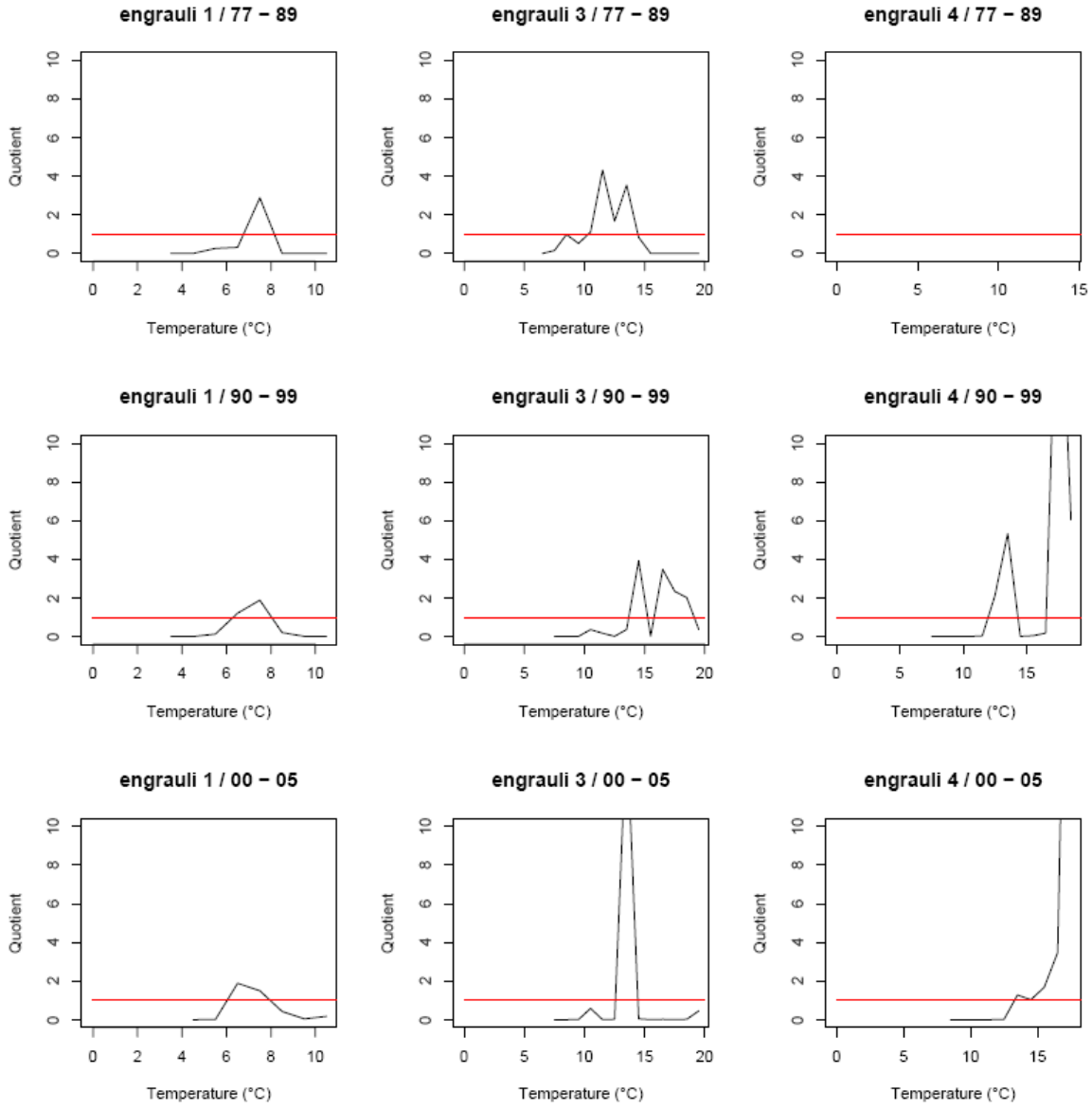


Figure 2.1.29. Quotient plots for *Engraulis encrasicolus* (anchovy), by quarter and three periods. A ratio of one signifies 'no preference' for a given temperature range. A ratio above 1 means preference and a ratio below one means avoidance of a given temperature range.

2.1.3.3.2 Demersal species

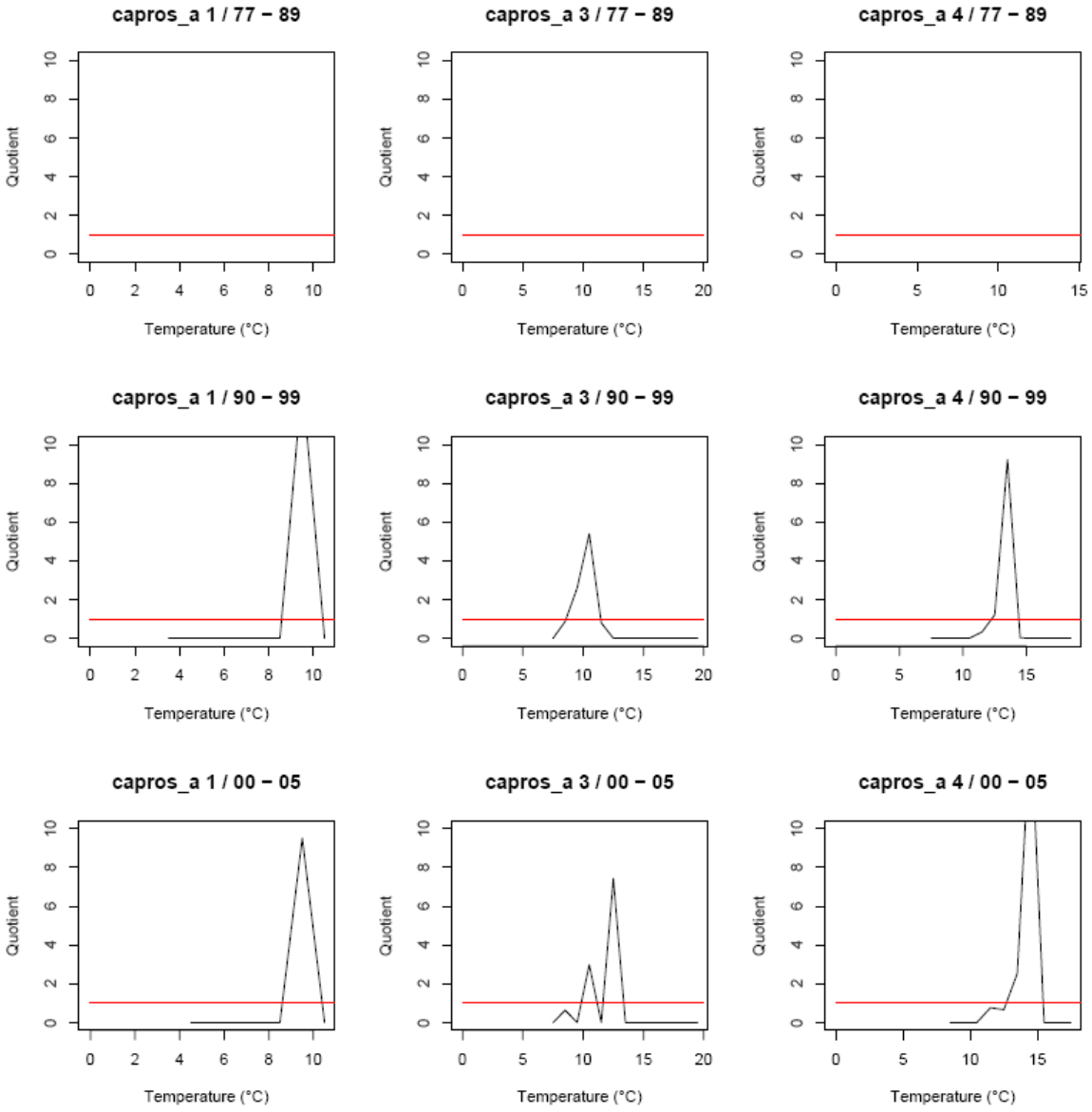


Figure 2.1.30. Quotient plots for *Capros asper* (boarfish) by quarter and three periods. A ratio of one signifies 'no preference' for a given temperature range. A ratio above 1 means preference and a ratio below one means avoidance of a given temperature range.

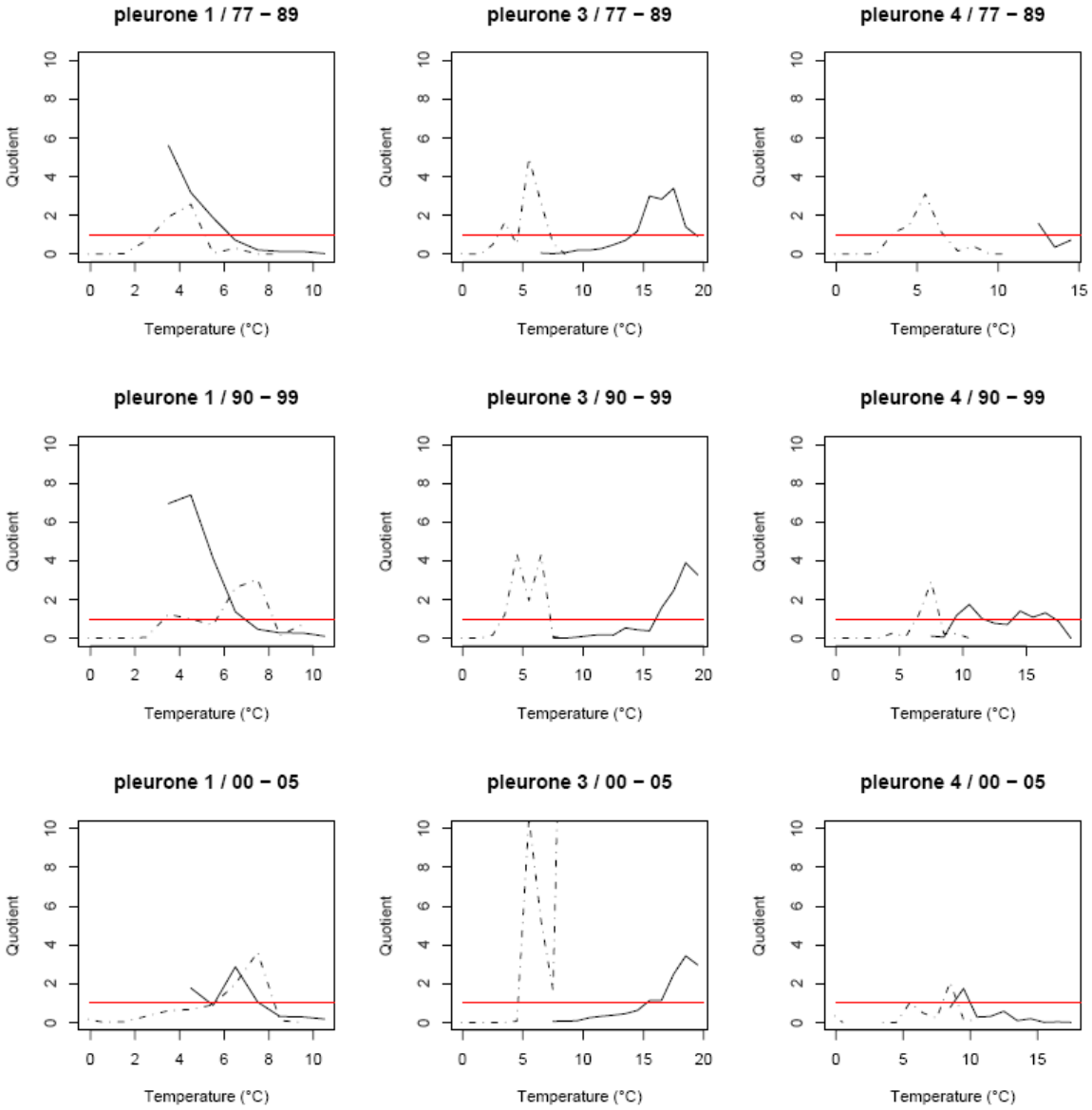


Figure 2.1.31. Quotient plots for Pleuronectes platessa (American plaice) by quarter and three periods. A ratio of one signifies 'no preference' for a given temperature range. A ratio above 1 means preference and a ratio below one means avoidance of a given temperature range.

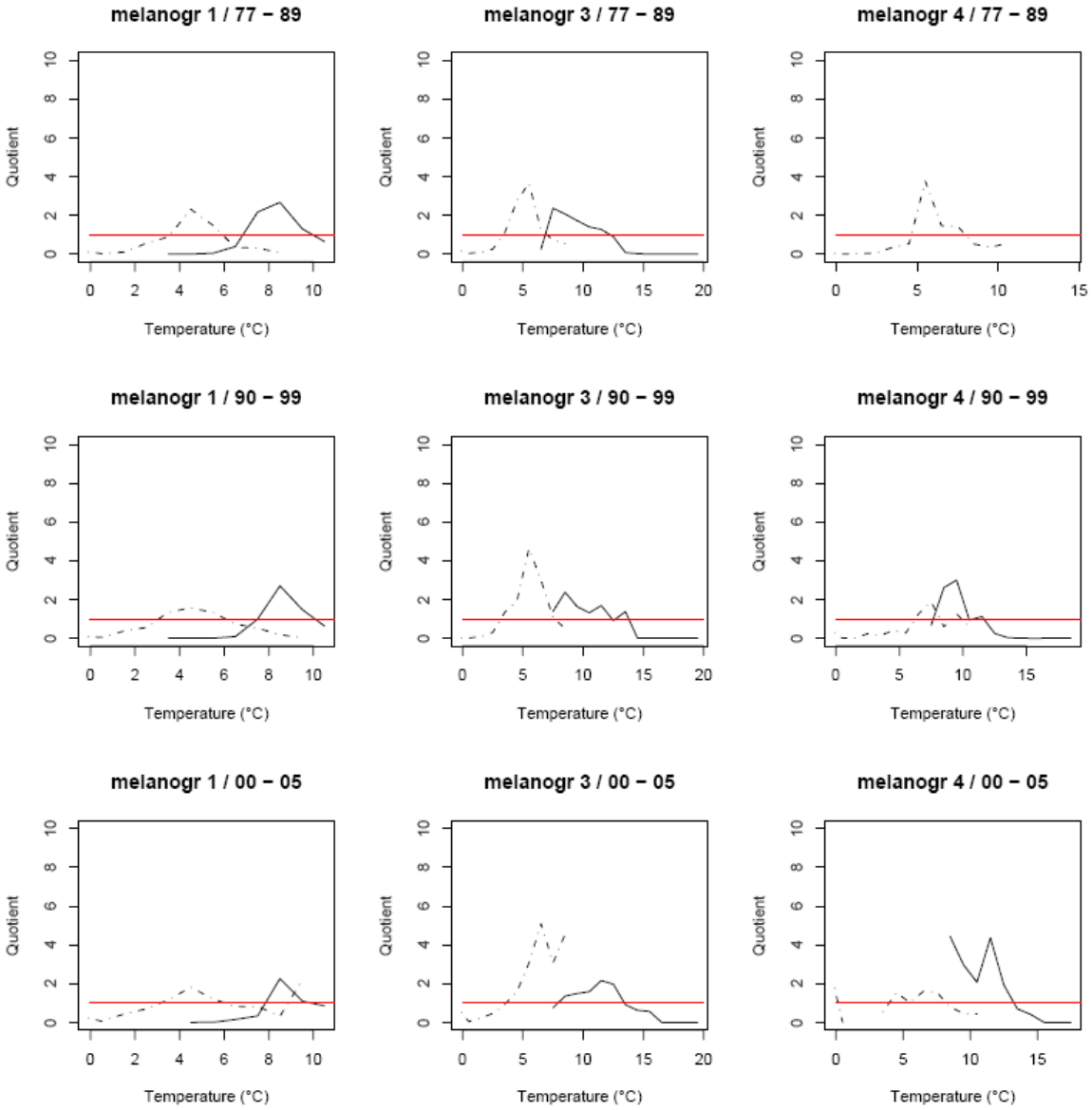


Figure 2.1.32. Quotient plots for *Melanogrammus aeglefinus* (haddock) by quarter and three periods. A ratio of one signifies 'no preference' for a given temperature range. A ratio above 1 means preference and a ratio below one means avoidance of a given temperature range.

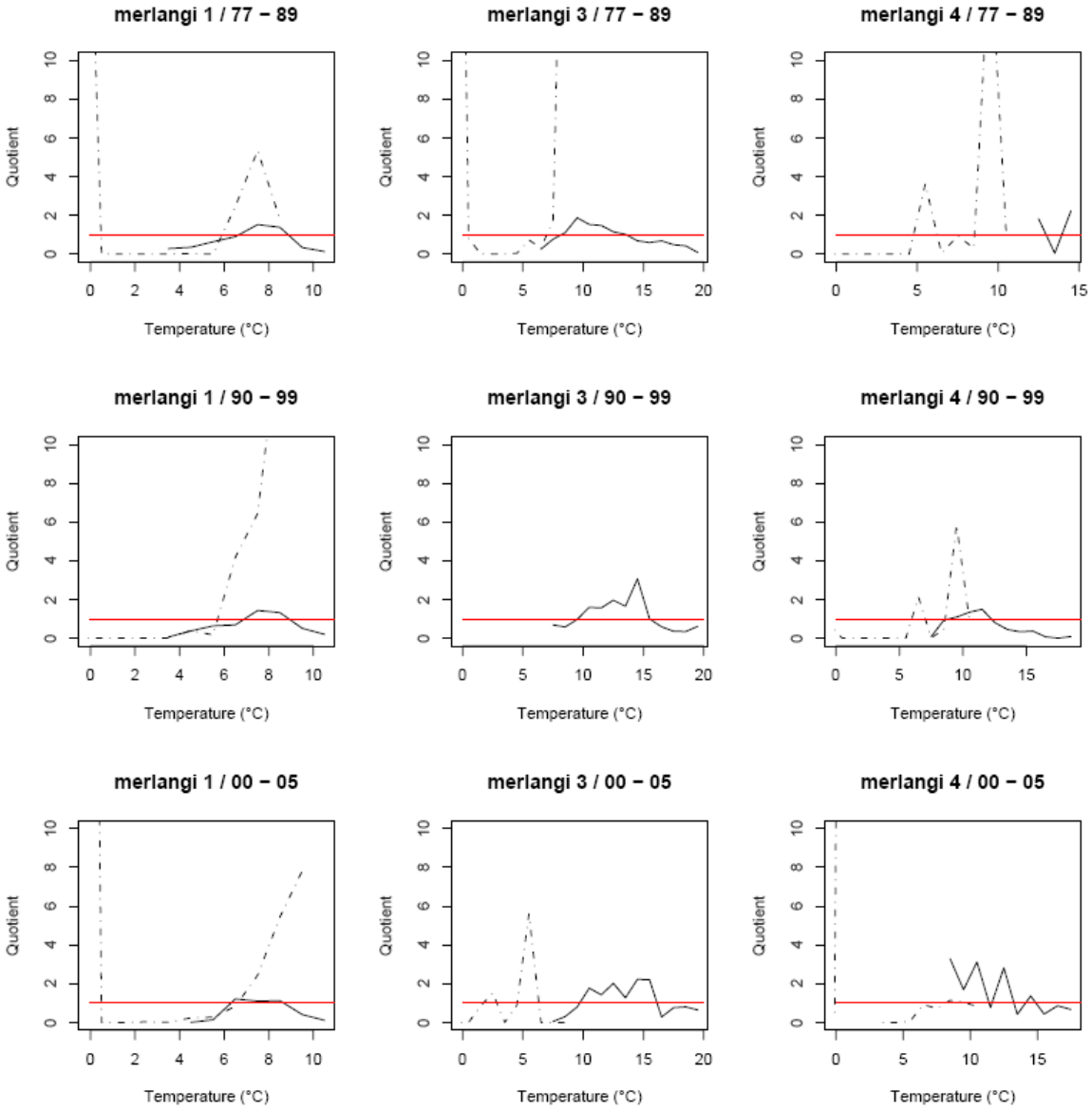


Figure 2.1.33. Quotient plots for *Merlangius merlangus* (whiting) by quarter and three periods. A ratio of one signifies 'no preference' for a given temperature range. A ratio above 1 means preference and a ratio below one means avoidance of a given temperature range.

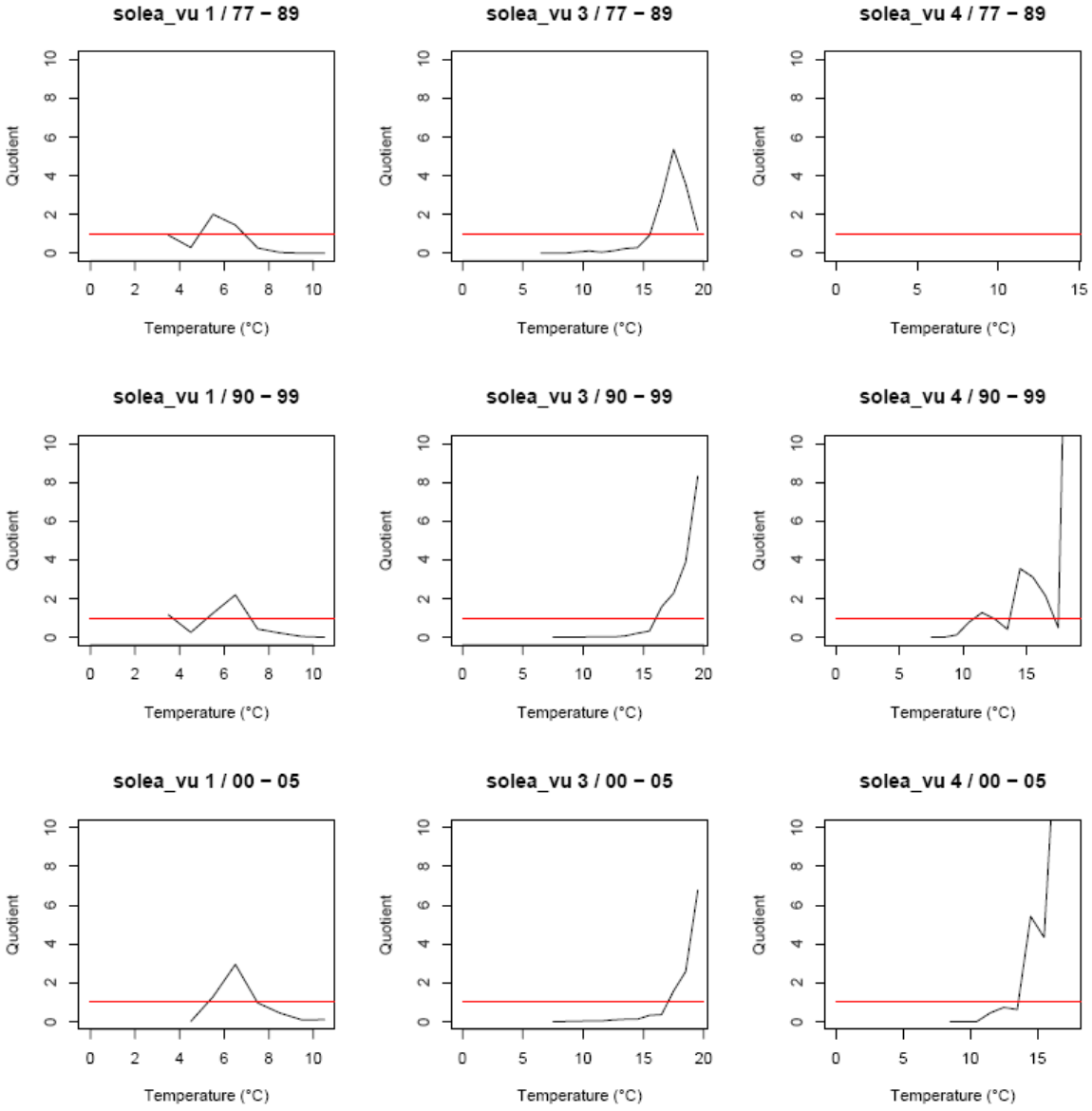


Figure 2.1.34. Quotient plots for *Solea solea* (sole) by quarter and three periods. A ratio of one signifies 'no preference' for a given temperature range. A ratio above 1 means preference and a ratio below one means avoidance of a given temperature range.

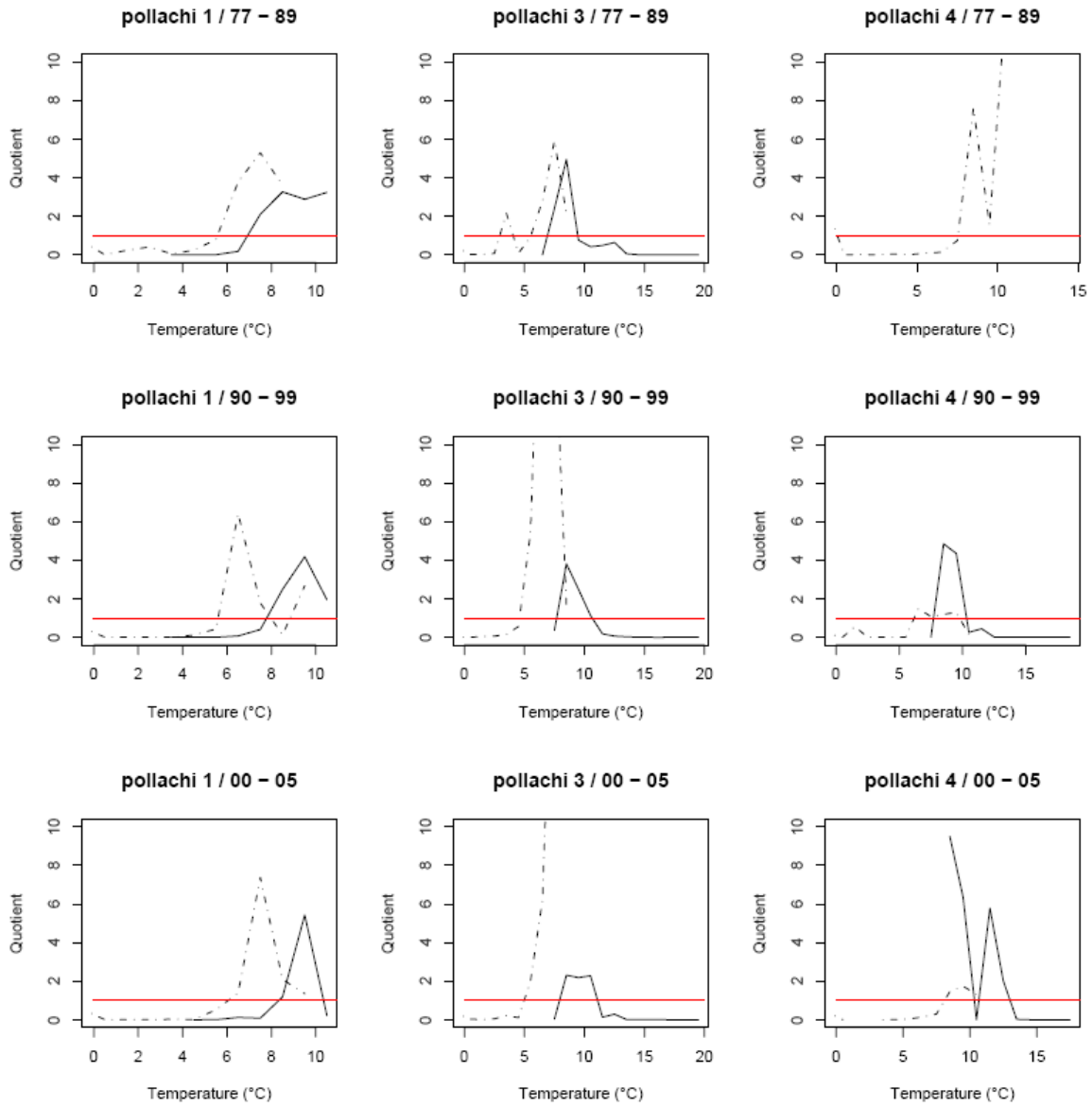


Figure 2.1.35. Quotient plots for *Pollachias virens* (saithe) by quarter and three periods. A ratio of one signifies 'no preference' for a given temperature range. A ratio above 1 means preference and a ratio below one means avoidance of a given temperature range.

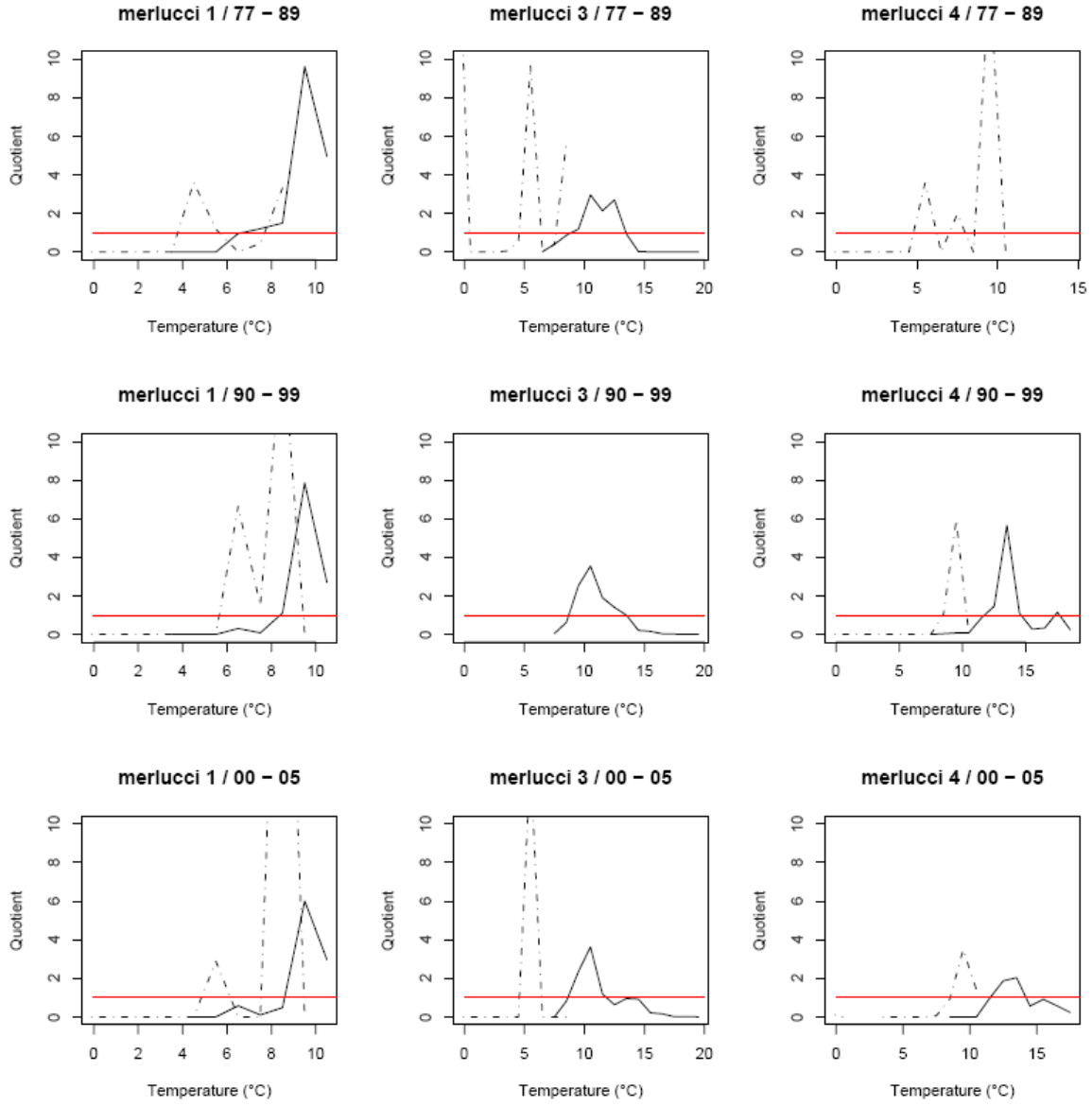


Figure 2.1.36. Quotient plots for *Merluccius merluccius* (hake) by quarter and three periods. A ratio of one signifies 'no preference' for a given temperature range. A ratio above 1 means preference and a ratio below one means avoidance of a given temperature range.

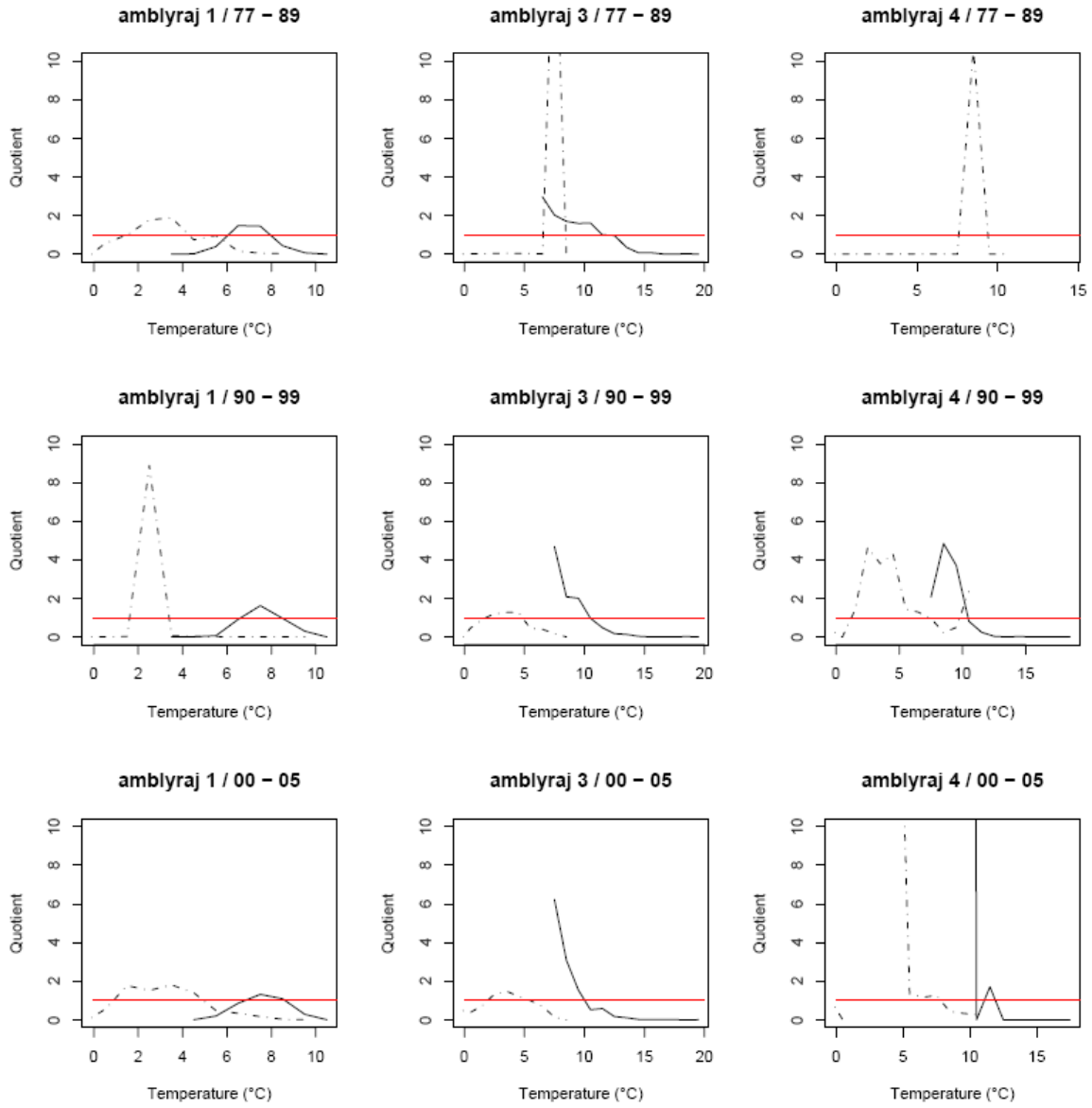


Figure 2.1.37. Quotient plots for *Amblyraja radiata* (starry ray) by quarter and three periods. A ratio of one signifies 'no preference' for a given temperature range. A ratio above 1 means preference and a ratio below one means avoidance of a given temperature range.

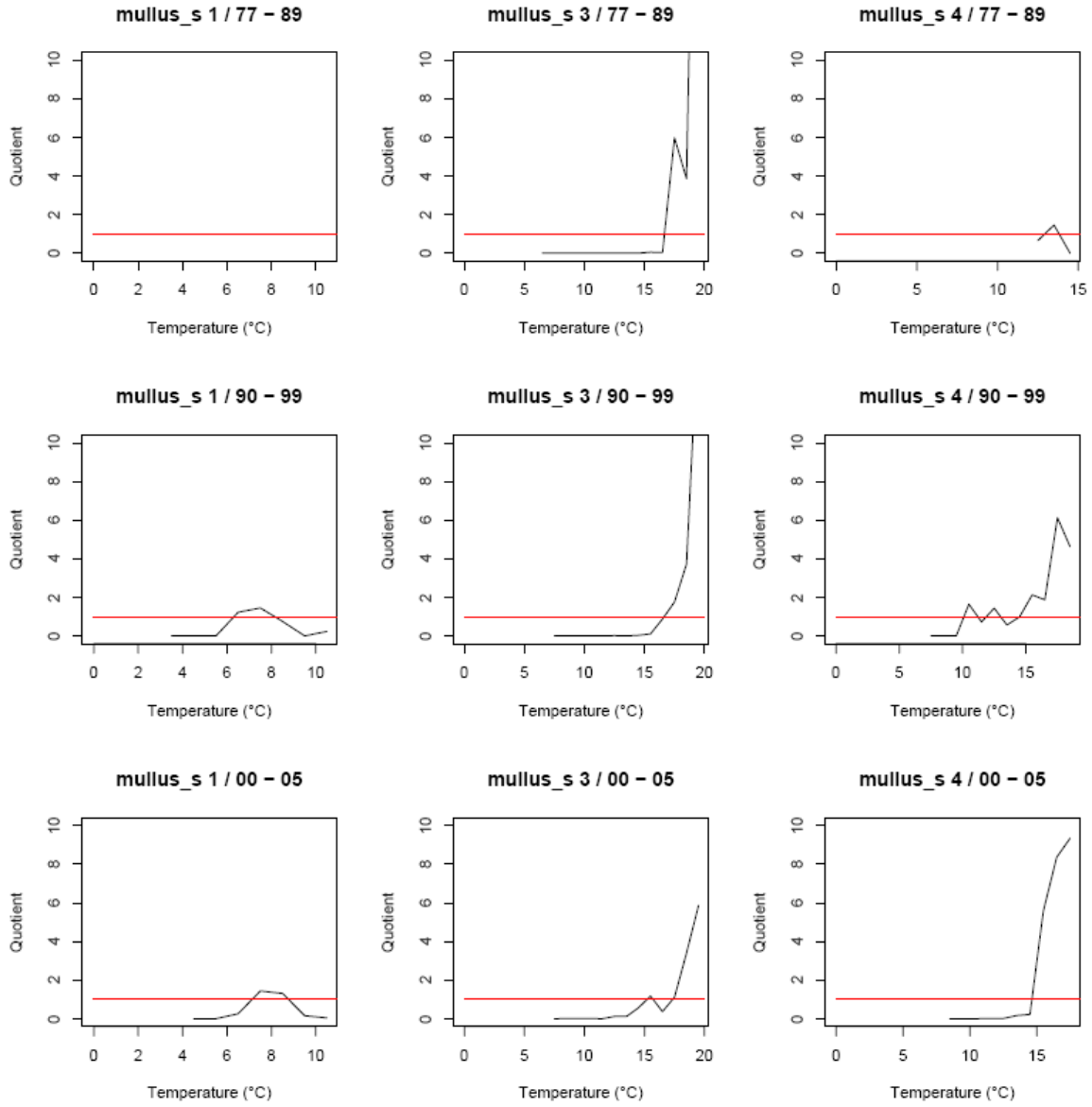


Figure 2.1.38. Quotient plots for *Mullus surmullus* (striped red mullet) by quarter and three periods. A ratio of one signifies 'no preference' for a given temperature range. A ratio above 1 means preference and a ratio below one means avoidance of a given temperature range.

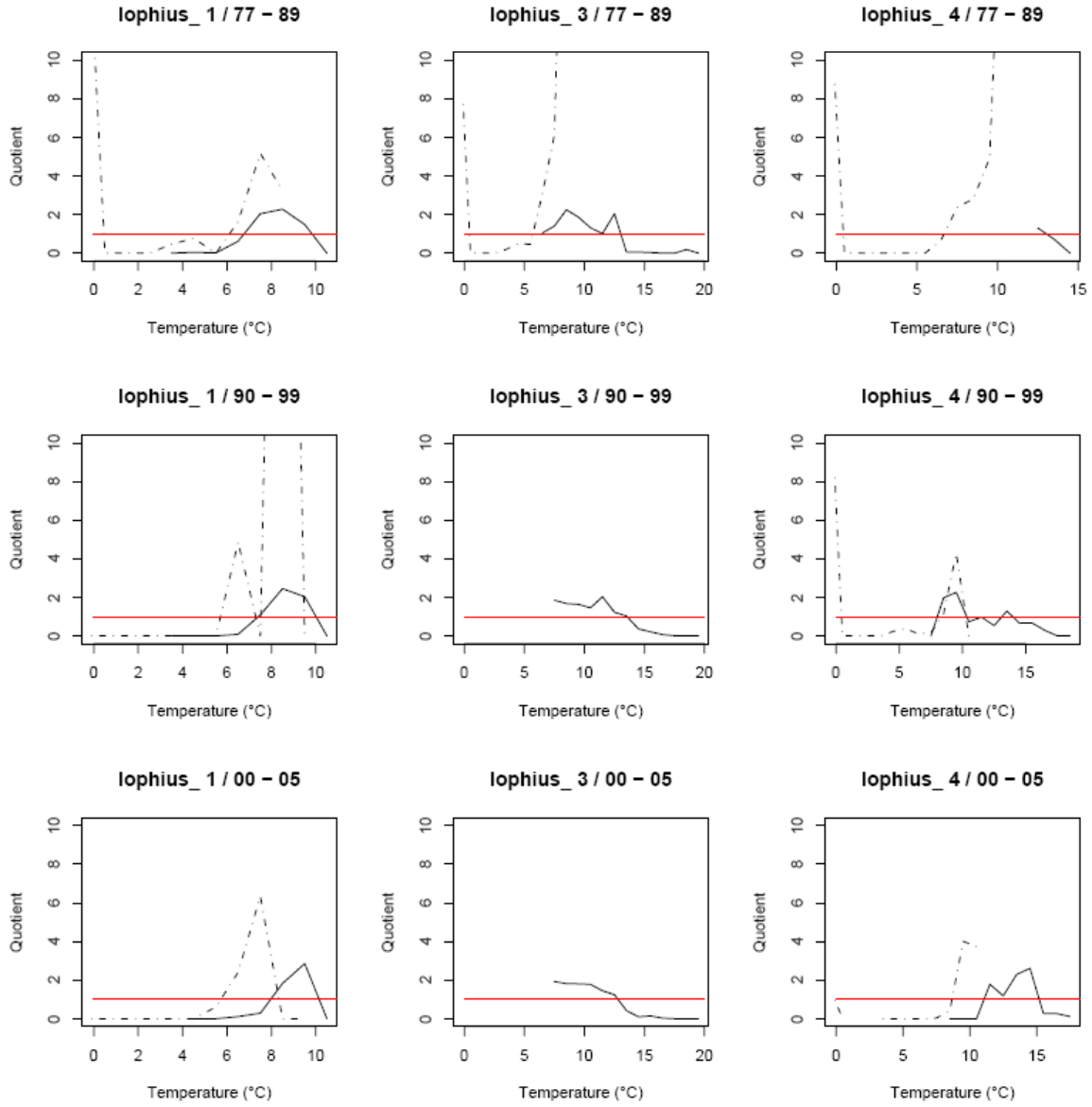


Figure 2.1.39. Quotient plots for *Lophius piscatorius* (monkfish) by quarter and three periods. A ratio of one signifies 'no preference' for a given temperature range. A ratio above 1 means preference and a ratio below one means avoidance of a given temperature range.

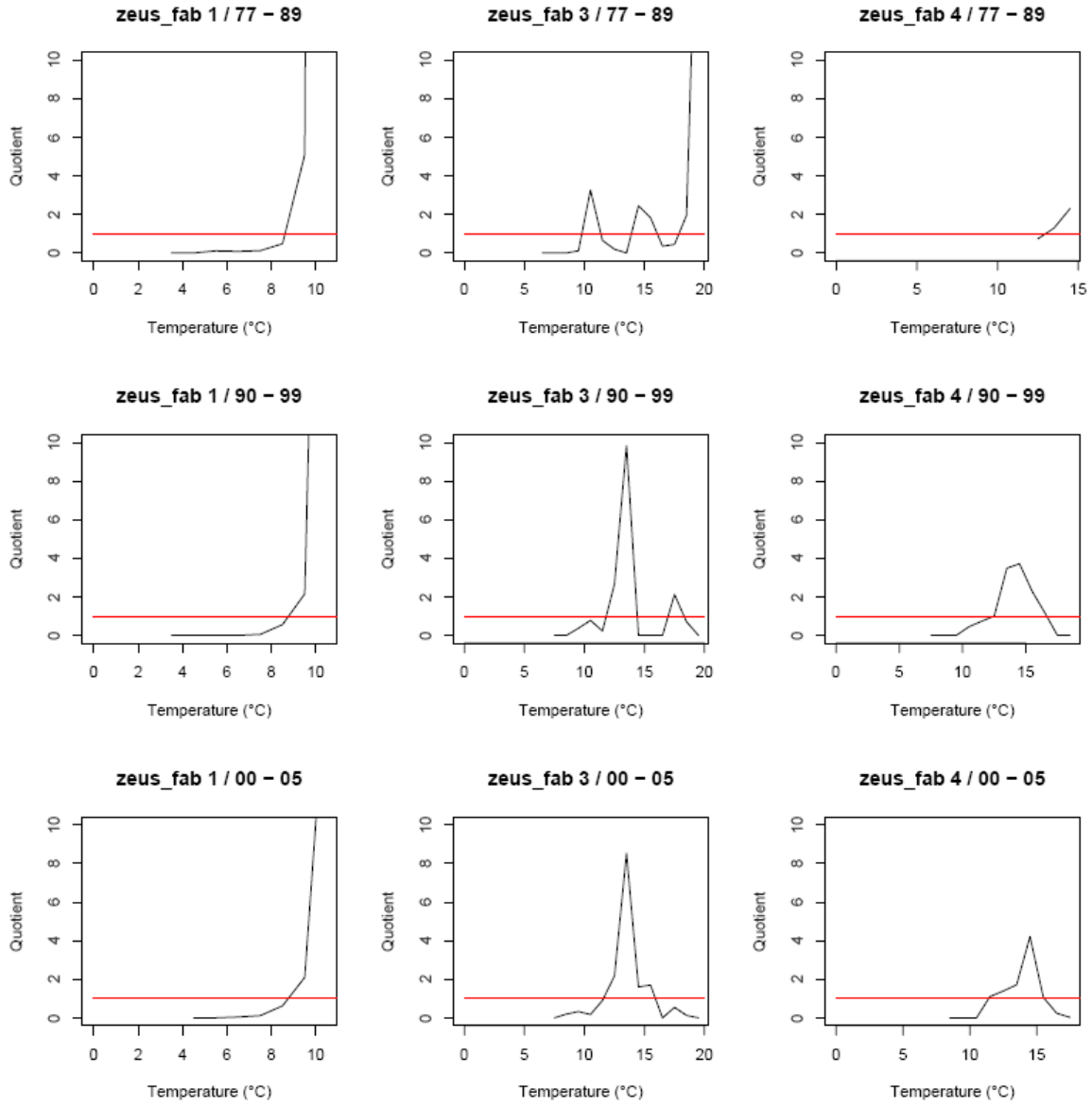


Figure 2.1.40. Quotient plots for *Zeus faber* (John Dory) by quarter and three periods. A ratio of one signifies 'no preference' for a given temperature range. A ratio above 1 means preference and a ratio below one means avoidance of a given temperature range.

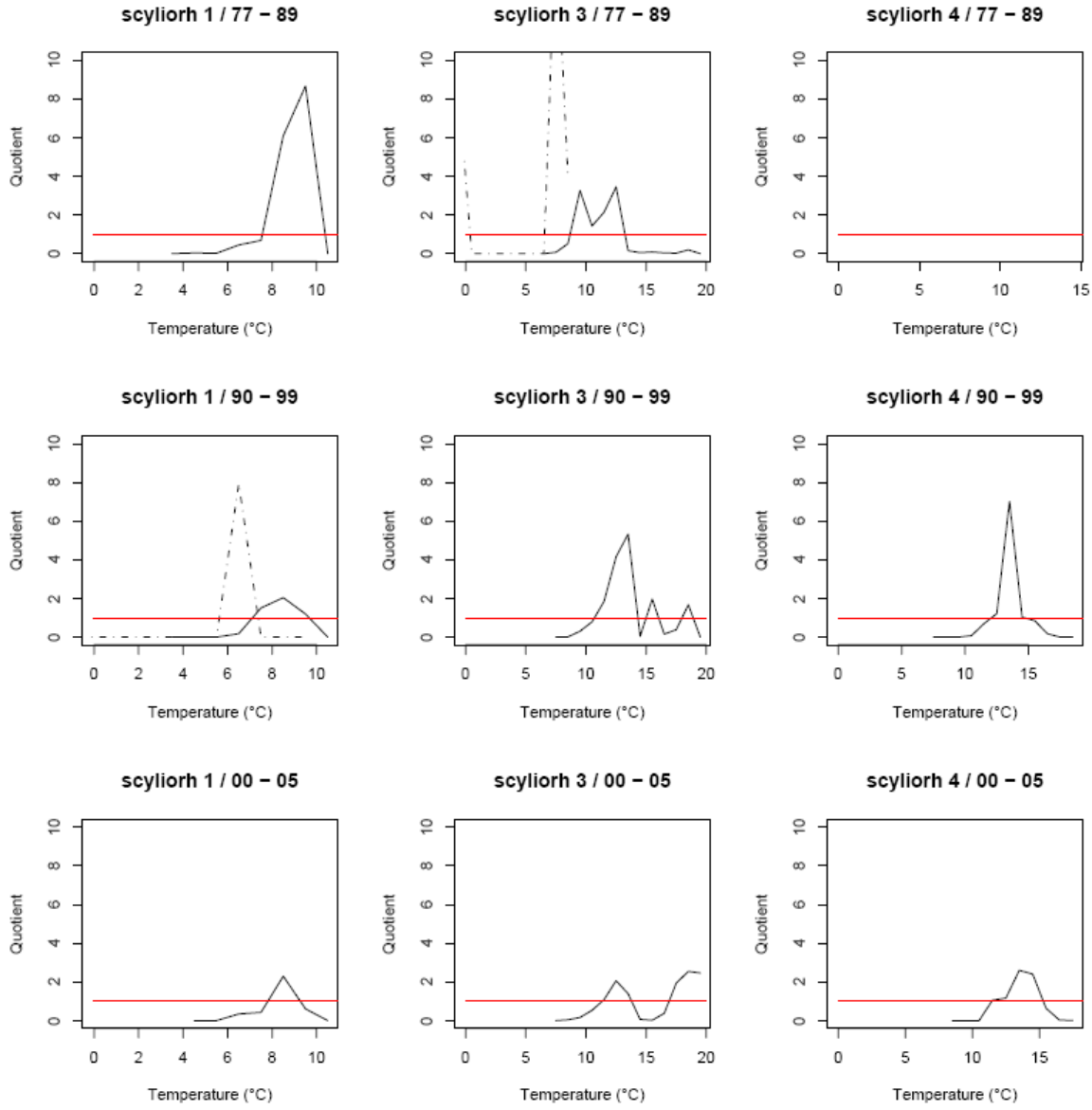


Figure 2.1.41. Quotient plots for *Scyliorhinus canicula* (lesser spotted dogfish) by quarter and three periods. A ratio of one signifies 'no preference' for a given temperature range. A ratio above 1 means preference and a ratio below one means avoidance of a given temperature range.

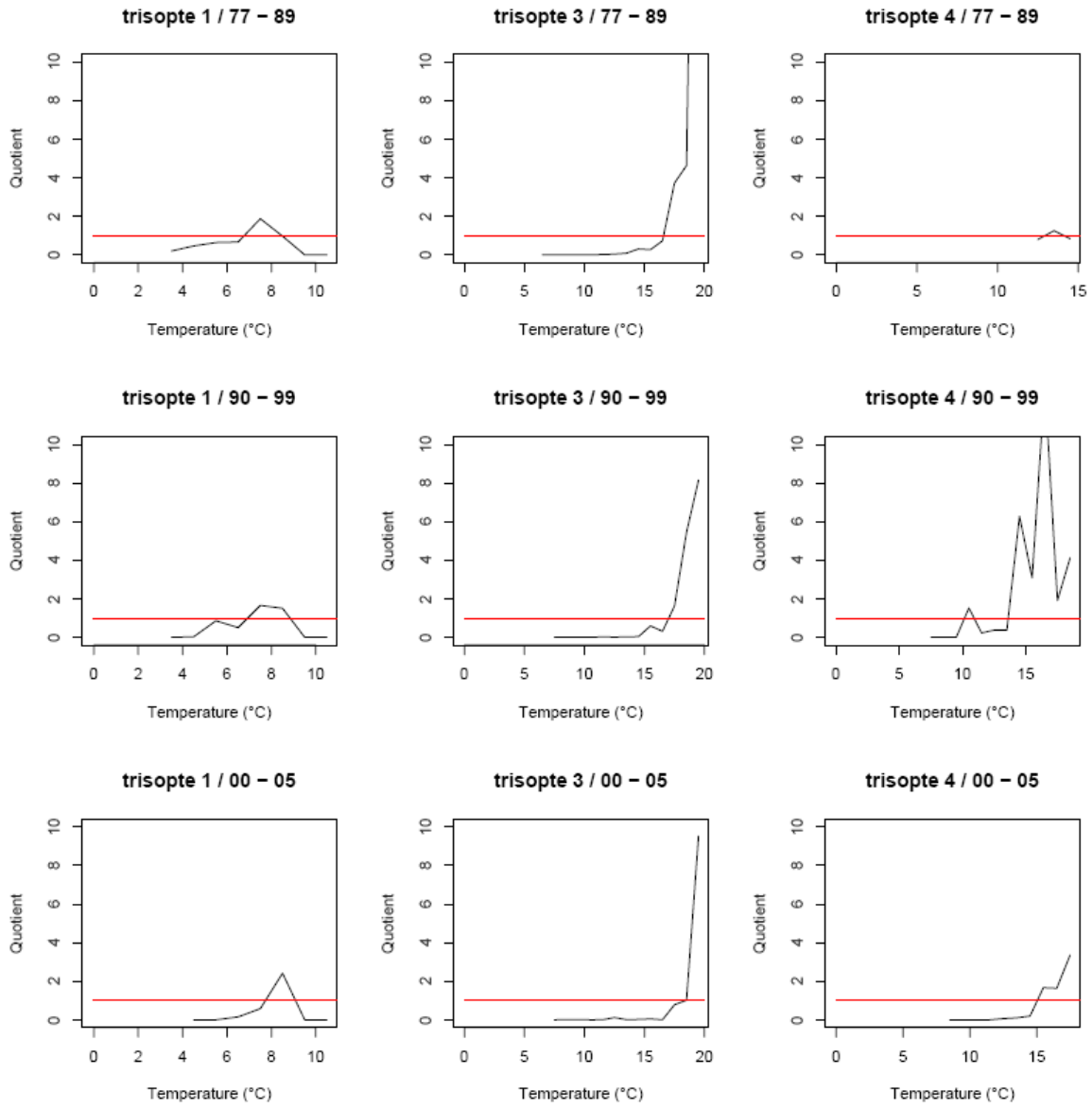


Figure 2.1.42. Quotient plots for *Trisopterus luscus* (bib) by quarter and three periods. A ratio of one signifies 'no preference' for a given temperature range. A ratio above 1 means preference and a ratio below one means avoidance of a given temperature range.

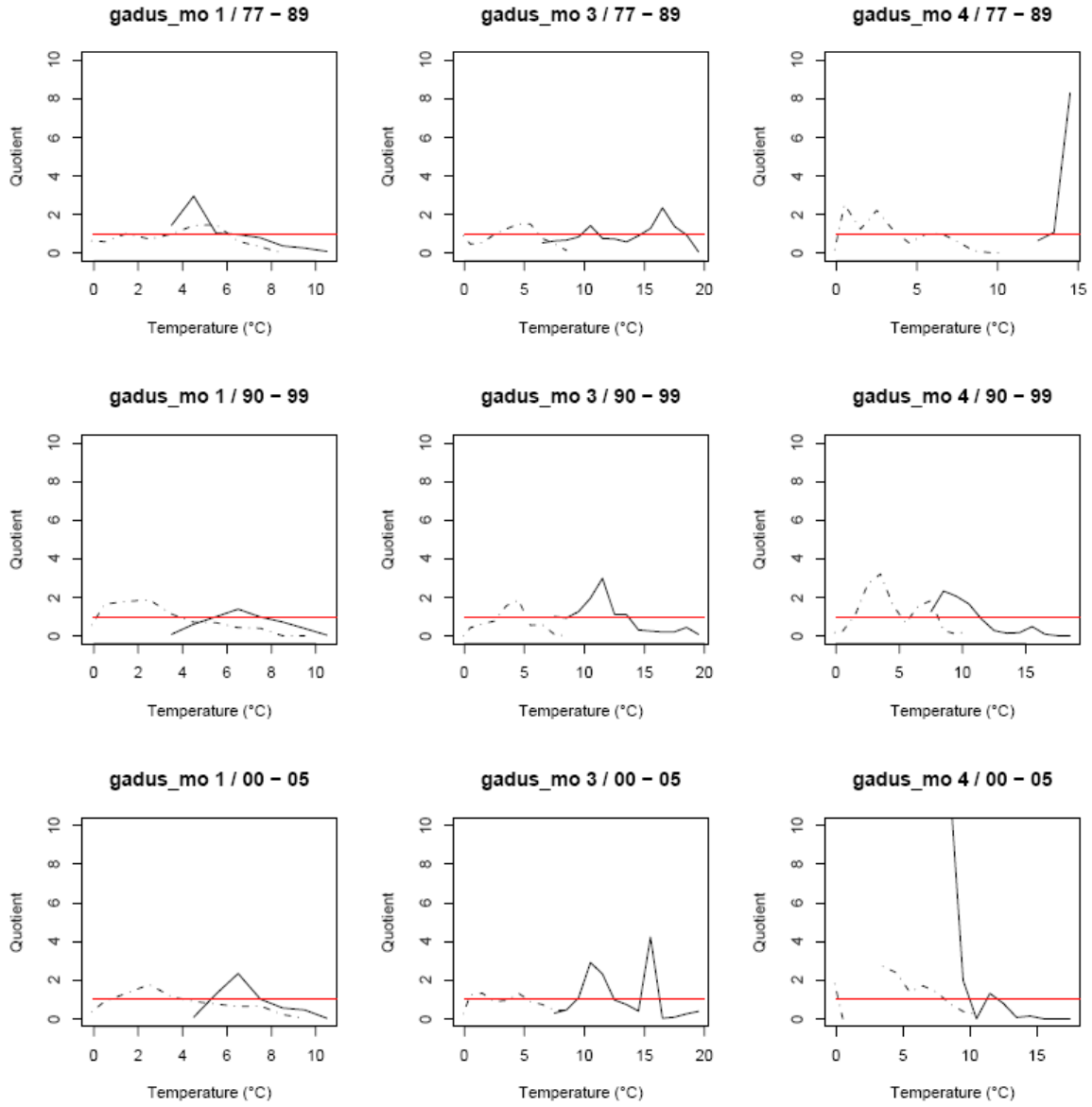


Figure 2.1.43. Quotient plots for *Gadus morhua* (Atlantic cod) by quarter and three periods. A ratio of one signifies 'no preference' for a given temperature range. A ratio above 1 means preference and a ratio below one means avoidance of a given temperature range.

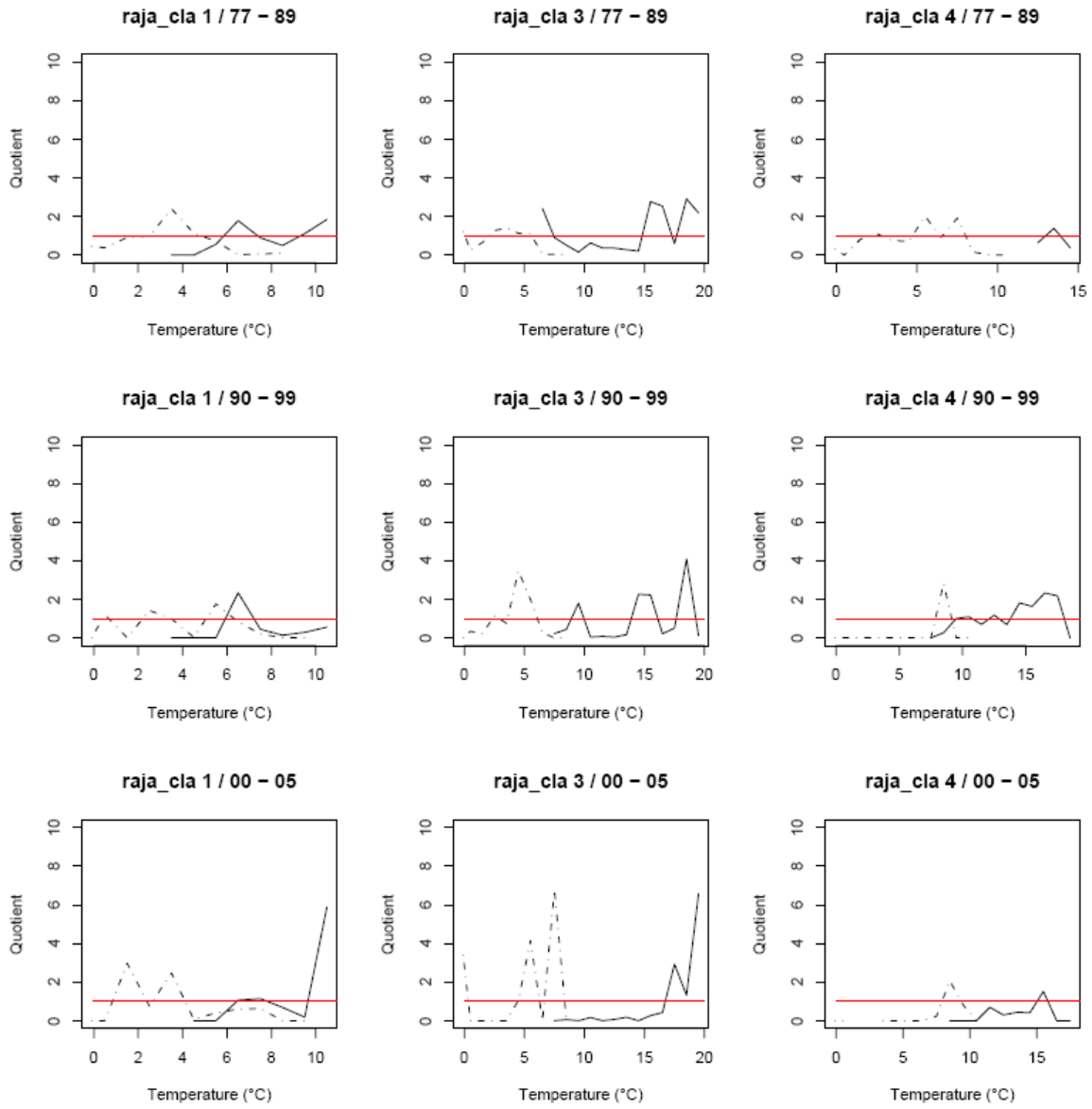


Figure 2.1.44. Quotient plots for *Raja clavata* (thornback ray) by quarter and three periods. A ratio of one signifies 'no preference' for a given temperature range. A ratio above 1 means preference and a ratio below one means avoidance of a given temperature range.

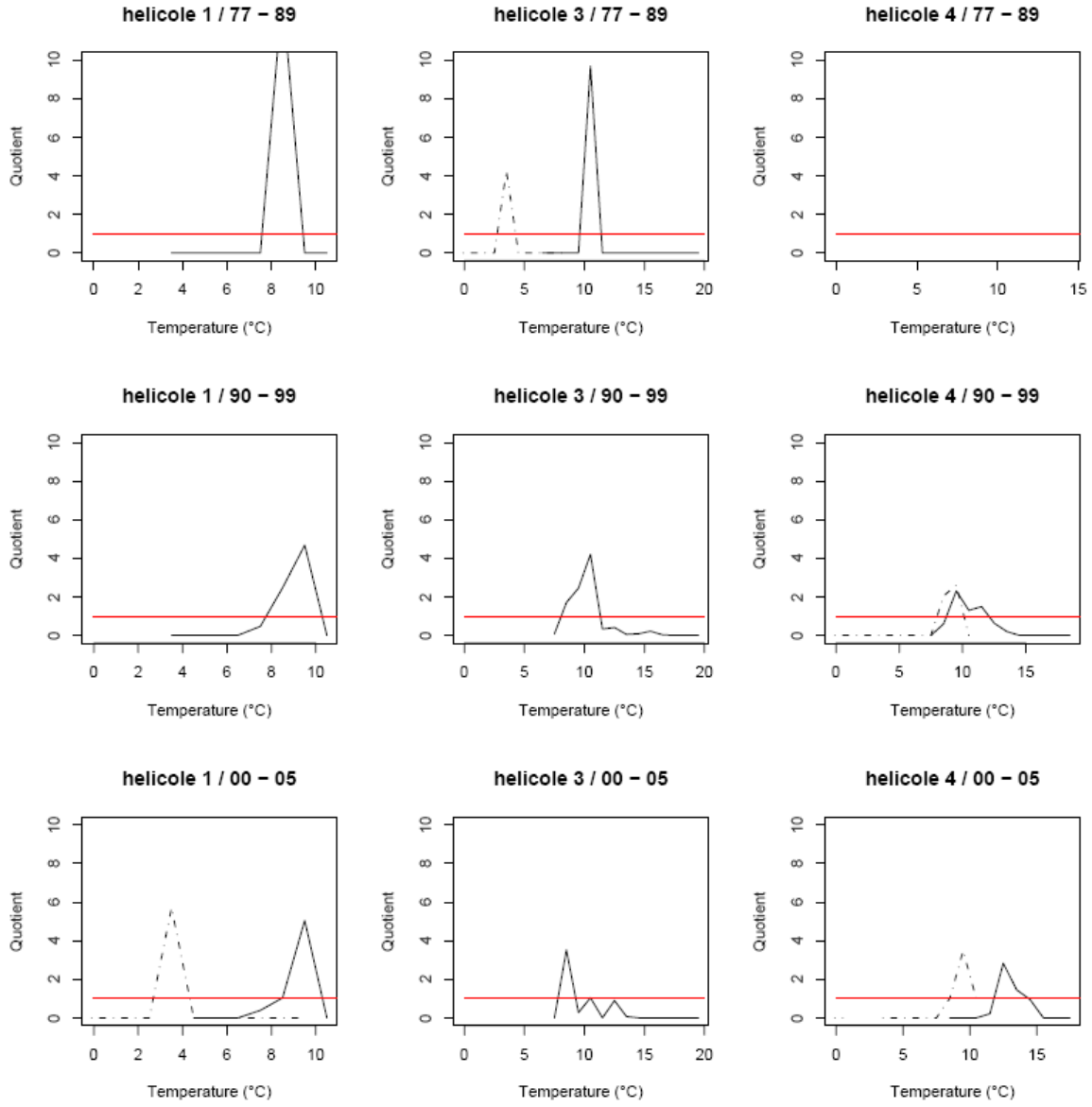


Figure 2.1.45. Quotient plots for *Helicolenus dactylopterus* (blue mouth redfish) by quarter and three periods. A ratio of one signifies 'no preference' for a given temperature range. A ratio above 1 means preference and a ratio below one means avoidance of a given temperature range.

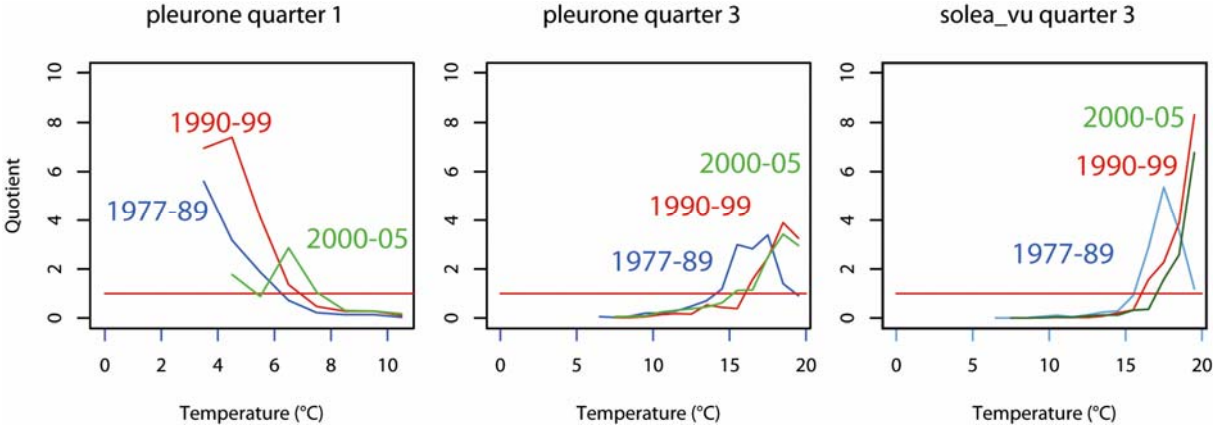


Fig 2.1.46. Changes in temperature preferendum between the three periods 1977–1989, 1990–1999 and 2000–2005 for plaice in quarter one (left) and three (middle) and for sole in quarter three (right) using GOV data. The quotient plots suggest that the preferred temperature for plaice in quarter one has increased steadily whilst for plaice and sole in quarter three the change in temperature preferendum is mostly seen between the first (1977–1989) and second (1990–1999) period.

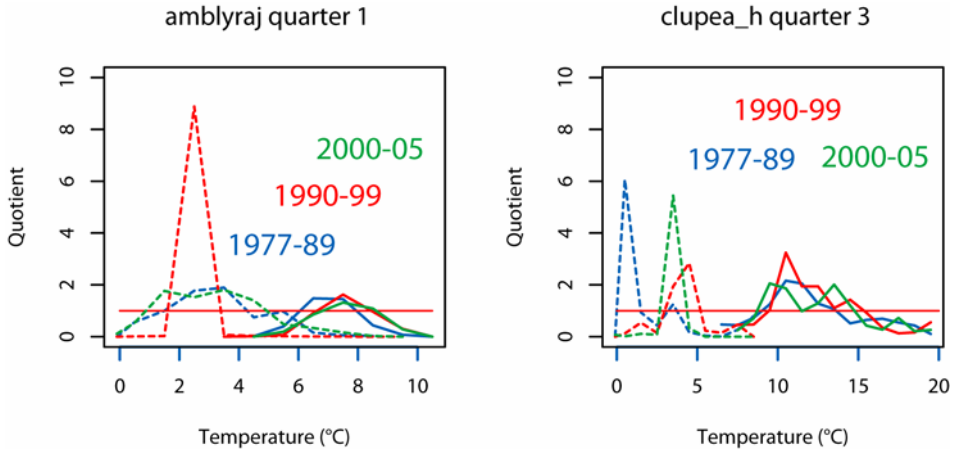


Fig 2.1.47. Changes in temperature preferendum between the three periods 1977–1989, 1990–1999 and 2000–2005 for thorny skate in quarter one (left) and herring in quarter 3 (right) using GOV (plain line) and CAM (dotted line) data. The southern (sampled by GOV) and northern (sampled by CAM) populations show distinct temperature preferendum throughout the period of study.

2.2 Change in spatial distributions and habitat colonisation

2.2.1 Mechanisms for changes in spatial distribution

Consistent changes in spatial distribution result from changes in the spatial organisation of the life cycle and therefore in the occupation of habitats. Different factors will affect the occupation of habitats which can be grouped into two categories: (1) external factors such as hydro-climate will act as forcing conditions on the suitability of the habitats; (2) internal factors such as demography and behaviour will determine the capability of the population to effectively occupy all its potential habitats. Changes in spatial distributions occur because the distribution of potential habitats changes with climate or because the population's internal behaviour changed under demographic change. An ecological typology of habitats was proposed by GLOBEC (van der Lingen *et al.*, 2005) which distinguished potential habitats from effectively occupied habitats. The capacity to fully use potential habitats will be determined by species interactions as well as internal population characteristics. Behaviour and learning between generations was recognised to play a major role in governing internal population capacity to occupy potential habitats (e.g. McQuinn, 1997; Corten, 2001; Huse *et al.*, 2002; Petitgas *et al.*, 2006).

The many changes in habitat characteristics and occupation are listed in Table 2.2.1 with their underlying mechanism. A variety of different data, methods and tools are necessary to evidence the different types of changes and their mechanisms (Table 2.2.1). Thus long-term large-scale fisheries data and sea surface temperature which have been used sofar to address change in fish distribution under climate change can only allow to examine changes in the potentiality of habitats. A larger variety of data and tools would allow us to address combined changes in potential suitability of habitats and in realised occupancy, in particular, dynamic models based on physical biological interactions and models of species interactions.

Temperature is a logical explanatory environmental parameter to consider as it is implicated in many biological processes. The many direct and indirect implications of temperature are listed in Table 2.2.2. Based on that Table, changes in fish spatial distribution that can be related to temperature are expected to be:

- recruitment pulse and increase in occupancy resulting from density-dependence and habitat suitability
- change in adult migration timing
- major forcing on habitat suitability modifying adult distribution [El Niño type of forcing]

But temperature may not always convey the appropriate signal of environmental forcing. Depending on ecosystems, river discharge or oxygen could be more appropriate proxies.

Table 2.2.1. List of factors affecting habitat occupation, type of habitat change, data and tools necessary to identify habitat change.

MECHANISM	DISTRIBUTION CHANGE	EXAMPLES	DATA, TOOLS & METHODS
External potential factors : -Climate and ecosystem forcing habitat suitability	-Change in potential habitats	-Expansion of anchovy spawning habitat in the North Sea (ICES, 2006) -Change in NEA mackerel migration in the northern North Sea (Reid et al, 2006)	-Historical series : Fish, hydrology -Statistical analysis and modelling -Satellite images -Dynamic model outputs : hydrodynamics, coupled NPZD
Internal potential factors : -Behaviour, learning between generations, demography, numerical dominance -Old fish : movement and choice of habitats [conservatism] -Young fish : search for suitable habitats [innovation and numerical dominance]	-Change in capacity to occupy potential habitats (gain / loss) :	-Re-colonisation by herring of Aberdeen bank as a spawning ground (Corten, 2001) -Re-colonisation by herring of Georges bank as a spawning ground (Smith and Morse, 1990)	-Historical series : Length / age -Population demography and distribution Focus on habitats where loss and re-colonisation occurred -Experimental surveys (acoustics) : behaviour (schooling, feeding) -Individual markers
Interaction external / internal : -Mortality, growth (larvae, juveniles, adults) -Density dependence (ideal free distribution) -Species interactions	Change in habitat realised occupancy	-Variation in the occupation of potential spawning habitat for anchovy and sardine (ICES, 2006) -North Sea cod habitat as an interaction between temperature, density dependence and growth potential (Blanchard *et al.*, 2005)	-Fish data: Growth, Abundance -Static and statistical models -Dynamic and mechanistical fish models coupled with NPZD models: Larvae drift and survival IBM Adult growth and reproduction IBM -Multi-species models

Table 2.2.2. Role of temperature as an active factor or a proxi in determining change in spatial distribution.

	MECHANISM	EFFECT ON DISTRIBUTION
Temperature = limiting factor	Level of action: individual , population , species interaction (diet increase) Metabolism : growth, spawning, survival (juv.winter, eggs larvae) Trigger : migration	-Change in habitat suitability -Timing of migration -Timing and location of spawning
Temperature = proxi for ecosystem change	-Oxygen as limiting factor -Plankton production under the control of oceanographic features: water column stratification, fronts, meso scale structures	-Change in habitat suitability -Recruitment success -Density dependence

2.2.2 Questions asked and analyses performed:

What is the mechanism for expansion in the spatial distribution? Does an observed increase in the species presence in the North Sea result from movements from southern populations or from pulses of North Sea relict populations that already had their life cycles in the North Sea?

Time series of an abundance index was estimated and abundance related to positive occupied area. Seasonality and life cycle spatial pattern were characterised by seasonal maps and seasonal length distribution. Length modes were tracked along successive surveys. Smallest length mode was used for picking a recruitment signal. This information was combined to evaluate whether the first observed colonisers in the 'new' areas were small (young) or large (old) fish and whether the species completed its entire life cycle in the North Sea or not.

DATRAS North Sea data was used in the first and third Quarters of the year (Q1 and Q3). Survey coverage of the North Sea prior to 1980 was somewhat less complete than for later years. Survey catchability of the selected species was unknown. Although information in the data could be coherently interpreted, complementary information would be necessary for confirming the understanding.

Trawl station numbers were averaged by ICES rectangle. The survey scheme being most regular, no spatial weighting was made (though some surveys had lower spatial coverage than others). The species abundance index was the data average by ICES rectangle by quarter, by year. The mean spatial distribution by quarter was estimated by taking the average of the log abundance by ICES rectangle across years. The mean length distribution by quarter was the average percent of fish numbers by length class across years.

Maps were characterised by their gravity centre and inertia (Woillez *et al.*, 2005). Inter-annual variability in spatial distribution was characterised by the Global Index of Colocation (GIC: Woillez *et al.*, 2005) which allowed for clustering years based on the GIC values used as distances between yearly maps.

The average length distribution by quarter revealed several modes. A visually defined length threshold was used to define small and large fish. Yearly length distributions were scrutinised and the presence/absence of large and small length modes was coded as 1/0. A length mode was considered to be present when the length frequency peaked with a frequency higher than 0.05.

2.2.3 Rationale for the selection of species and area

The southern North Sea experienced a regime shift in the late 1980s (Beaugrand, 2004). Some species with southern affinities were documented to have expanded in the North Sea (Beare *et al.*, 2004). Thus the North Sea was considered a laboratory where to study colonisation of habitats under climate change. Short lived species are more prone to respond quickly to changes in their environment and therefore were selected for the analysis. Colonisation was analysed for a pelagic species (anchovy) and a demersal species (red mullet). These species were reported to have expanded in the North Sea (Beare *et al.*, 2004). Sprat served as a reference species as its core biogeographical distribution range was the North Sea.

Anchovy is a short lived species with fast growth, high mortality and high fecundity (Alheit, 1989; Motos, 1996). Fecundity is undeterminate, meaning that spawning seasons and areas depend on current temperature and zooplankton production. Anchovy occupies coastal habitats under the influence of river discharge as well as marine habitats off-shore (at least in Biscay: ICES, 2006). Anchovy presents characteristics of an opportunistic species that makes it a candidate for analysing pelagic fish response to climate change. In the NE Atlantic the main population is located in the Bay of Biscay. Smaller populations exist, along the coast of Portugal, bay of Cadiz, and in the Mediterranean (ICES, 2006). Anchovy forms small schools

near the bottom during day time and usually attempts to escape trawls by going closer to the bottom. It can thus be captured by trawls like the GOV though its catchability is not known.

Red mullet shares similar aspects with anchovy but in the demersal domain. It is a short lived species with fast growth and rapidly acquired maturity at age-1 (Mahé *et al.*, 2005). In the North Sea it is at the northern limit of its biogeographical distribution range which extends to SW European Atlantic waters, the Mediterranean and Western Africa.

Sprat also shares very similar aspects with anchovy (Alheit, 1989). It is a short lived species with fast growth, high and undeterminate fecundity. It shares similar habitats than young anchovies, in particular coastal waters under the influence of river discharge (ICES, 2006). It is more constrained to these habitats than anchovy (in the Bay of Biscay at least, as observed with acoustic surveys). It is a more northern species than anchovy: the North Sea is in the latitudinal core of its biogeographical distribution range and the Bay of Biscay near its southern limit. It was selected with the idea that it could serve as a reference species for the time series of North Sea short lived species.

2.2.4 Anchovy in the North Sea

Results:

Figure 2.2.1: Mean numbers per hour and area occupied (positive area). A pulse of anchovy was observed in the mid 70s (see also Figure 2.1.8). Since 1995, although with high variation, anchovy has expanded in the North Sea. Abundance varied with area occupied.

Figure 2.2.2: Seasonal spatial pattern and length distribution. Anchovy showed a marked seasonal spatial pattern. Quarter 3 distribution is mainly in the southern North Sea, Skagerrak and Kattegat. Quarter 1 distribution is more to the North West along the UK coast (including Dogger Bank), Skagerrak and Kattegat. The average maps were well correlated with occurrence maps (not shown) meaning that high average values were not due to one year only. Length distributions showed 2 distinct peaks. In Q3 the first peak is centred around 10 cm and the second around 17cm while in Q1 they are centred around 12 cm and 18 cm. The length range 5 – 8 cm is in higher proportions in Q3 than in Q1. According to the growth pattern in Biscay, these small fish could be 4 months old, meaning that anchovy spawning could occur in April. It is more likely that these small fish in Q3 are in fact 1-year old and thus they were not available to the gear in Q1 (ICES, 2006).

Figure 2.2.3: Gravity centres, inertia and GIC. Interannual variability in spatial distribution is higher in Q1 than in Q3. Interannual variability is very high in Q1, with 4 major groups of spatial patterns. The spatial distribution is variable from one year to the other.

Figures 2.2.4and 2.2.5: The first re-occurrence of anchovy in the North Sea since the mid-1970s was detected in the 1994 Q3. In 1994 Q3 the length distribution showed one length mode only (12cm: corresponding to the younger fish) including a typical recruitment length range of small fish (5–8cm). In the subsequent survey in 1995 Q1, the SW North Sea was occupied. In 1995 Q1, this same length mode was observed again (12cm). The interpretation is either a recruitment pulse or a mixture between recruits and adults. The largest abundance peak in the series was observed in 2003 Q3 and following in 2004 Q1. A similar story occurred. In 2003 Q3 the length distribution was unimodal, the mode being in the length range of recruitment. This mode was observed again (slightly shifted) in 2004 Q1. In 2004 Q1, the entire coastal waters around the North Sea were occupied by anchovy. Periods of decrease in abundance are less easily interpretable using the present length data only.

Conclusions:

A seasonal variation in the spatial distribution was well marked and length modes were traced seasonally, which supported the evidence for an established life cycle in the North Sea. The

seasonal pattern could be (partly) explained by temperature as warmest waters are in the South in Q3 and in the North West in Q1 (see Figure 2.1.3). The waters occupied corresponded not only to warmer temperatures but also to areas under the influence of river discharges and/or tidal mixing (Dogger Bank) where productive fronts develop.

The first (re-)appearance of anchovy in significant numbers the North Sea since the mid-1970s occurred in the mid-1990s. Anchovy presence has been recurrent since. In the first re-appearance observation, small fish only was detected, meaning that North Sea habitats were colonised de novo by young fish exclusively.

Expansion of anchovy in the North Sea was found controlled by recruitment pulses. The hydro-climate and species interactions determining successful recruitment windows are unknown. The image of anchovy in the North Sea is that of a low abundance level resident population in the Southern North Sea that has recruitment invading pulses.

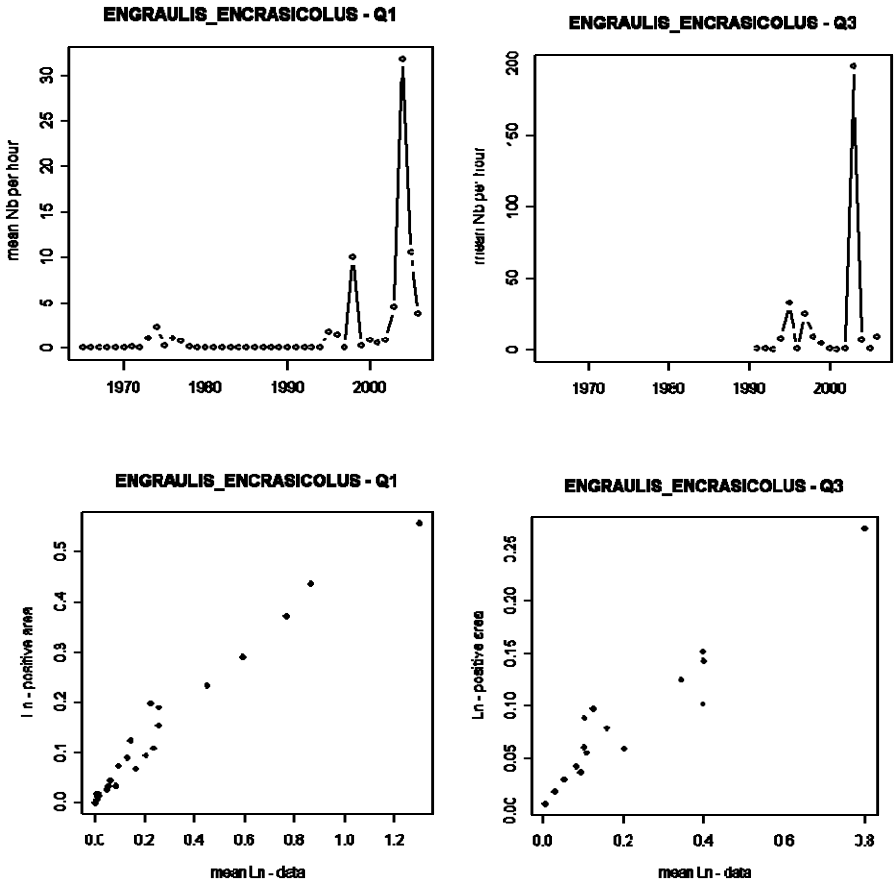


Figure 2.2.1. Abundance index time series and relation with area occupied.

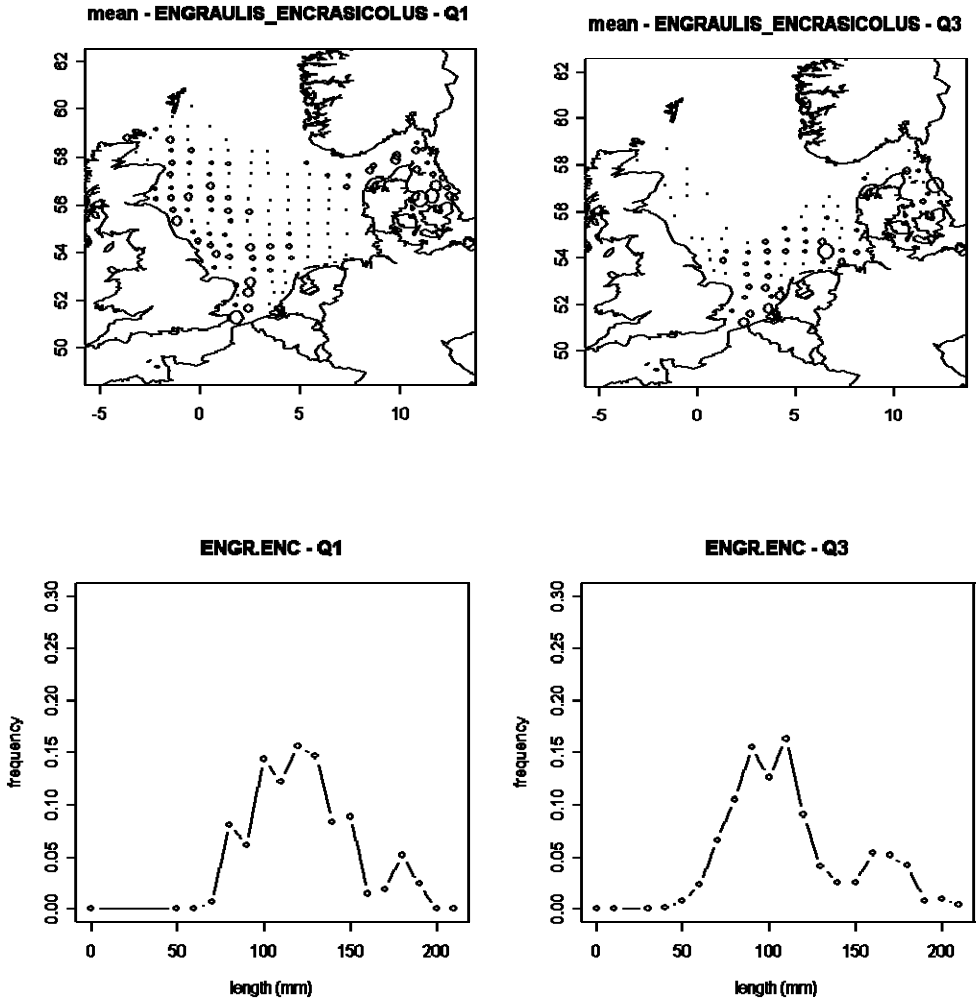


Figure 2.2.2. Mean seasonal spatial distribution and mean length distribution by quarter

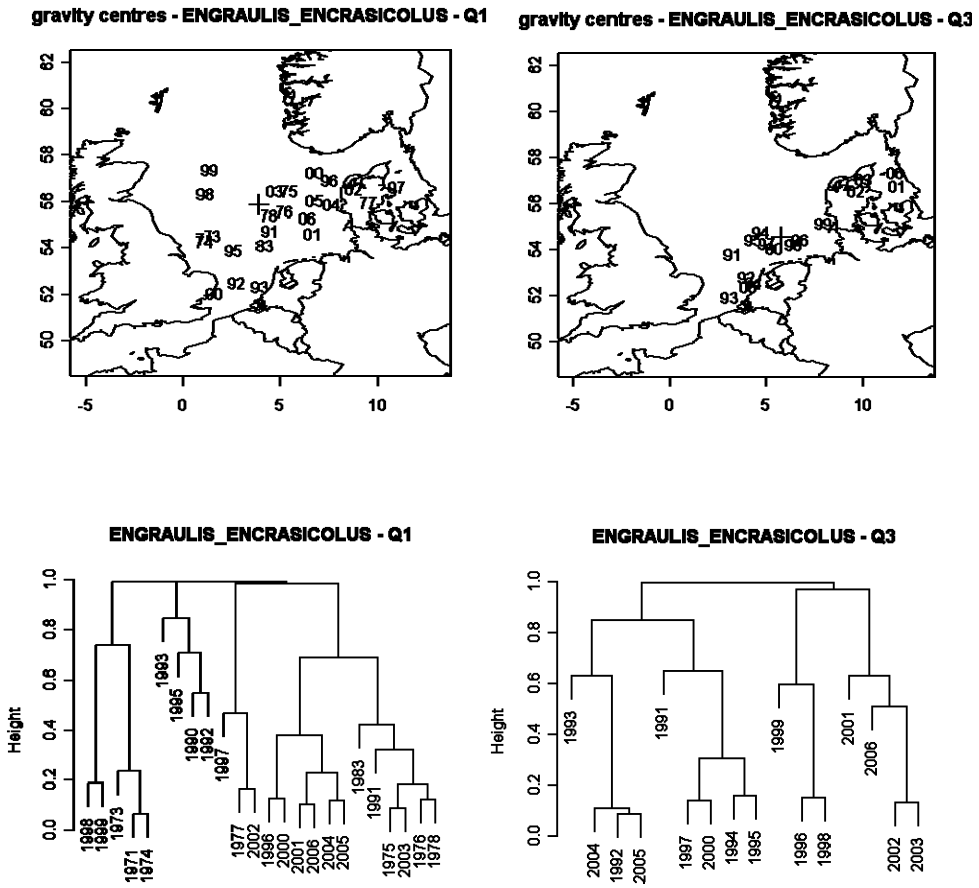


Figure2.2.3. Inter-annual variation in spatial distribution for Quarters 1 and 3 as revealed by gravity centres, inertia and global index of collocation.

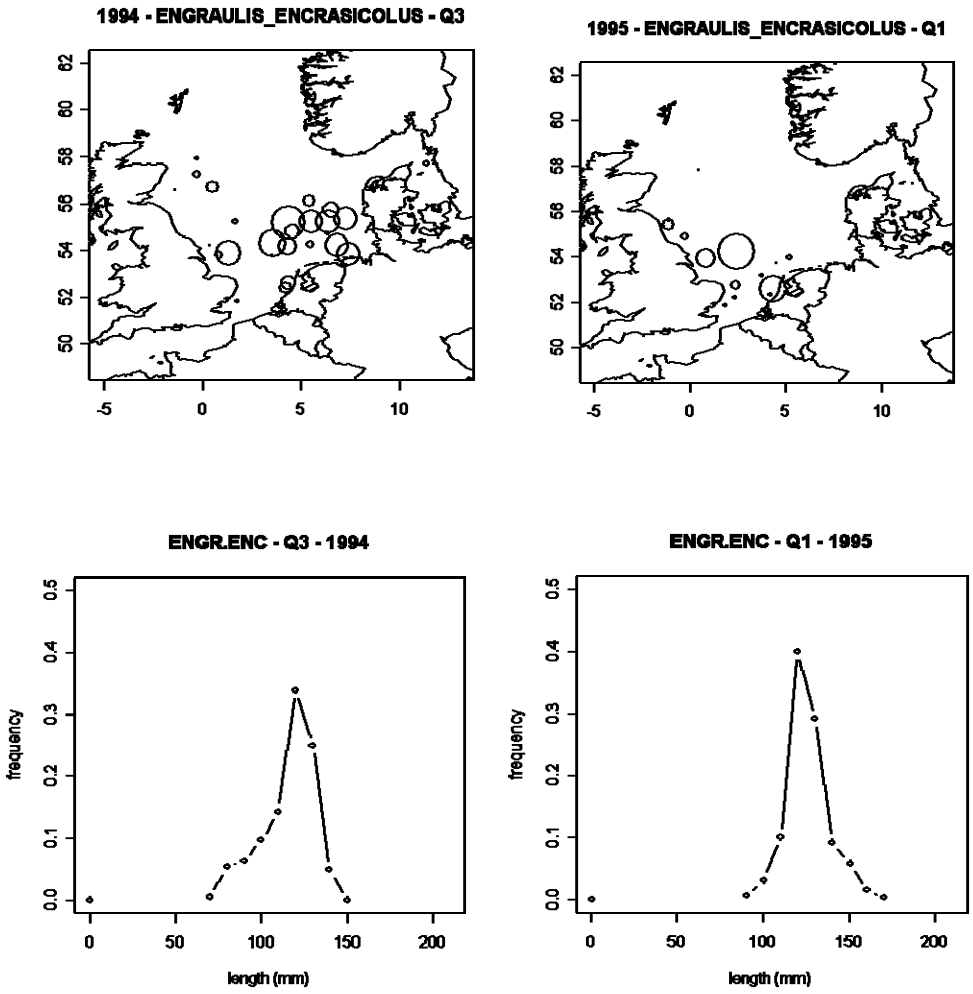


Figure 2.2.4. Spatial distribution of length distribution in the first survey of appearance since the mid-1970s and subsequent survey.

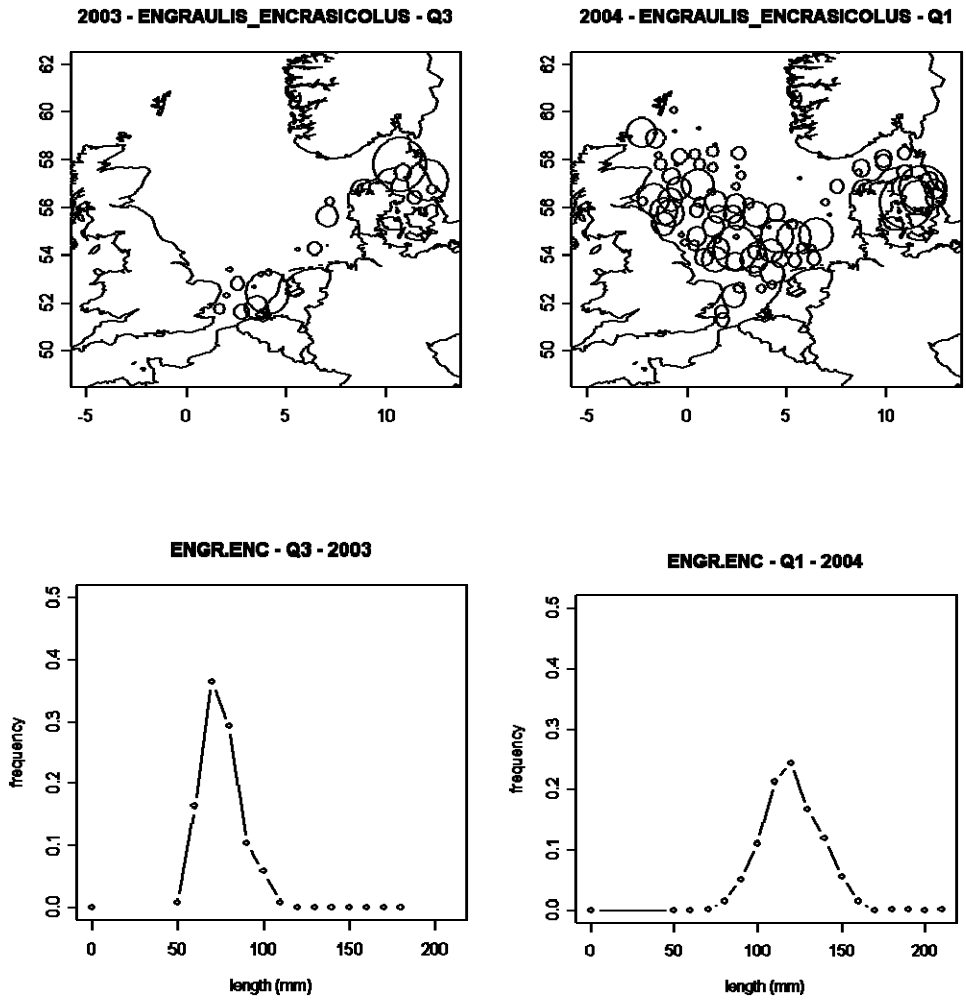


Figure 2.2.5. Spatial distribution and length distribution of anchovy for the first large peak in abundance

2.2.5 Red Mullet in the North Sea

Results:

Figure 2.2.6: Red mullet first appeared in the survey data in 1990 Q1 in low abundance. It has been recurrent since. Its first peak in abundance was in 1995 Q3, but coherence in the peak signals from one survey to the next is achieved at the second peak in 2003 Q3 and 2004 Q1. Abundance varied with area occupied.

Figure 2.2.7: Red mullet showed a strong seasonal pattern in its spatial distribution. The fish were located in the NW North Sea along the UK coast in Q1 and in Q3 close to the Belgium and Dutch coasts east of 5°E. The length distribution was bi-modal in both Q1 and Q3 with modes at 13 and 24 cm in Q1 and in Q3 at 10 and 20 cm. The length range 5–8 cm is typical of recruits appearing in the data in Q3.

Figure 2.2.8: Inter-annual variability between spatial distributions is larger for Q1 than for Q3. The spatial distribution in Q3 showed remarkable year-to-year consistency along the Belgium and Dutch coast. Apart from 1990 Q1 (date of first appearance of the species in the data), 2 major groups of spatial distributions can be identified in Q1.

Figure 2.2.9: The first appearance of Red mullet in the data occurred in 1990 Q1. The spatial distribution was particular, unlike any other years and closely connected to the English

Channel. The length distribution was unimodal (10cm), meaning that the newly colonised area was due to young fish only.

Figure 2.2.10: The first large peak in abundance occurred in 1995 Q3. Here, the length distribution was bimodal (12 cm and 19 cm) with the second mode more represented than the first one. But in 1996 Q1 abundance was low and the second length mode not observed, as if the large fish had left the North Sea and a small portion of the small fish had stayed in the North Sea and made the typical seasonal migration to the NW.

Figure 2.2.11: The second large peak in abundance occurred in 2003 Q3. Here again, the length distribution was bimodal with the first mode more represented than the second one. In 2004 Q1 abundance stayed high but the second mode was not observed. A similar story happened but this time recruitment was sufficiently strong and a large proportion of fish stayed in the North Sea during winter.

Conclusions:

The spatial distribution of red mullet in the North Sea was strongly related with warm waters (SW in Q3, NW in Q1). The Belgium and Dutch coasts are a red mullet hot spot in Q3. Small fish (10 cm) was observed to undergo this typical seasonal change. In contrast, large fish (20 cm mode) were seen in the North Sea in Q3 but rarely in Q1. In the first appearance of red mullet in the North Sea, small fish only were observed, meaning that the North Sea habitats were colonised de novo by young fish exclusively. Recruitment pulses ensured that significant numbers of small fish stayed in the North Sea. The image of red mullet in the North Sea is that of a population with its large fish related with the English Channel and the small fish resident in the North Sea. Expansion in the North Sea can then be due to large fish moving into the North Sea in Q3 combined with recruitment pulses that reside in the North Sea in Q1.

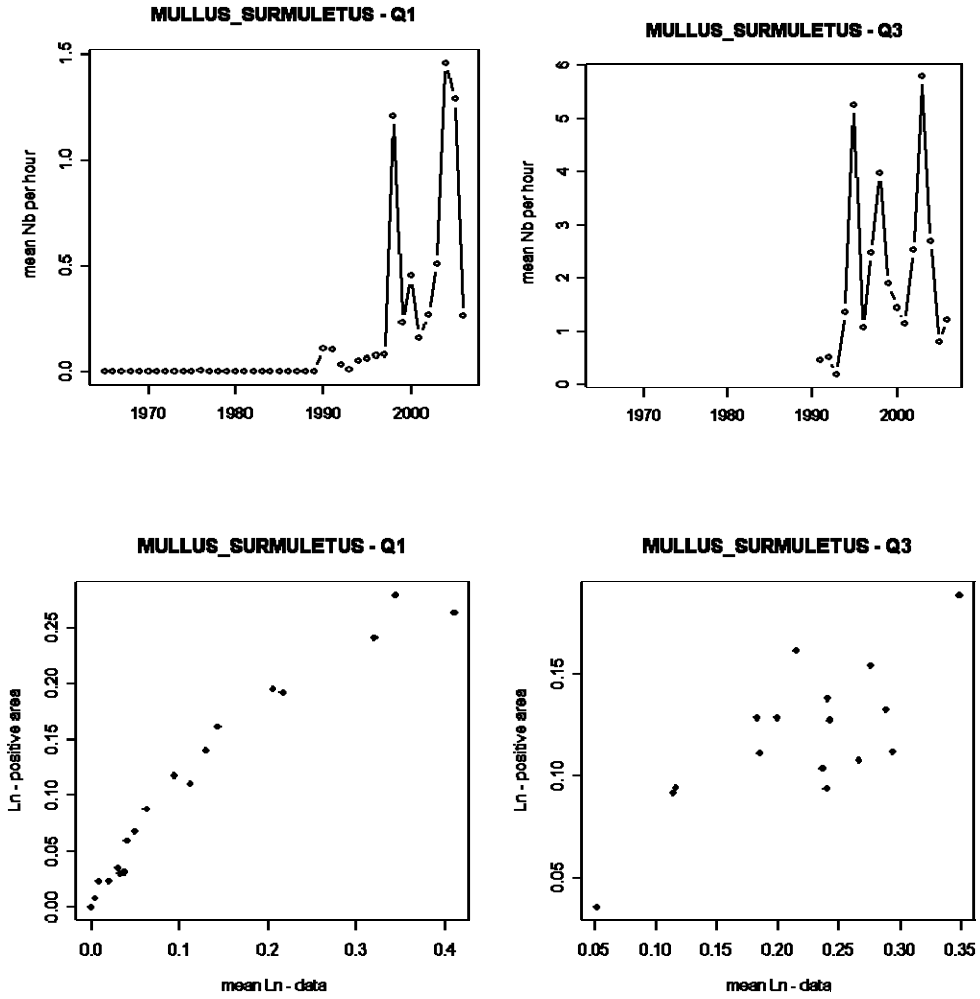


Figure 2.2.6. Abundance index time series of mullet and relation with area occupied.

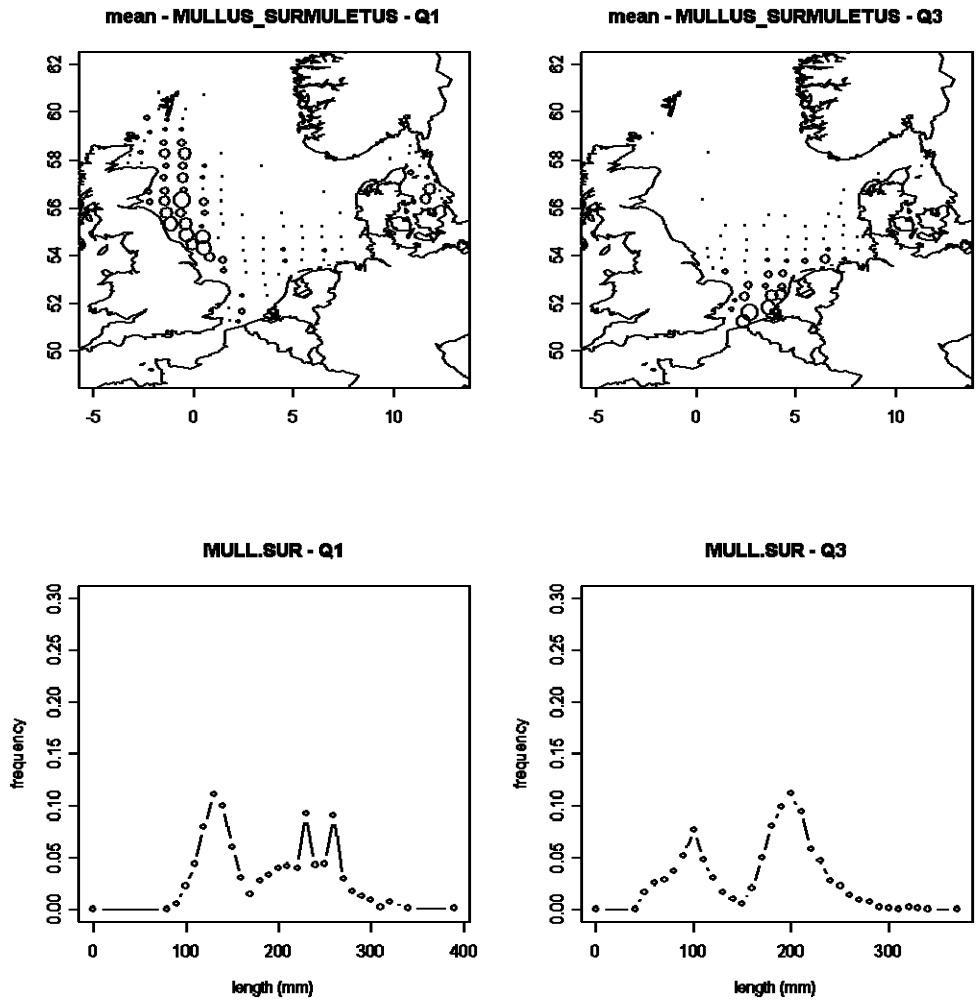


Figure 2.2.7. Mean seasonal spatial distribution and mean length distribution by quarter

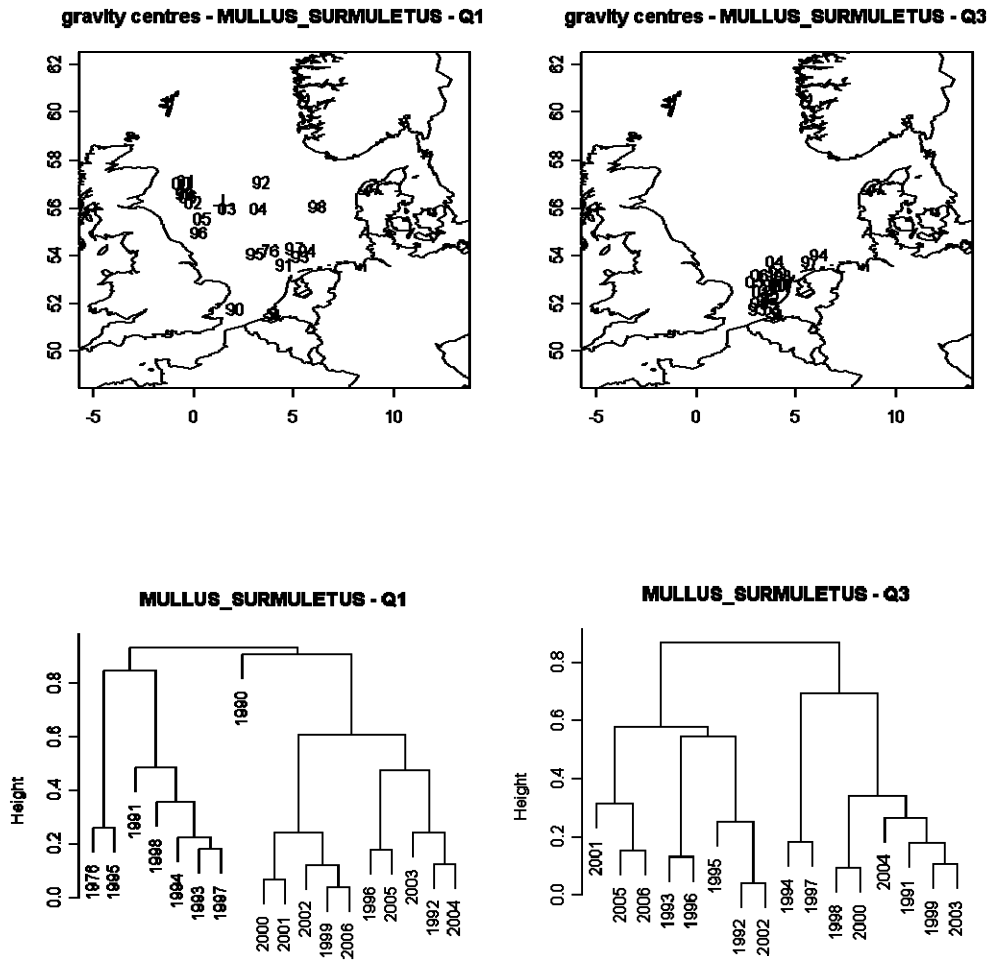


Figure 2.2.8. Inter-annual variation in spatial distribution for Quarters 1 and 3 as revealed by gravity centres, inertia and global index of collocation.

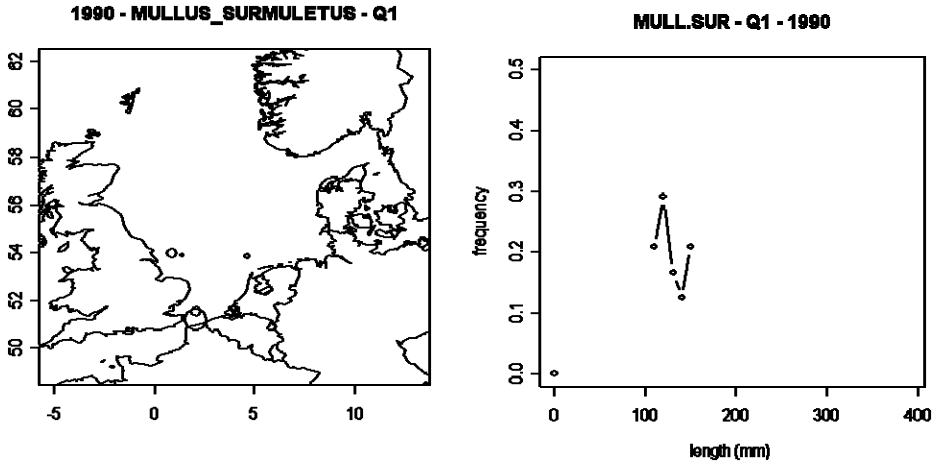


Figure 2.2.9. Spatial distribution and length distribution in the first survey of appearance.

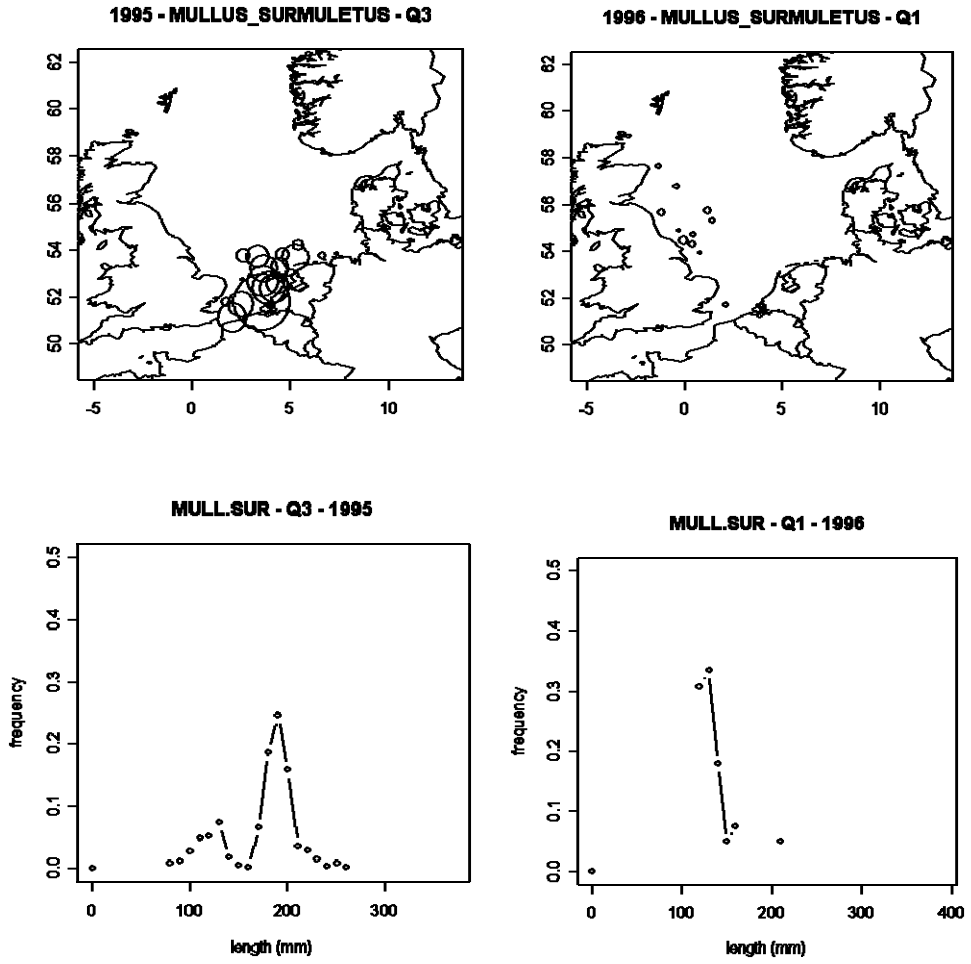


Figure 2.2.10. Spatial distribution and length distribution in the first large abundance peak.

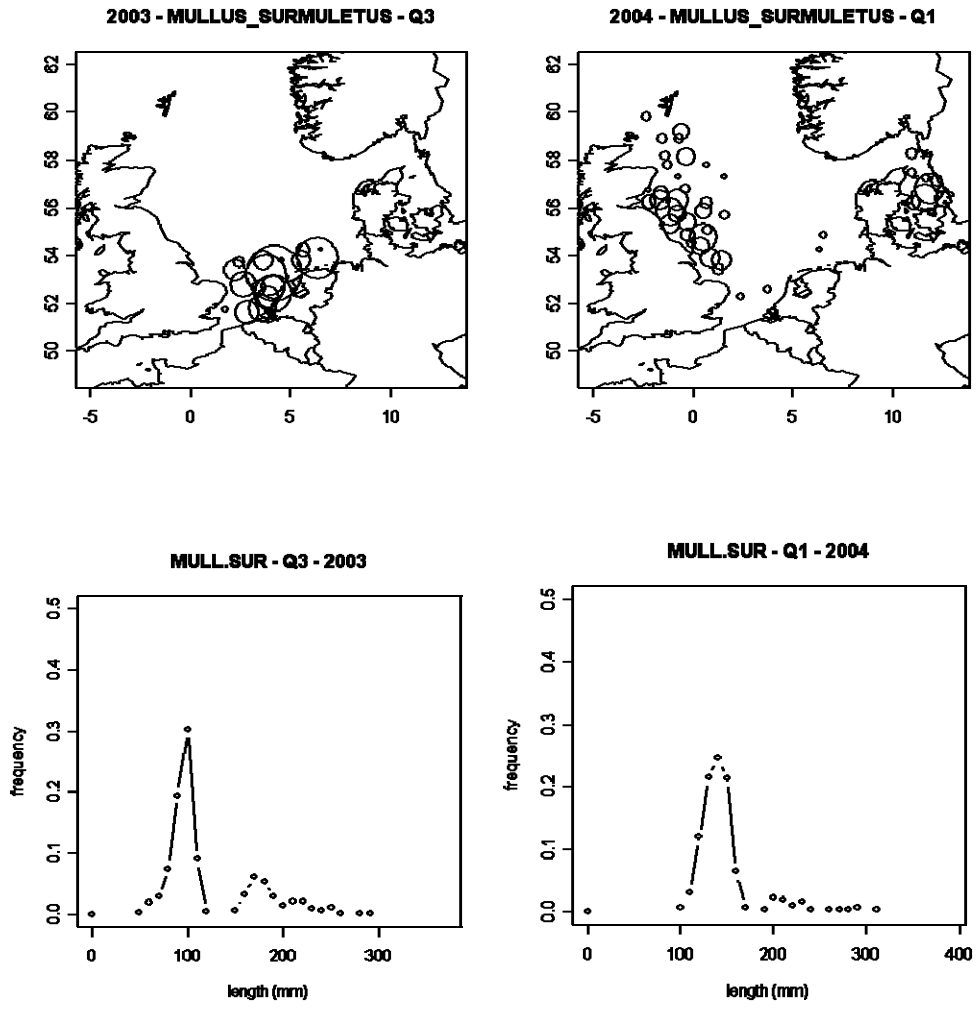


Figure 2.2.11. Spatial distributions and length distributions for the second large abundance peak for which coherence is observed in abundance and length between the Q3 survey and the subsequent Q1 survey.

2.2.6 North Sea sprat

Results

Figure 2.2.12: The species has been present all through the survey time series. The abundance index showed no long-term trend visually but with interannual variability. A series of successive low abundance numbers were observed for the years 1982–1986. Abundance and occupied area were related.

Figure 2.2.13: There was no obvious seasonal change in the mean spatial distribution between Q1 and Q3. Sprat occupied all the coastal areas along the UK, Belgium, Holland and Germany and in the Skagerrak and Kattegat in both seasons. Temperature did not seem therefore to control the spatial distribution. These areas are also known to be under the influence of river discharge and tidal mixing where productive fronts develop. Similar length modes were present in both seasons, although in Q1 fish were smaller than in Q3.

Visual inspection of length distributions in Q1 and Q3 by year showed that fish with length < 10 cm were present in Q1 for all years but were more sporadically present in Q3 across years. Presence of these small fish in the Q3 surveys happened in 1991–1994, 1999, 2003–2004. In these years, additional recruitment windows may have happened.

Figure 2.2.14: Interannual variability between maps is larger in Q1 than in Q3. Variability relates to gravity centres more to the SW or the NE in Q1 and Q3, and to the NW in Q1.

Figure 2.2.15: In 1995 Q1 a large abundance peak occurred. The figure shows the sequence of observations for 1994 and 1995: in 1994 Q1 a small length mode only; in 1994 Q3, two length modes meaning that a recruitment wave has arrived; in 1995 Q1 again two length modes meaning that another recruitment wave has arrived; in 1995 Q3 one larger length mode only (13 cm) meaning that no recruitment wave arrived in that year.

Conclusions

Sprat showed consistent spatial distribution over time. The recruitment signal was captured in Q1 in all years. Additional recruitment signals were observed in Q3 in particular years 1991–1994, 1999, 2003–2004.

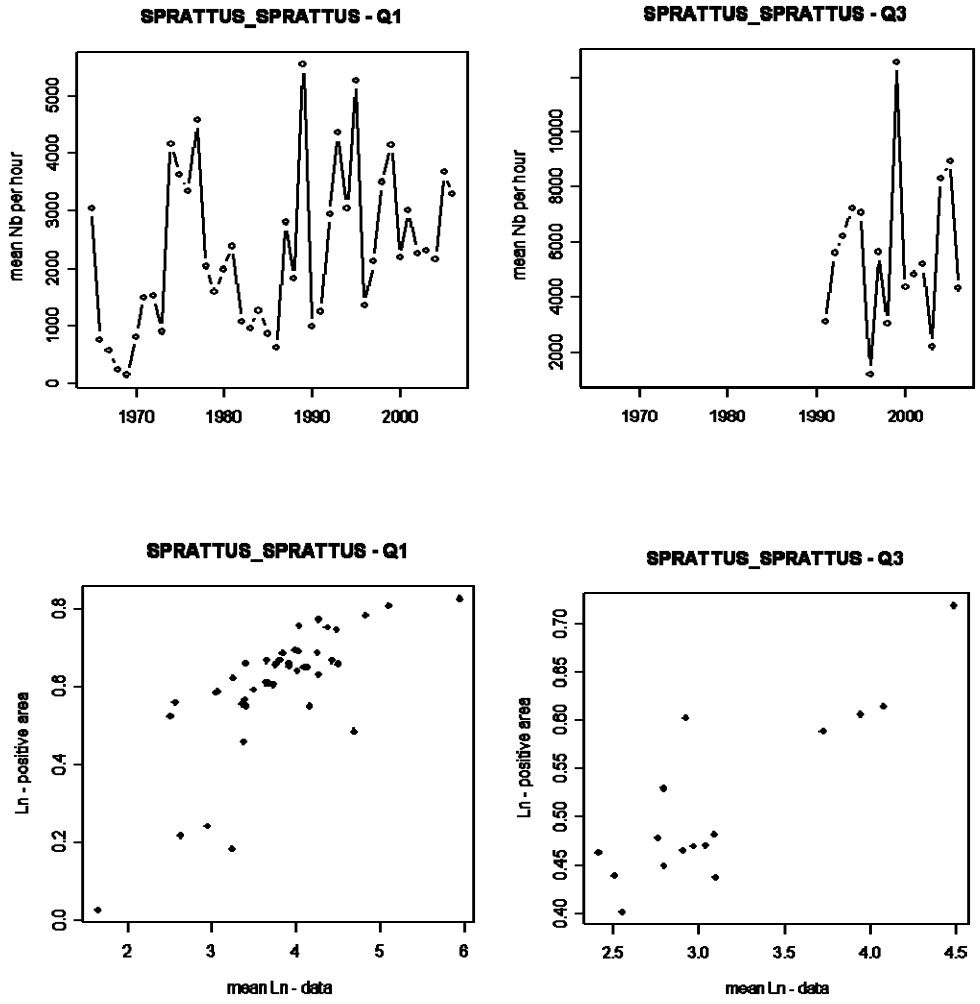


Figure 2.2.12. Abundance index times series and relation with occupied area.

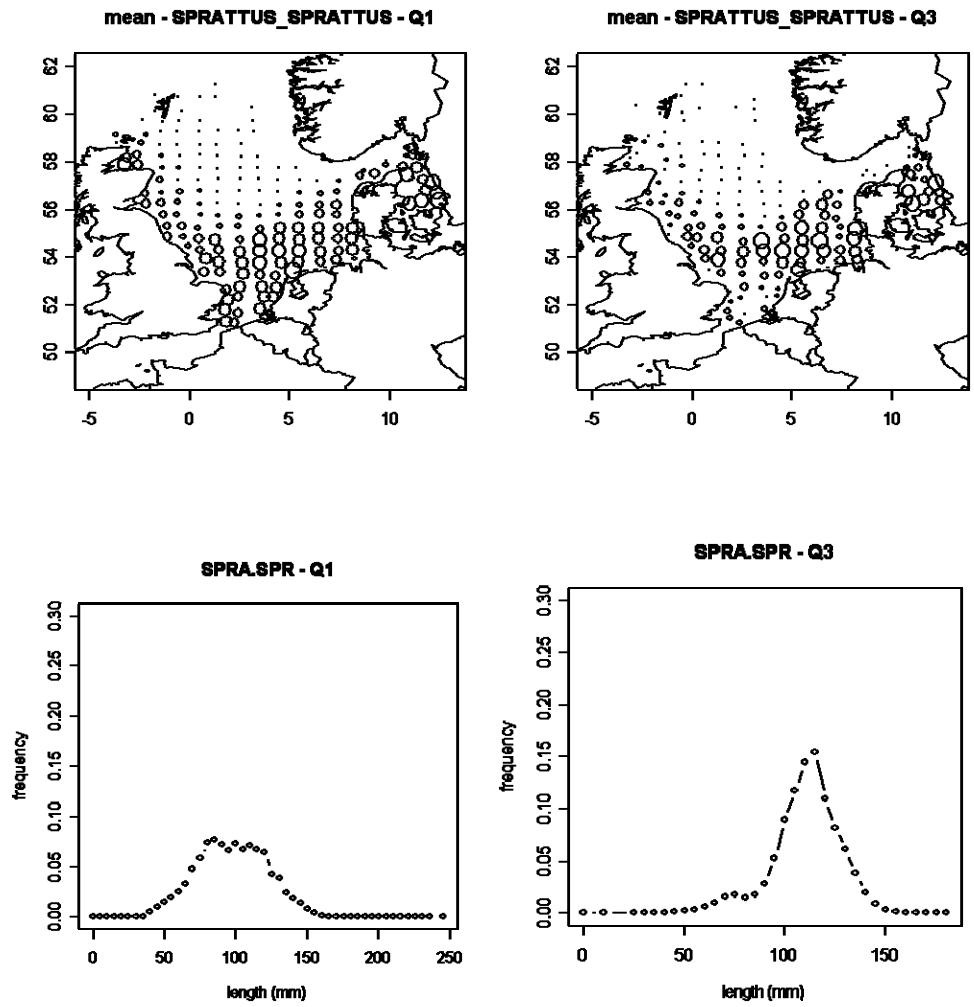


Figure 2.2.13. Mean seasonal spatial distribution and mean length distribution by quarter.

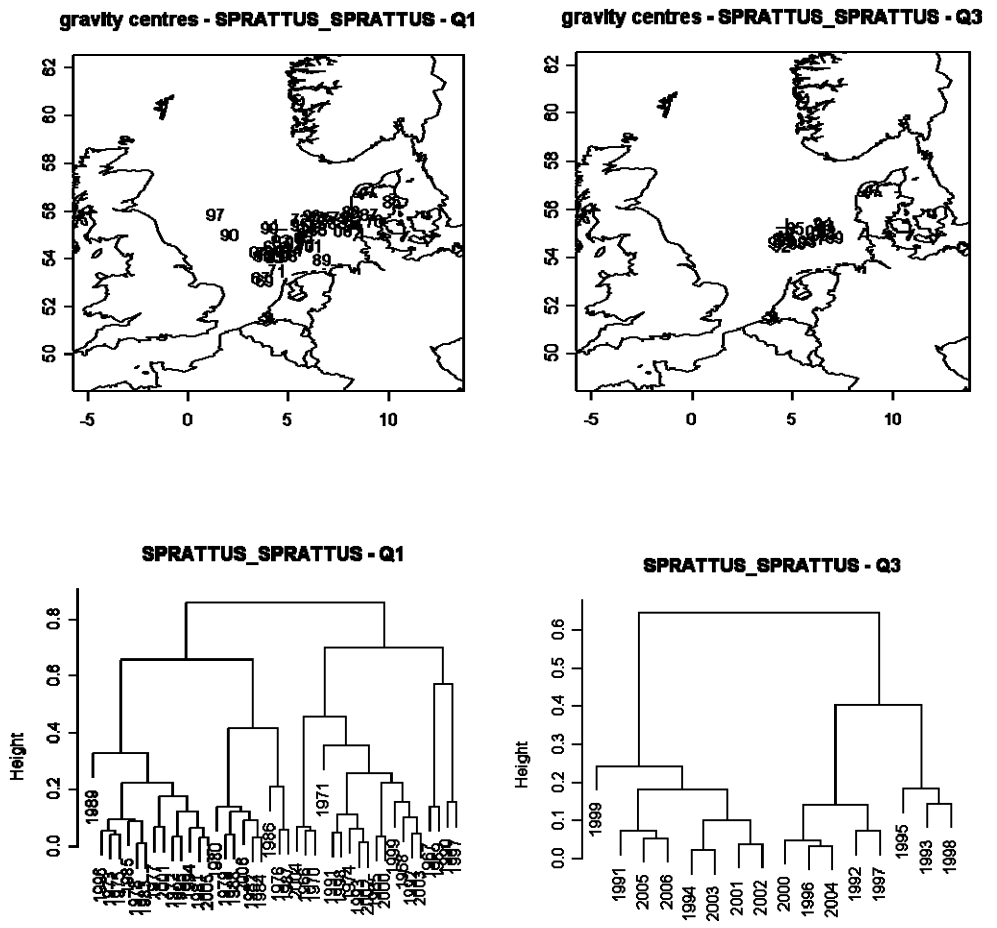


Figure 2.2.14. Inter-annual variation in spatial distribution for Quarters Q1 and Q3 as revealed by gravity centres, inertia and global index of collocation.

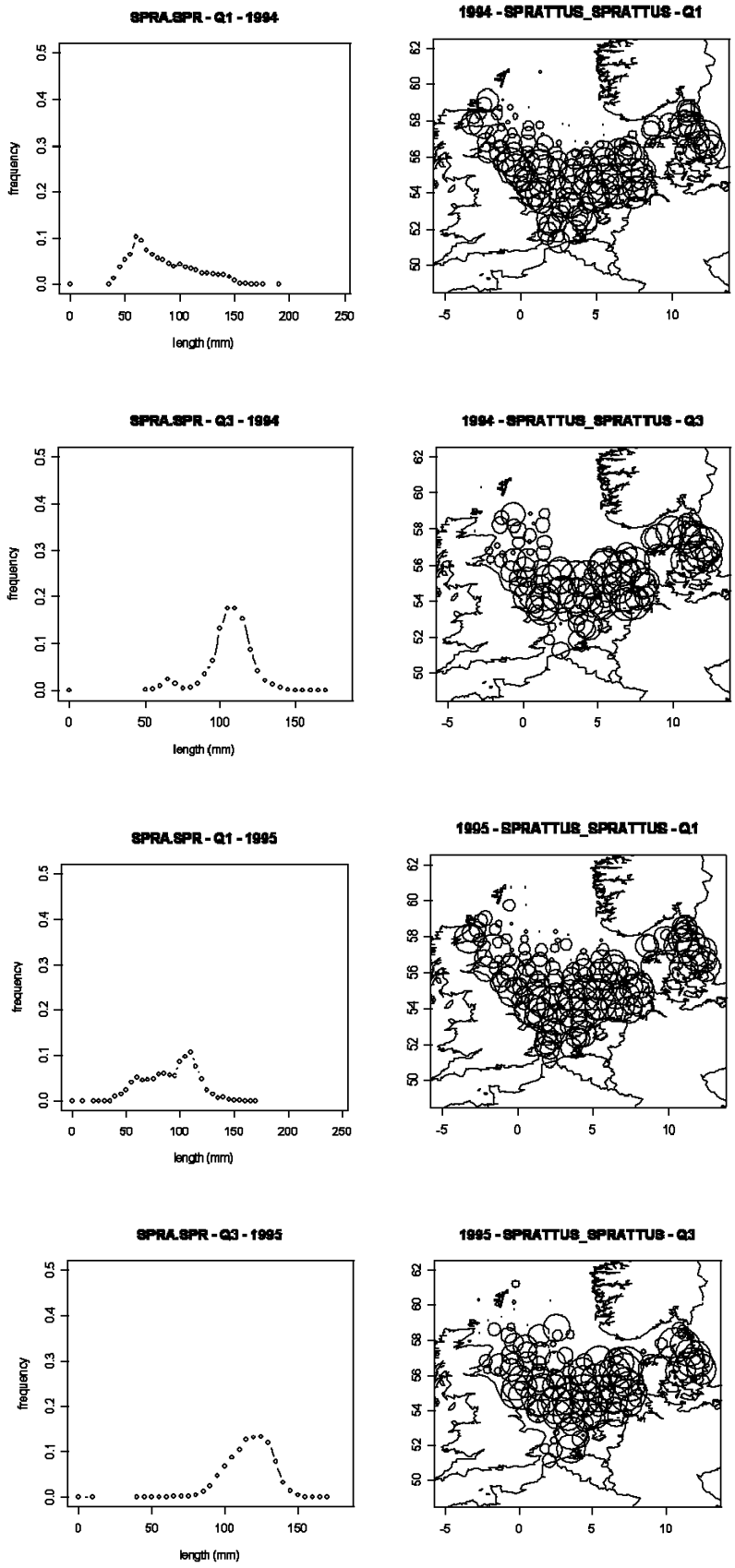


Figure 2.2.15. Illustrative variation of the small fish mode (recruitment signal) across Quarters

2.3 Changes in the potential spawning habitat of anchovy 1977–2005

Based on multiannual (2000–2005) records of anchovy eggs and hydrographic conditions in the Bay of Biscay and around the Iberian Peninsula, the conditions potentially suitable for anchovy spawning are modelled.

The modelling technique is derived from ICES/SGRESP06 and is based on quantile regression smoothers. The technique provides estimates of the potential spawning maximum under given environmental conditions. Surface temperature, surface salinity and log (Bottom depth) were considered as environmental controls. Individual quantile models for each variable are shown in Fig 2.3.1.

Using the model fitted to observations south of 49N, it is possible to predict the potential for spawning of anchovy in other regions given known surface temperature, surface salinity and bottom depth. The model does not predict spawning but predicts whether the environmental conditions are limiting or not for spawning.

Using the ICES Hydrology database, we have plotted the modelled potential for spawning of anchovy during the 2nd and 3rd quarter (April to September) for three periods which correspond to three temperature regimes in the North Sea: 1977–1988, 1989–1998, 1999–2005 (Figure 2.3.2).

Although caution must be taken when interpreting the spatial distribution constructed, it appears that during the first period (1977–1988), the conditions suitable for spawning in the North Sea (in terms of temperature, salinity and depth) were restricted to the south-eastern part, i.e. in the northwest, favourable conditions for spawning were never recorded. During the second period (1989–1998) the area of potential suitable spawning habitat extended further West and further North. This pattern seems to have continued in recent year when it appears that non-limiting conditions for spawning have been recorded in nearly every area of the North Sea.

The preliminary results above would need to be complemented by potential habitat models based on re-analysis of temperature and salinity fields over regular grids and with fine temporal resolution (e.g. week) for the period 1977–2005 or longer.

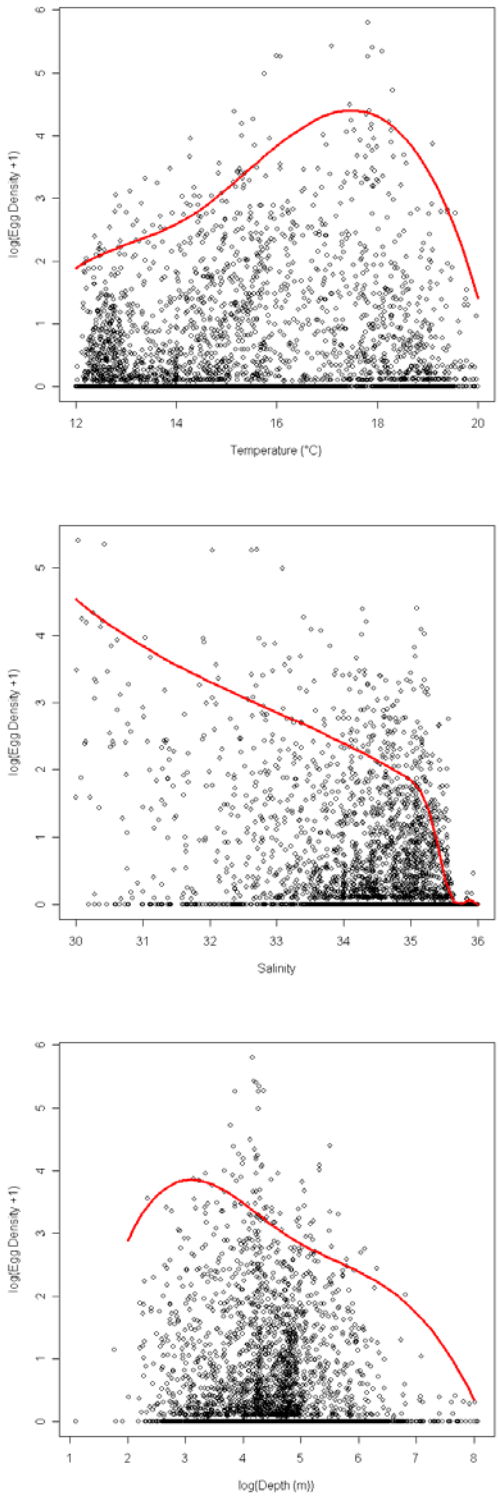


Figure 2.3.1. Modelled 95% quantiles of anchovy egg abundance as a function of surface temperature, surface salinity and log(Bottom Depth). The models are fitted to data collected from southern Spain to the West of Brittany for the period 2000–2005. Egg data were collected with the Continuous Underway Fish Egg Sampler (CUFES) by IPIMAR, IEO and Ifremer institutes.

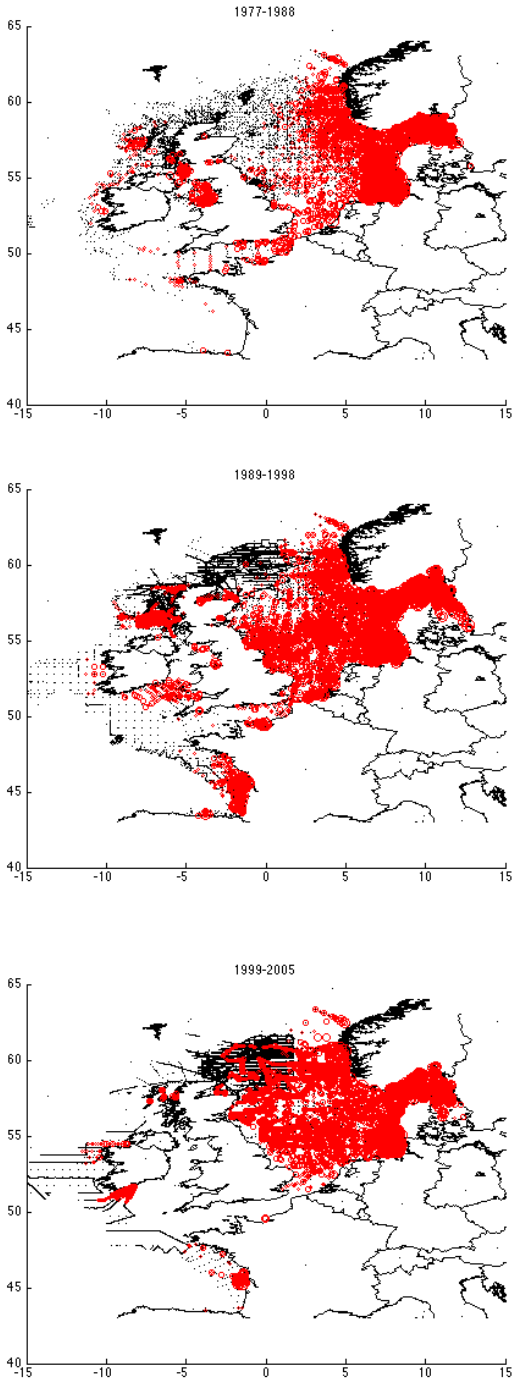


Figure 2.3.2. The spatial distribution of potential for spawning of anchovy estimated from ICES hydrography database records for the periods 1977–1988, 1989–1998, 1999–2005. Black dots indicate low spawning potential (<6.4 egg/10m^3). Red circles indicate higher potential and circle area is proportional to potential values (in egg/10m^3).

2.4 Changes in fish condition in relation to changes in marine environmental variables

Previous studies have found a positive relationship between fish condition and water temperature. Rätz and Lloret (2003) examined ten cod stocks in the North Atlantic and found a significant correlation between mean bottom temperature and Fulton's condition index ($K=100*W/L^3$). Stocks in warmer waters were in better condition, with K rising by approximately 0.02 for every 1ºC temperature increase.

To assess the effects of changing sea temperature on fish condition in the ICES area, the working group used individual weight/length measurement from the DATRAS database to calculate Fulton's condition index. Individual fish weights were available only for the North Sea in the years 2003 to 2005. Over this short period, insufficient variation was observed to allow analysis of the relationship to environmental variables (Fig 2.4.1). It is known that several countries have large numbers of individual weight/length data in national databases and these could be used in future studies to examine variation in fish condition over a longer time period and with a greater degree of spatial and temporal resolution.

The working group also considered weight at age data, taken from the WWGNSSK 2006 report, to determine whether these could be used as an indicator of condition. However, it was found that a strong year-class effect, resulting from variation in growth rates, accounted for much of the variation in age at weight. Age at weight was therefore considered inappropriate as a proxy for condition.

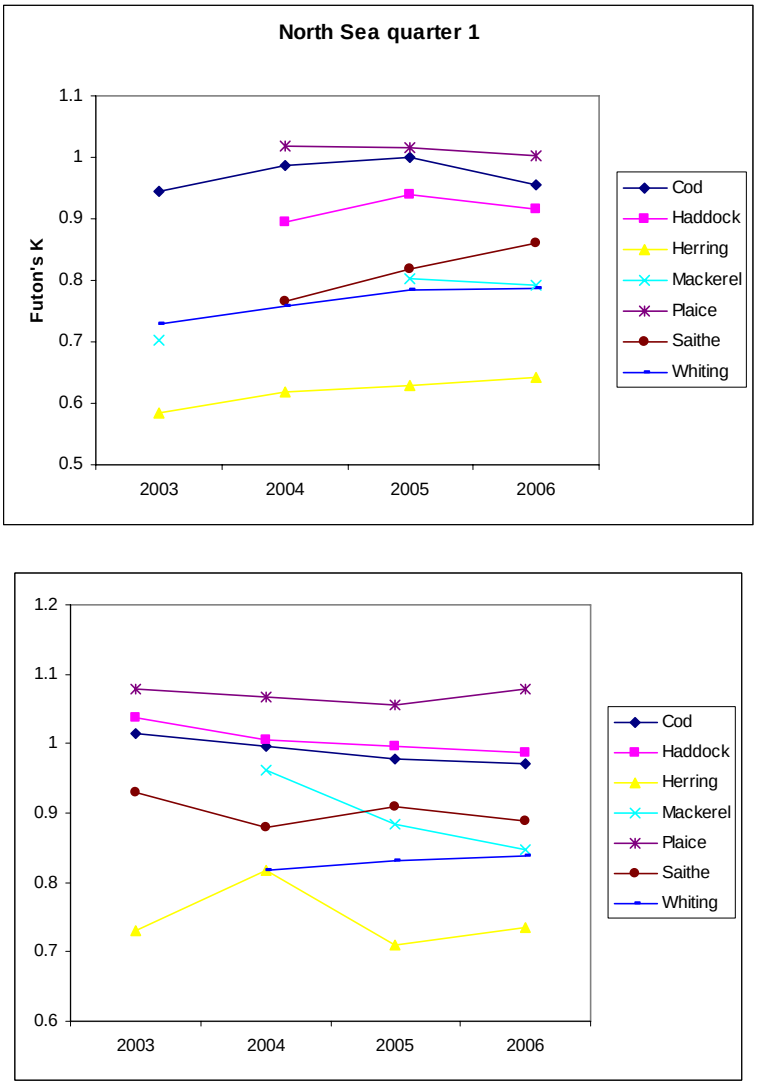


Figure 2.4.1. Change in fish condition (Fulton's K) 2003 to 2006 for 7 species of fish in North Sea IBTS surveys.

2.5 Temporal changes in plankton abundance

During the last decades, as the temperature regime has changed in the North Sea, important changes have also taken place in the ecosystem and in the plankton community in particular.

Figure 2.5.1 shows historical changes in the seasonal patterns of selected plankton indicators derived from the Continuous Plankton Recorder survey.

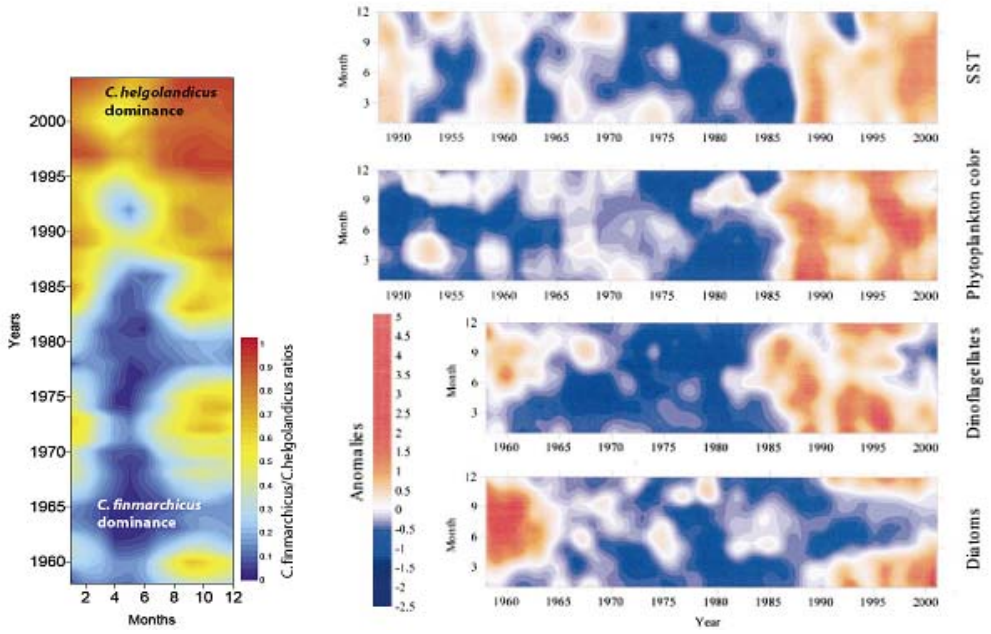


Figure 2.5.1. Left: changes in the ratio of *Calanus helgolandicus* over *Calanus finmarchicus* (from Edwards, M., Johns, D.G., Licandro, P., John, A.W.G. and Stevens, D. P. 2006. Ecological Status Report: results from the CPR survey 2004/2005. SAHFOS Technical Report, 3: 1–8. ISSN 1744-0750). Right: changes in Phytoplankton colour index, total dinoflagellates and total diatoms recorded by the CPR survey (from Edwards, M., Johns, D.G., Leterme, S.C., Svendsen, E. and Richardson, A.J., 2006. Regional climate change and harmful algal blooms in the northeast Atlantic. Limnol. Oceanogr. 51(2), 820–829).

2.6 A preliminary comparative analysis of interannual variation in North Sea fish distributions, temperature and hydrography

2.6.1 Introduction

Climate change has led to marked changes in environmental conditions in the North Sea over the last century (Cushing, 1982). Sea surface temperatures have been rising by 0.2 – 0.6 ºC per decade over the past 30 years in the North Atlantic and UK coastal waters and warming is greatest within the English Channel and North Sea (Marsh and Kent, 2006).

Climate change is predicted to shift plant and animal distributions northward (Hickling *et al.*, 2006; Parmesan and Yohe, 2003; Walther *et al.*, 2002). Some of most marked systematic changes have been documented in the distributions of fish populations. In the North Sea species with southerly range boundaries have retracted northwards and species with northerly range boundaries have expanded northwards as temperatures have warmed (Perry *et al.*, 2005). Waves of exotic fishes with southerly biogeographical affinities are invading the North Sea, including anchovy, red mullet, pilchard, John Dory and snake pipefish (Beare *et al.*, 2004; ICES, 2006; Quero, 1998). Other species have increasing in abundance in the North Sea since 1925, including: red and tub Gurnards, bib, poor cod and lesser weaver (Beare *et al.*, 2004). Climate induced movement may lead to mismatches between and predator and prey populations, changes in recruitment success, growth performance, population reductions and

local extinctions (Attrill and Power, 2002; Drinkwater, 2005; Kell *et al.*, 2005; Pörtner and Knust, 2007; Sims *et al.*, 2001).

The dominant mode of climatic variability in the North East Atlantic region is the North Atlantic Oscillation (NAO), which is defined as atmospheric pressure difference between Iceland and Gibraltar (Hurrell, 1995). The NAO is a key indicator of environmental variability in the North Atlantic and is associated with variation in atmospheric pressure, sea surface temperatures (SSTs), salinity and wind-driven turbulence and advective processes. For example, NAO-driven westerly winds influence the strength of the North Atlantic Current (NAC) which in turn drives inflows of warmer Atlantic water into the Northern North Sea (Pingree, 2005). In general, positive NAO years are associated with warmer SSTs and southerly winds, compared to negative NAO years which have colder SSTs and northerly winds (Stenseth *et al.*, 2005). However, recently these NAO-climate relationships have been found to be much more complex, and less tightly coupled to environmental conditions than previously thought (Polyakova *et al.*, 2006). There has been an eastward shift in the sea level pressure pattern that was previously associated with interannual variability of the NAO. This may be a consequence of the trend towards higher NAO index during the last several decades of the 20th century (Peterson 2003).

Not only are the physical NAO-climate relationships variable in strength but environment-ecological response relationships are notoriously variable and often fade with time (Myers, 1998). Notwithstanding the problems documented in Myers (1998), numerous studies have linked variability in the North Atlantic oscillation to changes in recruitment, growth and distribution of fishes and their zooplankton prey. Climatic forcing by the NAO is consistently the most important parameter explaining variation in assemblage composition, abundance and growth of estuarine-dwelling juvenile marine fish (Attrill and Power, 2002). Juvenile cod in the North Sea shifted northwards following warm winters and southerly winds associated with positive NAO years (Rindorf and Lewy, 2006). NAO-linked wind-driven changes in the strength of Atlantic inflow have been linked to zooplankton dynamics in the Northern North Sea. Since 1987 increased inflow corresponds to the regime shift in zooplankton from domination by the boreal *Calanus finmarchicus* to the southern *C. helgolandicus* (Reid *et al.*, 2003). Consequently many fish populations in the North Sea are shifting latitudinal and depth distribution, particularly those with range boundaries centred on this area (Perry *et al.*, 2005). Only about half (15/36) of the populations considered in this study exhibited a response to climate variability. Within those populations responding to warming, smaller-bodied species tended to exhibit greater shifts in distribution compared to larger-bodied species (Perry *et al.*, 2005). This work sought to identify general climate-distribution links using a composite climate variable comprised of 5-year running means of temperature, NAO, Gulf Stream index and the ratio of abundances of northern and southern calanoid copepod species (Perry *et al.*, 2005).

The longer-term goal of this work is to generate a biological index of the effect of climate variability and change on North Sea fish distribution, for the shorter-term the purpose of this work is to determine (i) which North Sea species shift geographic distribution in response to climate variability (temperature and hydrography) and (ii) which measures of climate variability are most tightly linked to changing fish distributions.

2.6.2 Methods

We used the North Sea English groundfish survey data to assess changes in fish abundance and distribution. Currently, a survey grid of up to 112 statistical rectangles, with an average depth <200m, are fished annually throughout the North Sea. Not all stations in the survey grid are fished every year due to poor weather, equipment damage or ship failure, and in the earlier surveys more stations were sometimes surveyed (for more details see Maxwell and Jennings, 2005). Only 18 stations have been fished in each year. Stations were fished with a Granton

demersal trawl until 1991, but from 1992 a Grand Ouverture Verticale (GOV) demersal trawl was used. Tow duration was 60 min. up to 1991, from 1992 onwards the tow duration was 30 min (B. Harley, Cefas, Lowestoft, Suffolk, UK pers. comm.). The Granton trawl gear was fitted with a cod-end liner of 14 mm stretched mesh and the GOV trawl was fitted with a cod-end liner of 20 mm stretched mesh. Both gears were towed at a speed of approximately 4 knots. All fishes caught were identified and measured. Catch rates were raised to number of individuals caught per 60 min tow.

> Species were included in this analysis provide they were reliably identified and were consistent catchable by the gear (Dulvy *et al.*, 2006; Knijn *et al.*, 1993; Maxwell and Jennings, 2005; Sparholt, 1990). The thirty-five species retained for analysis were representative of the breadth of morphology, life histories, ecology and taxonomic diversity of the bottom-dwelling and pelagic fishes sampled by the English groundfish survey in the North Sea.

Species distributions were described using seven measures: average latitude, maximum and minimum latitudes, mean and maximum depth and occupancy. Latitude and depth measures were estimated as the centre of gravity. The centre of gravity of depth or latitudinal distribution was estimated over statistical rectangle in which the species was present as the sum of average depth or latitude in each rectangle weighted by the natural log of the average catch (Rindorf and Lewy, 2006). Occupancy was defined and measured and the proportion of EGFS rectangles occupied in each year (Webb *et al.*, 2007). The biogeographic affinities of each species were derived from the primary literature (Yang, 1982). The mean temperature and temperature range experienced by each species was estimated from the mean temperature in each ICES rectangle occupied by each species in each year. 'Warm' and 'cold' species were defined as those with the warmest and coldest mean temperatures based on the rank order of the mean temperature experience by each species. Thermal "specialists" and "generalists" were defined as species with narrowest and widest temperature ranges respectively (Table 2.6.1).

Six indices of climatic variation were considered; including average sea bottom temperature across the North Sea, sea bottom temperature anomalies in each rectangle, average annual North Atlantic Oscillation index (NAO), winter NAO index, North Sea inflow data and the Gulf Stream Index. Sea bottom temperatures were provided by the ICES Oceanographic database (www.ices.dk/ocean/). All temperature measurements were used and averaged for each rectangle in each year. The North Atlantic Oscillation index is the normalised sea level pressures difference between Gibraltar and Iceland and was derived from the Climate Research Unit of University of East Anglia (www.cru.uea.ac.uk/cru/data/ (Jones *et al.*, 1997). Both annual and winter NAO averages were considered. The Gulf Stream Index measures the relative position of the northern wall of the Gulf Stream along the east coast of North America (web.pml.ac.uk/gulfstream/data.htm).

The relationship between year-by-year climate variation and species distribution was assessed by fitting linear models using robust regression (Fox, 1997; Venables and Ripley, 2002) .

2.6.3 Results

2.6.3.1 Species thermal affinities

Most species were generalists with 12 and 15 cold and warm generalist species and 5 and 3 cold specialists and warm specialists respectively (Table 2.6.1). The cold specialists were wolfish, silvery pout, witch, anglerfish and spurdog and the warm specialists were megrim, cuckoo ray and lesser spotted dogfish.

2.6.3.2 Species climate variation – biogeographic distribution

There were greater than expected significant climate-biogeography relationships – $n = 5$, only approximately one significant test would be expected from 1470 tests with an alpha of 0.001. Three species with strongest relationships were herring, wolfish and Norway pout – all exhibit boreal cold temperature distributions with the herring more generalist tolerating a wider temperature range than the other two which are classified here as specialists (Table 2.6.1). Herring was found at higher latitudes and greater depth in warmer years and in response to local warming (i.e. positive rectangle anomalies; Figure 2.6.1). The southern range boundary of wolfish retracted northwards most strongly in response to local warming ($p = 0.0005$; Figure 2.6.2), though there is little change in average and maximum latitude. The Norway pout depth range was shallower years of positive Gulf Stream Index values ($p = 0.0003$).

The criteria for an ecologically significant climate – distribution response was relaxed and 22 of the 36 species exhibited at least one significant relationship at alpha = 0.01. The most responsive species were the herring and the lesser weaver which exhibited seven and four significant responses, respectively. In warmer years herring is more widespread, occupying a greater proportion of the survey area and occurs at greater average depth and more northerly average latitudes. The lesser weaver is also more widespread and occurs at more northerly latitudes, but in contrast to the herring, the lesser weaver has a shallower average depth in warmer years (Figure 2.6.3). Wolfish, grey gurnard, whiting, redfish and sole each exhibited two significant relationships, and the remaining 15 species exhibited at one significant climate – distribution relationship between.

Temperature, temperature anomalies, and to a lesser degree the annual average NAO index, appeared to be most often related to the interannual variation in biogeographic distribution compared to Gulf Stream Index and North Sea inflows. The most responsive biogeographic distribution measures were average depth, average latitude, maximum latitude and minimum latitude.

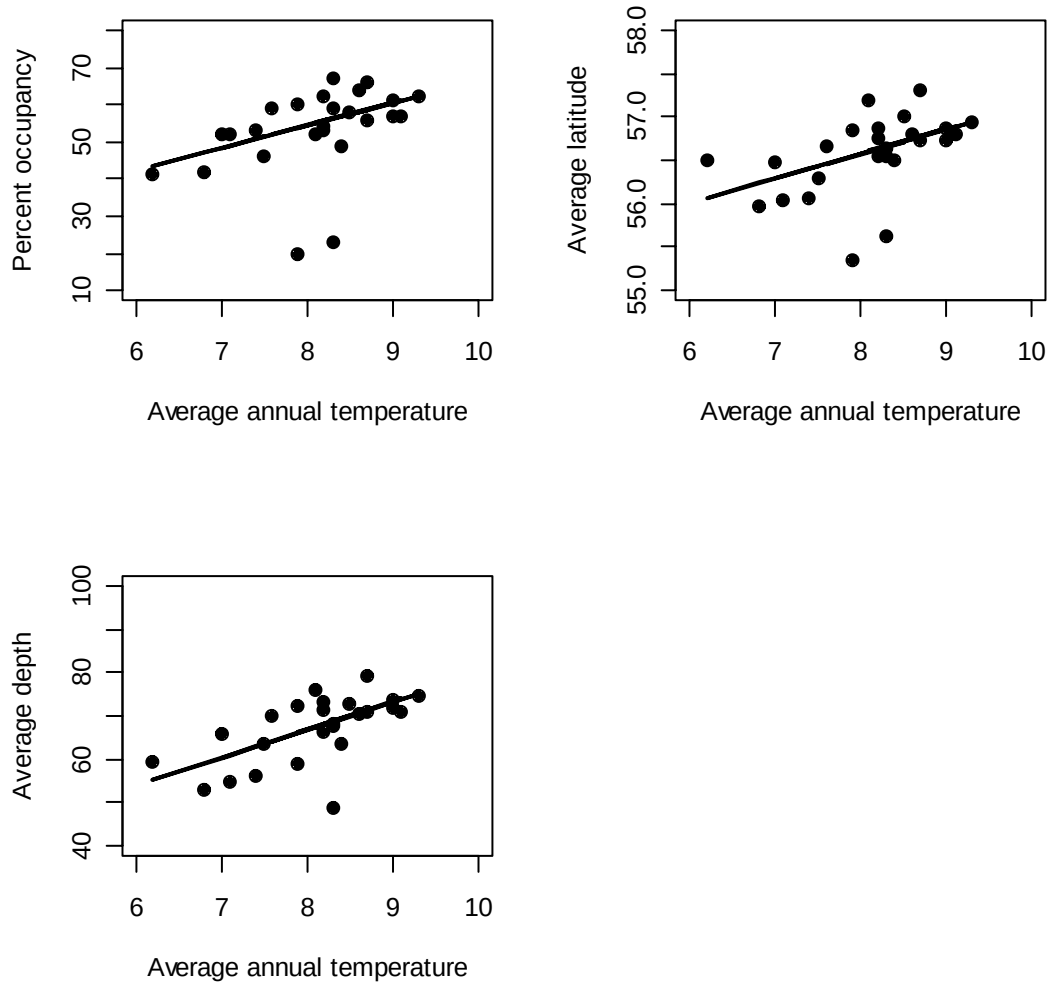


Figure 2.6.1. Change in herring occupancy, average latitude and average depth with average annual temperature across the English Groundfish Survey from 1980–2004. Lines are robust regression model fits and are significant at $P<0.001$.

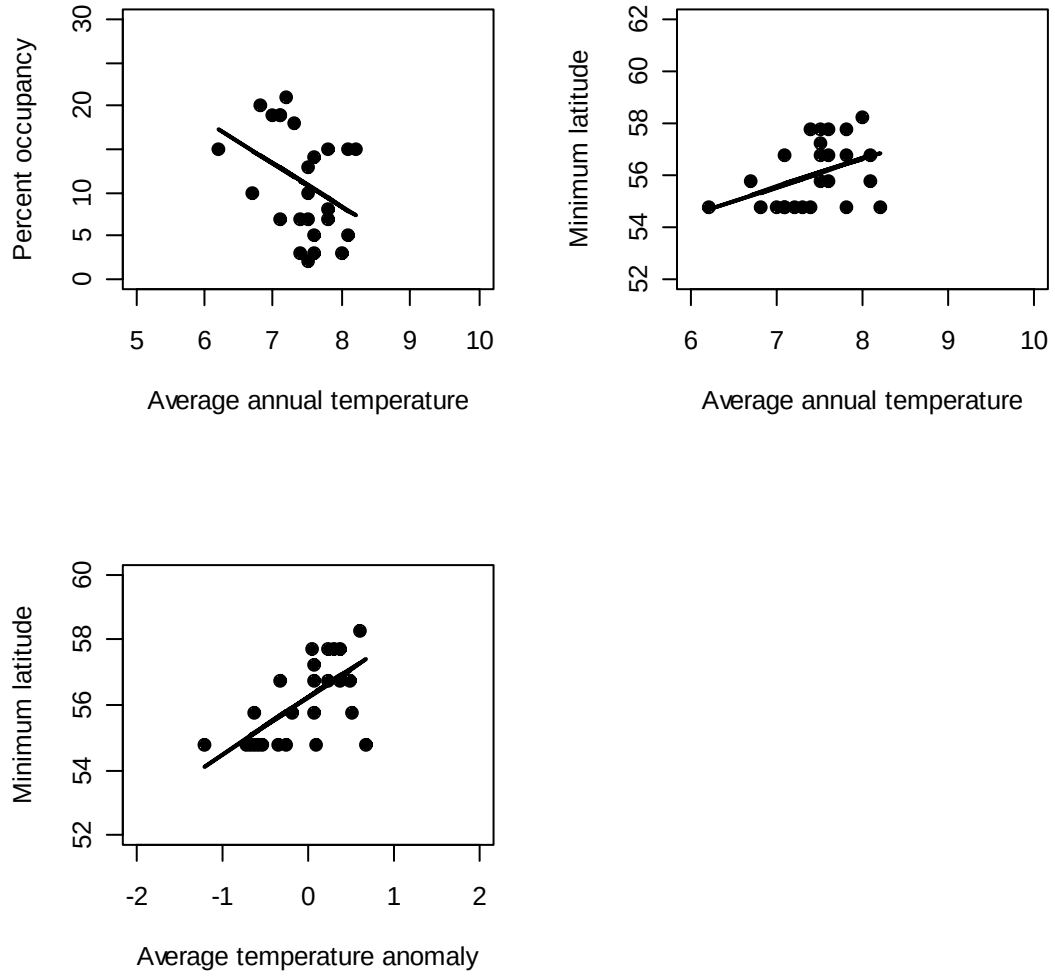


Figure 2.6.2. Change in wolfish occupancy and minimum latitude with average annual temperature, and bottom left panel is change in minimum latitude with average annual temperature anomaly across the English Groundfish Survey from 1980–2004. Note that only the bottom left panel is significant at $P<0.001$).

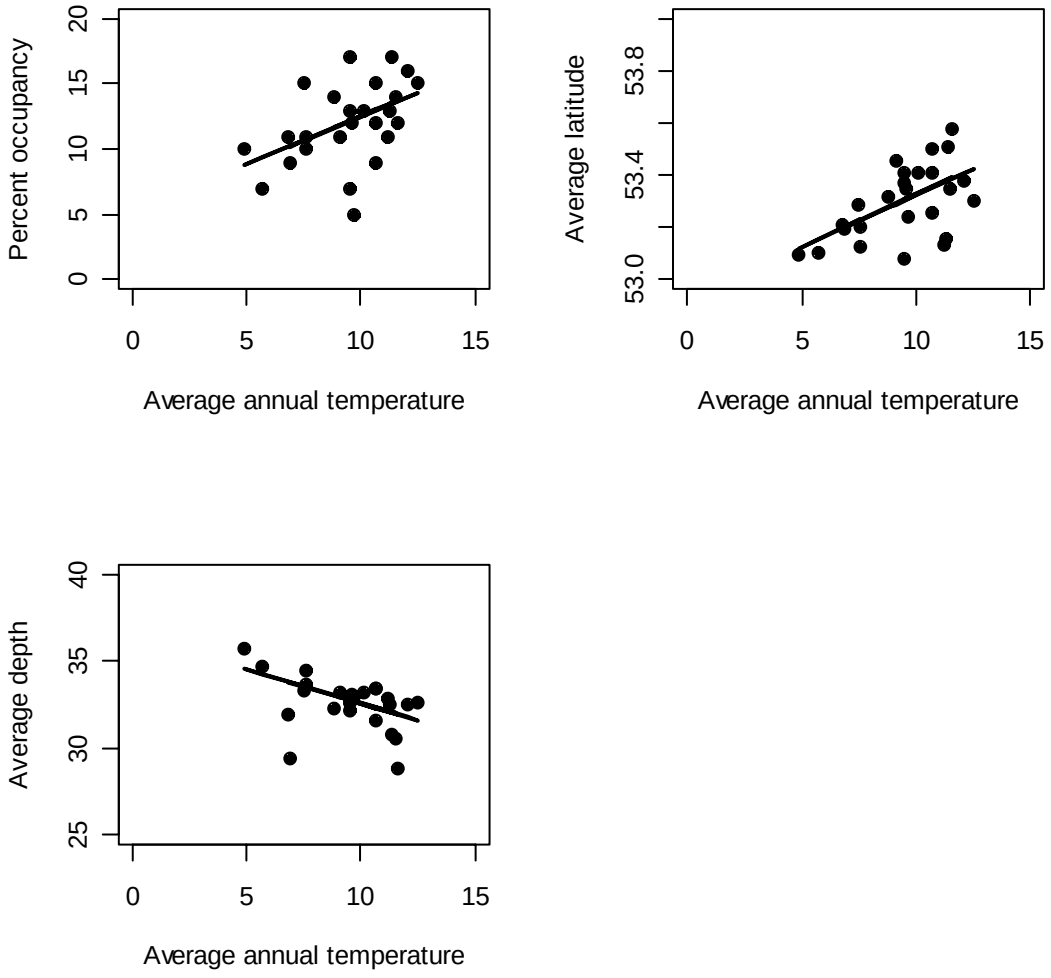


Figure 2.6.3. Change in lesser weaver occupancy, average latitude and average depth with average annual temperature across the English Groundfish Survey from 1980–2004. Lines are robust regression model fits and are significant at $P<0.01$.

2.6.3.3 Body size and temperature – distribution relationships

The relationship between the average annual temperature and occupancy is negative across all demersal fishes considered in this sample (robust $F = 3.85$, $P = 0.061$; Figure 2.6.4d). Smaller species spread out in warmer years, exhibiting positive annual temperature and occupancy relationships. The occupancy of larger species retracts in warmer years; with the strongest negative annual temperature and occupancy responses exhibited by spurdog and wolfish, which are boreal cold specialists (Figure 2.6.4d). In contrast there are no clear relationship between body size and the slope of the average annual temperature and measures of latitude (Figure 2.6.4a-c).

The temperature-minimum latitude relationships are weak, except southern range boundary of the boreal cold specialist wolfish (wf) retracts northwards in warmer years. In contrast, the lesser-spotted dogfish (lsd), which is a Luscitanian warm specialist invading the North Sea via the Northwest, has a negative relationship resulting in southward movement of the minimum latitude in warmer years (Figure 2.6.4a). The maximum latitude of many species tends to move northwards in warmer years, exhibiting positive temperature – mean latitude

relationship. The relationships are particularly strong for boreal cold specialists: anglerfish (af) and Norway pout (Np; Figure 2.6.4b).

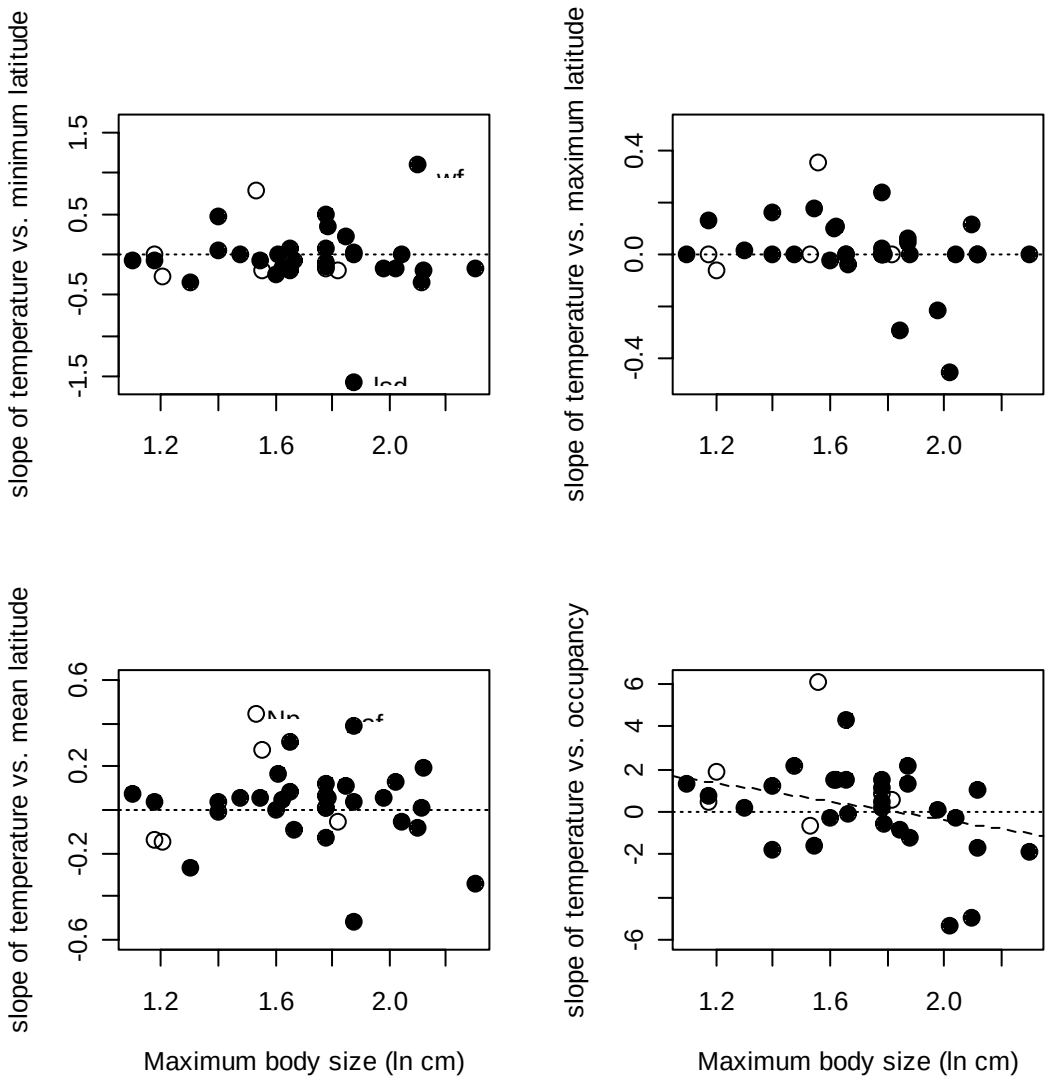


Figure 2.6.4. Maximum body size and the slope of the relationship between average annual temperature and (a) minimum latitude, (b) maximum latitude, (c) mean latitude and (d) occupancy. The line was fitted by robust regression and is significant at $P<0.06$.

2.6.3.4 Body size and NAO – distribution relationships

The slope of the relationship between the annual NAO index and occupancy is positive for all but one species, suggesting that species tend to spread out when the average annual NAO index is positive (Figure 2.6.5d). Overall larger-bodied species have a stronger positive association between the slope of the NAO-occupancy relationship and body size, compared to smaller-bodied species (robust F = 4.04, P = 0.053; binomial test, P = 0.028; Figure 2.6.5d). The increase in occupancy seems most apparent as a southward movement of the minimum latitude (Figure 2.6.5a). The relationship between the slope of the relationship between annual NAO index and average minimum latitude is negatively related to body size in demersal fishes. Smaller-bodied species tend to exhibit a positive relationship – with more northerly minimum latitudes in positive NAO years. By contrast, larger-bodied species, especially the

boreal cold specialists spurdog and ling and the warm specialist plaice, tend to exhibit a negative relationship – with the minimum latitudes expanding southwards in positive NAO years (Fig 2.6.5a).

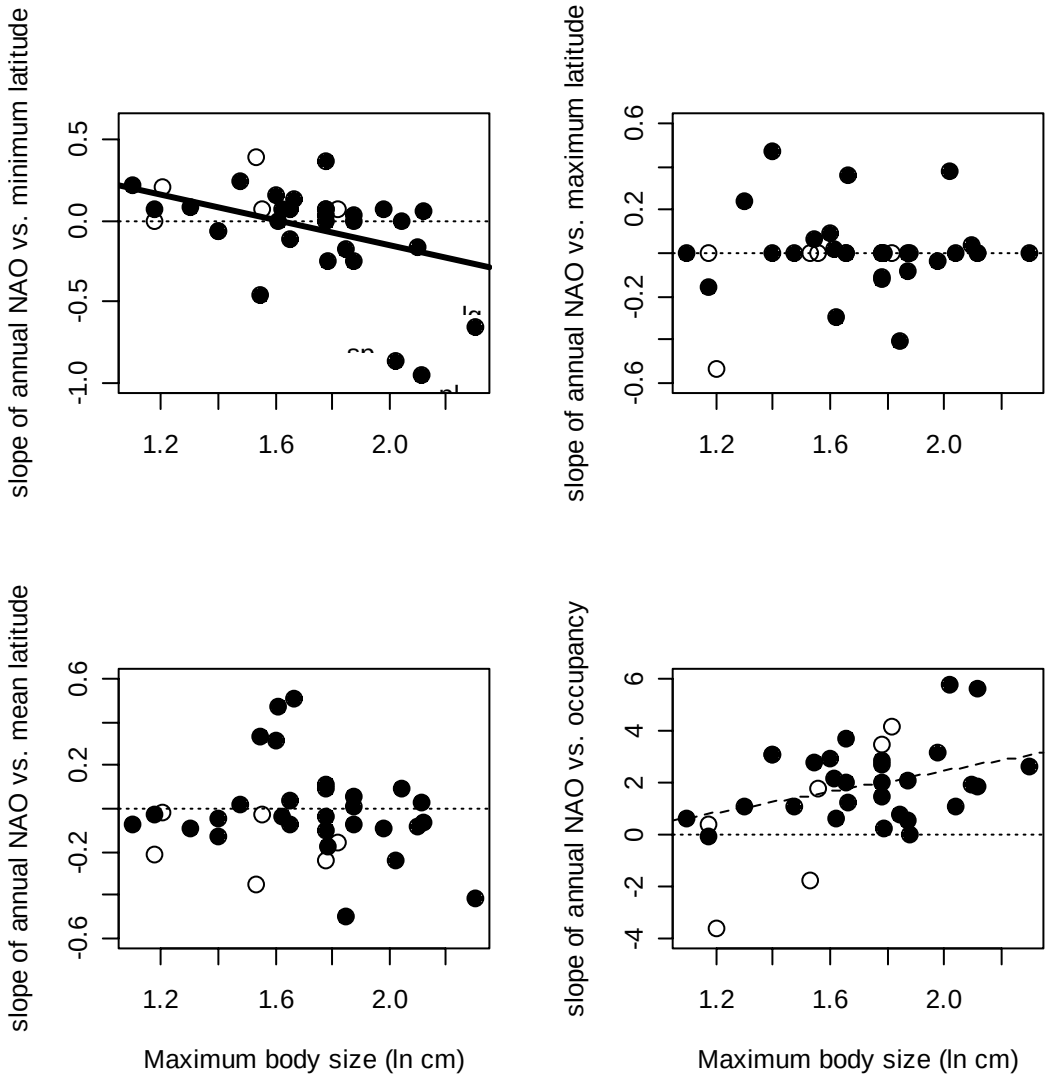


Figure 2.6.5. Maximum body size and the slope of the relationship between average annual NAO index and (a) minimum latitude, (b) maximum latitude, (c) mean latitude and (d) occupancy. Key: sp – spurdog, pl – plaice, lg – ling.

2.6.4 Discussion

This preliminary analysis is broadly consistent with the more synoptic analysis of Perry *et al.* (2005), and shows that species habitat occupancy and latitudinal and depth distributions are moving in response to interannual variation in climate-change driven variability in a range of hydrodynamics and sea temperatures. The Perry *et al.* (2005) paper was more focussed on longer term climate change rather than interannual environmental variability. The climate index used was a principle components axis comprised of 5-year running means of a range of environmental variables.

The main finding is that there is no single biogeographical measure that consistently responds to a single measure of hydrodynamics or temperature across the range of species on an interannual time-scale. There is considerable heterogeneity in species responses to the range of measures of interannual climate variability. This complexity may be underestimated as this

analysis has not considered lagging the distribution response to climate variability. The careful use of lags can provide considerable insight into the ultimate mechanisms underlying these large scale correlations (Rindorf and Lewy, 2006; Salen Picard *et al.*, 2002). Lags were not considered here as their use considerably increases the range of possible hypotheses to be tested. The diversity of hypotheses could be restricted based on field observations of scientists and fishers (Rindorf and Lewy, 2006), or through exploratory analyses of subsets of the data followed by confirmatory analyses (Cox, 1975; Myers, 1998).

There are an increasing number of single-species and comparative analysis, such as presented here, that consistently identify a number of species responding to one or several measures of temperature or hydrography. There is clearly scope to determine the underlying ecological factors, such as lifestyle (pelagic/demersal), trophic level and particularly body size, associated with the strength of response to environmental variation. This suggests that an appropriate goal might be to seek predictive models of population and community distribution in response to climate variability, as has been attempted with climate envelope models in terrestrial ecology (Davis *et al.*, 1998; Pearson and Dawson, 2003; Thomas *et al.*, 2004). However, comparative studies highlight a substantial proportion of species that do not appear to change distribution in response to climate variability. This raises two questions:

- What other aspects of their population biology may be responding to climate variation, such as population growth or mortality (Blanchard *et al.*, 2005; Kell *et al.*, 2005), and
- To what degree are species distributional responses to climate variability are constrained by a strong habitat association, say for a benthic habitat which may not be present further north or in greater depths?

This suggests that progress may require interaction between scientists focused on essential fish habitat and abundance-occupancy patterns and others focussed on climate-distribution issues.

In addition to these issues, the degree to which changes in fish distribution can be reliably linked to temperature and hydrography will depend on:

- a more detailed mechanistic understanding of not only of the physics of climate change, including non-stationarity of the NAO, and links among NAO, Gulf Stream, North Atlantic Current and North Sea influx (e.g. Pingree, 2005; Polyakova *et al.*, 2006),
- but also of how the physical processes influence critical stages in the life history and ecology of species (Reid *et al.*, 2003).
- Overshadowing all of this is the potentially confounding issue of changing distributions and occupancy as a consequence of fisheries exploitation (Fisher and Frank, 2004).

Table 2.6.1. North Sea species surveyed by the English Groundfish Survey, body size, biogeographic affinity and thermal characteristics.

COMMON NAME	LATIN BINOMIAL	BODY SIZE LMAX (CM)	BIOGEOGRAPHIC AFFINITY	MEAN TEMPERATURE	TEMPERATURE RANGE	THERMAL CLASSIFICATION
pogge	Agonus cataphractus	20	boreal	8.9	17.9	warm generalist
wolfish	*Anarhichas lupus*	125	boreal	7.3	4.7	cold specialist
argentine	*Argentina sphyraena*	35	lusitanian	7.7	4.6	cold specialist
scaldfish	*Arnoglossus laterna*	25	lusitanian	9.6	15.7	warm generalist
solenette	*Buglossidium luteum*	12.5	lusitanian	9.6	18.2	warm generalist
herring	*Clupea harengus*	36	boreal	8.2	18.2	cold generalist
grey gurnard	*Eutrigla gurnardus*	45	lusitanian	8.3	18.2	warm generalist
silvery pout	*Gadiculus argenteus*	15	lusitanian	7.8	4.6	cold specialist
cod	*Gadus morhua*	131.8	boreal	8.0	13.5	cold generalist
witch	*Glyptocephalus cynoglossus*	60	boreal	7.6	7.8	cold specialist
long-rough dab	*Hippoglossoides platessoides*	30	boreal	7.8	9.7	cold specialist
megrim	*Lepidorhombus whiffiagonis*	61	lusitanian	8.2	4.7	warm specialist
dab	*Limanda limanda*	42	boreal	8.3	18.2	warm generalist
angler	*Lophius piscatorius*	74.6	lusitanian	7.9	7.9	cold specialist
haddock	*Melanogrammus aeglefinus*	75.5	boreal	7.8	9.7	cold generalist
whiting	*Merlangius merlangus*	44.9	lusitanian	8.2	18.2	warm generalist
hake	*Merluccius merluccius*	110	lusitanian	8.1	7.9	cold specialist
blue whiting	*Micromesistius poutassou*	34	atlantic	7.9	6.2	cold specialist
lemon sole	*Microstomus kitt*	60	boreal	8.0	14.7	cold generalist
ling	*Molva molva*	200	boreal	7.9	5.3	cold specialist
plaice	*Pleuronectes platessa*	95	lusitanian	8.4	18.2	warm generalist
saithe	*Pollachius virens*	130	boreal	7.7	8.3	cold specialist
cuckoo ray	*Raja naevus*	70	lusitanian	8.4	5.1	warm specialist
starry ray	*Raja radiata*	60	boreal	7.5	9.1	cold specialist
four-bearded rockling	*Rhinonemus cimbrius*	41	boreal	8.2	8.2	cold specialist
mackerel	*Scomber scombrus*	66	atlantic	8.4	18.2	warm generalist
lesser-spotted dogfish	*Scyliorhinus canicula*	75	lusitanian	9.0	9.2	warm specialist
sole	*Solea solea*	60	lusitanian	9.7	12.9	warm generalist

COMMON NAME	LATIN BINOMIAL	BODY SIZE LMAX (CM)	BIOGEOGRAPHIC AFFINITY	MEAN TEMPERATURE	TEMPERATURE RANGE	THERMAL CLASSIFICATION
sprat	*Sprattus sprattus*	16	lusitanian	8.9	15.1	warm generalist
spurdog	*Squalus acanthias*	105	atlantic	7.8	9.1	cold specialist
lesser weaver	*Trachinus vipera*	15	lusitanian	9.8	17.9	warm generalist
horse mackerel	*Trachurus trachurus*	60	lusitanian	8.7	18.2	warm generalist
Norway pout	*Trisopterus esmarki*	25	boreal	7.7	8.3	cold specialist
bib	*Trisopterus luscus*	46	lusitanian	9.7	14.4	warm generalist
poor cod	*Trisopterus minutes*	40	lusitanian	8.6	13.3	warm generalist

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3 Abundance-occupancy relationships

3.1 Case study — a preliminary evaluation of the effects of environmental change on abundance-distribution relationships in southern Gulf of St. Lawrence (Canada) marine fishes

Optimal foraging theory predicts that habitat selection should be density-dependent (Fretwell and Lucas 1970). Because of intraspecific competition, as abundance increases, individuals spread into less preferred habitat such that the "fitness" of all individuals in the population is equalized (Morris 1987). Distribution is expected to expand into marginal habitat as abundance increases, and contract into optimal habitat as abundance decreases (MacCall, 1990). This prediction that geographic range will contract as abundance declines has important consequences, as vulnerability to exploitation increases as geographic range declines (Paloheimo and Dickie, 1964). Other implications for fisheries include the planning of protected areas, the identification of core habitats, the scaling of dynamics from local to regional scales and the use of presence/absence as an index of abundance (reviewed in ICES 2004, 2005).

There is indeed considerable evidence to indicate that species spatial distribution may vary with population abundance, though there have been few interspecies comparisons for marine fish (ICES 2004, 2005; but see Fisher and Frank 2004). Defining the mechanism(s) underpinning these empirical results is not trivial (Shepherd and Litvak, 2004). Furthermore, though the theory is reasonably well developed, there is a lack of empirical understanding of how changes in the environment (e.g. availability of preferred prey or favourable thermal condition) may affect density-dependent interactions (e.g. Blanchard *et al.*, 2005). By affecting the suitability of habitats, environmental changes should affect the distribution of animals at a given abundance if indeed their habitat selection is ideal and free (Fretwell and Lucas, 1970), such that they can distribute themselves so as to maximize their fitness.

The relationship between species abundance and spatial distribution in the southern Gulf of St. Lawrence (sGSL) in eastern Canada was examined. Data from a synoptic bottom trawl survey were used. The degree to which the availability of waters of preferred temperatures affects the relationship was assessed using an empirical model. Both density-independent and dependent effects of habitat change were included in the analysis.

3.1.1 Methods

The ecosystem

The southern Gulf of St. Lawrence consists of shallow shelf (mainly <100m in depth), bordered by land on three sides and by a 400–500 m trench on the fourth. From late spring to early autumn, the waters of the southern Gulf are divided into three layers: a warm mixed layer (surface-30m; 2 to 20°C), a cold intermediate layer (CIL: 30–150m; -1 to 2°C) and a warmer saltier deep layer (3–4°C) (Gilbert and Pettigrew 1997). During the winter, the top layer merges with the CIL. The CIL touches the bottom of over 75% of the area. Temperatures in the CIL cooled throughout the 1980s, resulting in several consecutive years during the early 1990s of the coldest temperatures recorded in over 50 years of monitoring (Drinkwater and Gilbert 2004; Gilbert and Pettigrew 1997).

Most southern Gulf fishes are spring to autumn residents of the ecosystem. Many species overwinter in warm deep waters in and just outside the Gulf of St. Lawrence. Notable exceptions include Atlantic mackerel (*Scomber scombrus*) and spiny dogfish (*Squalus acanthias*), which overwinter in more southern waters.

Annual survey

The annual sGSL bottom-trawl survey has been conducted each September since 1971. The surveys follow a stratified random design, with stratification based on depth and geographic area. The target fishing procedure in all years was a 30-minute tow at 3.5 knots (all catches have therefore been standardised to a 1.75 nautical mile tow). Surveys were conducted using a Yankee 36 trawl from 1971 to 1985 and a Western IIA trawl since 1985. A 19-mm cod-end liner was used in all years to retain small fishes. The research vessels conducting the survey were the *E. E. Prince* from 1971–1985, the *Lady Hammond* using a Western IIA trawl in 1985–1991, the *Alfred Needler* in 1992–2002, the *Wilfred Templeman* in 2003, and both the *Alfred Needler* and the *Teleost* in 2004–2005. Comparative fishing experiments were conducted during or shortly before the September surveys in 1985, 1992, 2004 and 2005 in order to estimate relative fishing efficiency between gears and/or vessels. Species-specific adjustments based on these calibration experiments were applied to the survey data prior to analysis where necessary to ensure comparability of species abundance data from one year to the next (Benoît and Swain, 2003b).

Fishing was conducted only during daylight hours (07:00–19:00) in 1971–1984 but 24-h per day since 1985. In order to keep the two periods comparable, catches are normally adjusted for diel differences in fishing efficiency, as described in Benoît and Swain (2003a). Because these corrections do not adjust for possible diel differences in the probability of catching a species, only the daytime survey sets were included in the analyses for species showing a diel effect. All sets were included for the other species in order to maximize the precision of the various quantities calculated here.

Although the survey protocol since 1971 has been to sort (and record) catches of finfish by species, identification at sea is problematic for four genera in particular: *Liparis* (seasnails), *Lycodes* (eelpouts), *Sebastes* (redfish) and *Alosa* (gaspereau). While attempts are made to identify the former two genera to the species level, it is felt that this has been done inconsistently and the reliability of the identification is questionable and they are therefore grouped to the genus level here.

A proper analysis of species-specific abundance-distribution relationships requires that that a constant survey area be used in all years (He and Gaston 2000). Because two strata were not sampled in the 2003 survey, data from that year were dropped from the analyses. Furthermore, although over 70 fish taxa have been captured in the sGSL survey since 1971, only those that were captured in at least 10 survey years were included in this analysis. When making interspecific comparisons, all analyses were repeated restricting them to those species captured in >5% of survey tows since 1971 (~230 tows) to avoid possible biases related to low frequency of capture.

Computed Quantities

A number of quantities are calculated for each of the 57 taxa considered in the analysis.

Each tow (t) within each stratum (s) and year (y) was assigned a statistical weight (w_{yst}) that can be interpreted as the proportion of the survey area represented by that tow:

$$w_{yst} = \frac{A_s}{A}\frac{1}{T_{ys}} \tag{1}$$

where T_{ys} and A_s are respectively the number of tows and the area (km^2) area of stratum s and

$$A = \sum_{s=1}^{S_y} A_s \tag{2}$$

In a few instances where repeat tows were made at the same sites in year y, w_{yst} was divided by the number of repeat tows at t.

The mean survey catch rate (i.e. relative abundance index) in year y is therefore

$$\overline{U_y} = \sum_{s=1}^{S_y} \sum_{t=1}^{T_{ys}} w_{yst} u_{yst} \tag{3}$$

where u_{yst} is the catch of the species under study in tow t. The total area occupied (km^2) by that species in year y (B_y) is therefore:

$$B_y = A \sum_{s=1}^{S_y} \sum_{t=1}^{T_{ys}} w_{yst} I \qquad \text{where } I = \begin{cases} 1 \text{ if } u_{yst} > 0 \\ 0 \text{ otherwise} \end{cases} \tag{4}$$

Area occupied will decrease as population size decreases even if there is no increase in geographic concentration (Swain and Sinclair 1994). An index of geographic concentration that is insensitive to changes in abundance, D_{95}, the minimum area containing 95% of the individuals of the species (Swain and Sinclair 1994), was therefore used. To calculate D_{95}, *F(c)*, the catch-weighted cumulative distribution function (cdf) of species catch in y, was first calculated:

$$F_y(c) = 100 \sum_{s=1}^{S_y} \sum_{t=1}^{T_{ys}} w_{yst} \frac{u_{yst}}{\overline{U}_y} I \qquad \text{where } I = \begin{cases} 1 \text{ if } u_{yst} \le c \\ 0 \text{ otherwise} \end{cases} \tag{5}$$

where c is a level of catch (i.e. number per standard tow). *F(c)* provides an estimate of the proportion of the species that occurs at a local density of c or less. F was evaluated at intervals of 1 and the density c_{05} corresponding to F=5 was calculated. This is the density at or below which the most sparsely distributed 5% of the species are estimated to occur. The area containing the most sparsely distributed 5% of the species (including areas where no individuals were caught) was calculated as follows:

$$G_y(c_{05}) = A \sum_{s=1}^{S_y} \sum_{t=1}^{T_{ys}} w_{yst} I \qquad \text{where } I = \begin{cases} 1 \text{ if } u_{yst} \le c_{05} \\ 0 \text{ otherwise} \end{cases} \tag{6}$$

Thus, the minimum area containing 95% of the species (D) is given by:

$$Dy = A - Gy(c05) \tag{7}$$

One of the interests in this paper was to examine the influence of changing environmental conditions, in this case bottom-temperatures, on the relationship between species abundance and distribution. To do so, species-specific temperature series that reflect the preferences of each species were used. Perry and Smith (1994) developed an approach to compare species habitat preferences to available habitat. (Note that mere species-habitat probability distributions will only reflect species preferences if all habitats are equally available). When calculating temperature preferences for sGSL species it is important to account for depth, as the coldest temperatures in the system occur at intermediate depths in the CIL (Figure 3.1.1.1).

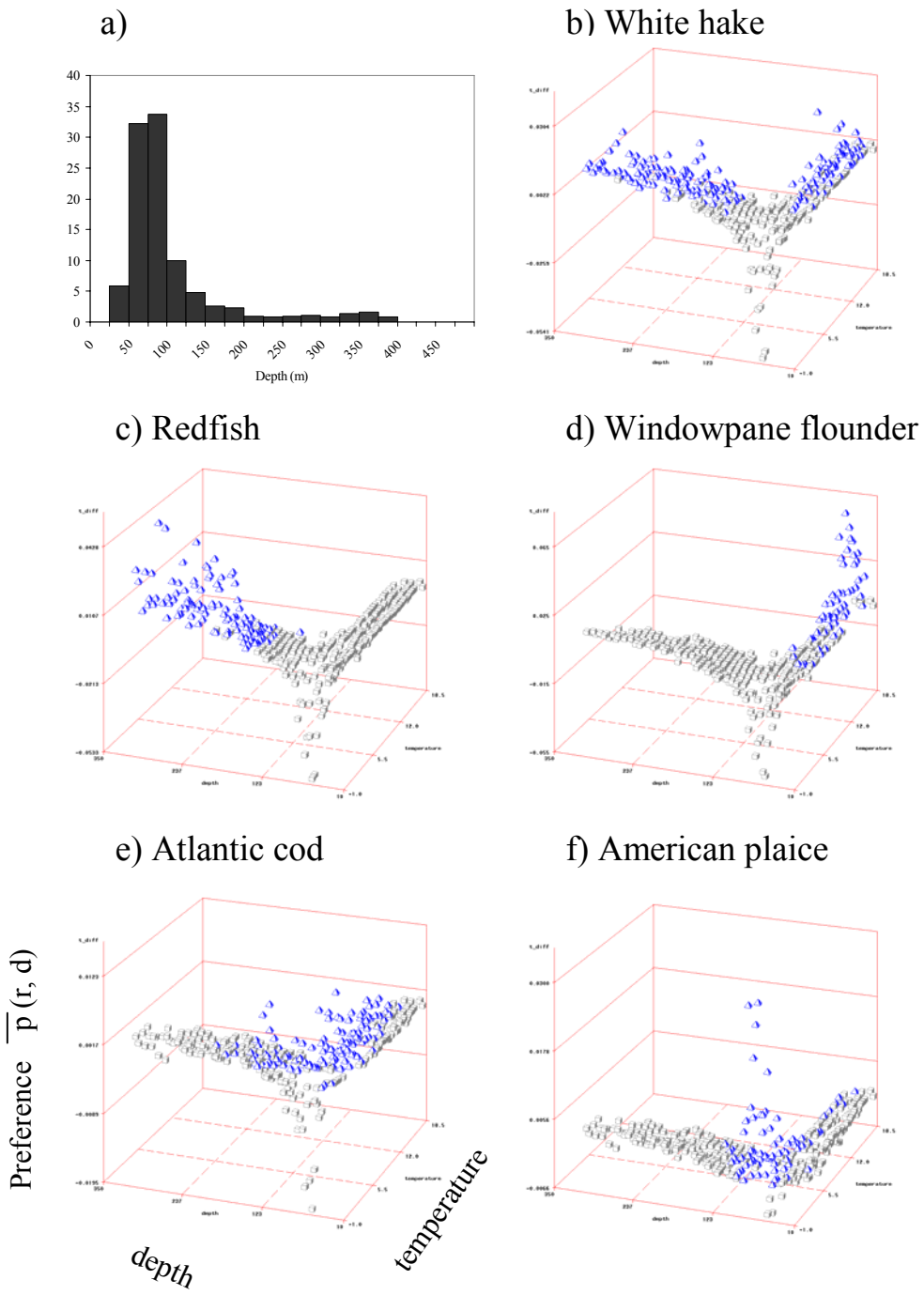


Figure 3.1.1.1. a) Availability of depths in the sGSL survey area. b-f) Examples of depth-dependent temperature preferences for b) a species that prefers intermediate and warm temperature waters, regardless of depth, c) a deep water species, d) a shallow water species, e) a broadly distributed species that prefers waters of intermediate temperature and f) a broadly distributed species that prefers cool waters. Dark pyramid points represent a preference, relative to availability, whereas lighter cubes represent avoidance.

The available bottom temperatures in year *y* were therefore described using the following pdf (in %):

$$f_y(r,d) = 100 \sum_{s(=8)}^{S_y} \sum_{t=1}^{T_{ys}} w_{yst} I \qquad \text{where } I = \begin{cases} 1 \text{ if } x_{yst} = r(d_t) \\ 0 \text{ otherwise} \end{cases}$$

where r and d are levels of temperature and depth, respectively, and x_{yst} is the bottom-temperature measured at the end of tow t which occurred at depth d_t.

The association of a given species to particular bottom-temperatures in year y was estimated using a catch-weighted pdf (again in %) given by

$$g_y(r,d) = 100\sum_{s=1}^{S_y}\sum_{t=1}^{T_{ys}} w_{yst}\frac{u_{yst}}{\overline{U}_y} I \qquad \text{where } I = \begin{cases} 1 \text{ if } x_{yst} = r(d_t) \\ 0 \text{ otherwise} \end{cases} \tag{9}$$

Temperature preference for a given species and year is therefore

$$py(r,d) = gy(r,d) - fy(r,d) \tag{10}$$

and an average temperature preference across n survey years is

$$\overline{p}(r,d) = \frac{1}{n}\sum_{y=1}^{n} p_y(r,d) \tag{11}$$

Examples of temperature preferences for five species are presented in Figure 3.1.1.1.

The time series of bottom-water temperature was calculated using two metrics of preference, based on eqn. 11. The first was chosen to reflect the survey area covered by waters that are generally preferred by the species. To that end, eqn. 11, calculated using all survey years (i.e. n=36), was reduced to a binary function [$\underline{p}_b(r)$] by setting all positive values to 1 and all negative values to zero. The total area (km^2) covered by generally preferred waters in year y (M_y) was then calculated as

$$M_y = A\sum_{s=1}^{S_y}\sum_{t=1}^{T_{ys}} w_{yst}\,\overline{p_b}(r = x_{yst}, d = d_{yst}) \tag{12}$$

The second metric of preference was chosen to directly reflect preference during periods of low abundance. Ideal free distribution (IFD) theory predicts that species will contract to core areas of preferred habitat at low abundance. Equation 11 was therefore calculated using the 25% of years of lowest non-zero abundance for each species. A non-dimensional weighted index of preferred temperature availability was then calculated as

$$N_y = \sum_{s=1}^{S_y}\sum_{t=1}^{T_{ys}} w_{yst}\,\overline{p}\,(r = x_{yst}, d = d_{yst})$$

$$I \qquad \text{where } I = \begin{cases} 1 \text{ if } \overline{p}\,(r = x_{yst}, d = d_{yst}) > 0 \\ 0 \text{ otherwise} \end{cases} \tag{13}$$

(Note that the original version of eqn. 11 is used in this equation rather than the binary version). This index therefore represents the availability of preferred waters, weighted by preference. Use of this index in analysis implies that a given change in the availability of waters of highly preferred temperatures will have a larger impact on species distribution than a change of similar magnitude in the availability of less preferred temperature waters. While Shackell *et al.* (2005) suggested that range may not contract to core areas as abundance decreases because of spatial patterns of fishing mortality; this was not believed to be the case for sGSL species, as most are fished during their annual migrations into or out of the area.

Finally, an index of mean preferred habitat saturation ($\overline{O}$), was calculated as

$$\overline{O} = \frac{1}{n}\sum_{y=1}^{n}\frac{M_y - B_y}{M_y} \tag{14}$$

which represents the mean proportion of generally preferred waters for the species that was unoccupied on average over all survey years.

Statistical analysis

There is no single agreed-upon used mechanistic model that relates abundance and distribution of animals. Consequently a simple empirical model was used. The general relationship between abundance and distribution was described as follows,

$$D \approx \overline{U}^{\beta} \tag{15}$$

such that mean density (individuals·km^2), $\overline{E}$, is

$$\overline{E} = \frac{\overline{U}}{D} \approx \overline{U}^{1-\beta} \tag{16}$$

From eqns. 15 and 16, it can be seen that if β=0 (i.e. constant distributional area), both local density and therefore commercial CPUE will reflect abundance. This is not true for values of β>0. Indeed if β=1, distribution is proportional to abundance and local density is constant.

The parameters of the linear version of eqn. 15 can be estimated using the regression equation

$$\ln D_y = \beta_0 + \beta_1 \ln \overline{U}_y + \varepsilon \tag{17}$$

In preliminary analyses, eqn. 17 appeared to fit the data well such that there were no trends or heteroscedasticity in the residuals.

Changes in habitat availability (bottom waters of preferred temperatures in the present case) can affect the relationship in eqn. 17 in two general ways. The first is a density-independent change in distribution with changing environmental conditions. Using X_y to represent the area covered by preferred temperatures in year y, eqn. 17 then becomes,

$$\ln D_y = \beta_0 + \beta_1 \ln \overline{U}_y + \beta_2 \ln X_y + \varepsilon \tag{18}$$

In such as scenario, for a given level of abundance, the distribution of individuals is predicted under IFD theory to spread as the area of preferred habitat increases (i.e. $\beta_2 \geq 0$), thereby reducing local density and intraspecific competition.

The second manner in which bottom-temperatures can affect the distribution-abundance relationship is via a density-dependent effect, such that β in eqn. 15 depends on the availability of preferred temperature waters. The simplest way to express this is as

$$D \approx \overline{U}^{\beta X} \approx \overline{U}^{\beta_i + \beta_{ii} X} \tag{19}$$

with the rightmost form allowing for both habitat independent and dependent effects on the exponent. This effectively amounts to adding an interaction term between abundance and preferred temperature availability to eqn. 18,

$$\ln D_y = \beta_0 + \beta_1 \ln \overline{U}_y + \beta_2 \ln X_y + \beta_3 \ln \overline{U}_y \cdot X_y + \varepsilon \tag{20}$$

Here the expectation would be that as the "favourability" of temperatures in occupied water increases, individuals will be less apt to spread out at a given abundance (i.e. $\beta_2 \leq 0$).

Throughout, the effects of environmental change on the abundance-distribution relationship were assumed to occur without a time lag. If indeed the relationship reflects density-dependence and that fish distribution is ideal and free, we would expect that environmentally-induced changes in individual's fitness should manifest themselves rapidly and individuals would re-distribute themselves accordingly.

The parameters of eon. 20 can easily be found using ordinary least squares (OLS) multiple regression. However this technique assumes that measurement error in the independent variable is much smaller than that of the dependent variable(s). Clearly this is not the case here. Standard minor axis regression (SMA; sensu McArdle, 1988), the multiple regression equivalent of reduced major axis regression (RMA), was therefore used. This technique is preferred when the error rate in the independent variables is greater than about half the error rate in the dependent variable (McArdle, 1988) – a reasonable assumption in the present case. The technique consists of employing a Principal Components Analysis on the correlation matrix. The component associated with the smallest eigenvalue is the linear combination of the original variables with the smallest variance, provided the data are approximately multivariate normal. The size of this smallest eigenvalue gives a measure of the residual variation or goodness of fit and the parameters in eqn. 20 are the coefficients of the eigenvector, each multiplied by the ratio of their respective sample standard deviation and that of the dependent variable (see McArdle, 1988 for details). As with RMA, the slopes obtained from SMA tend to be steeper than those obtained from OLS regression. Confidence intervals for the slopes were estimated using the bootstrap method. Prior to calculating the interaction term, ln-abundance and the temperature index were standardized (z-scores) to remove non-essential collinearity between the interaction and individual terms (Montgomery and Peck, 1982).

Finally for one analysis an absolute rather than relative measure of abundance was desired. This required making some adjustment for survey trawl catchability. Using hierarchical Bayesian analyses including survey catch rates and analytical stock assessment estimates of absolute abundance, Harley and Myers (2001) estimated length-dependent catchability models for a small number of species or species groups to attempt to correct for such differences (see also Harley *et al.*, 2001). Additionally, a length-dependent relationship for pelagic fishes using data from sGSL herring has been estimated (H. Benoît *unpublished analyses*). Although these length- and species-specific adjustments were applied it is very important to note that they are approximate at best.

3.1.2 Results

A statistically significant effect of abundance on distribution was found for the majority of sGSL species using SMA, where N_y was used as the index of water temperature availability (Figure 3.1.2.1a). Considering only the cases where β_1 was significantly different from zero, all but two had positive values. Even considering non-significant point estimates, the majority of cases were positive. This indicates in general that area occupied is not proportional to abundance, but rather hyperstable (Hilborn and Walters, 1992). This would suggest a mechanism in which commercial CPUE would also be hyperstable. The two cases of significant negative parameter estimates, American plaice and witch flounder are initially perplexing, as they indicate that as abundance increases, area decreases (i.e. density increases). This is clearly not expected under IDF theory and may reflect changes in size structure, for example, or the amount of available habitat (see below). Although further study is obviously needed, it is clear even in those cases that local abundance (density) will not provide an index proportional to abundance.

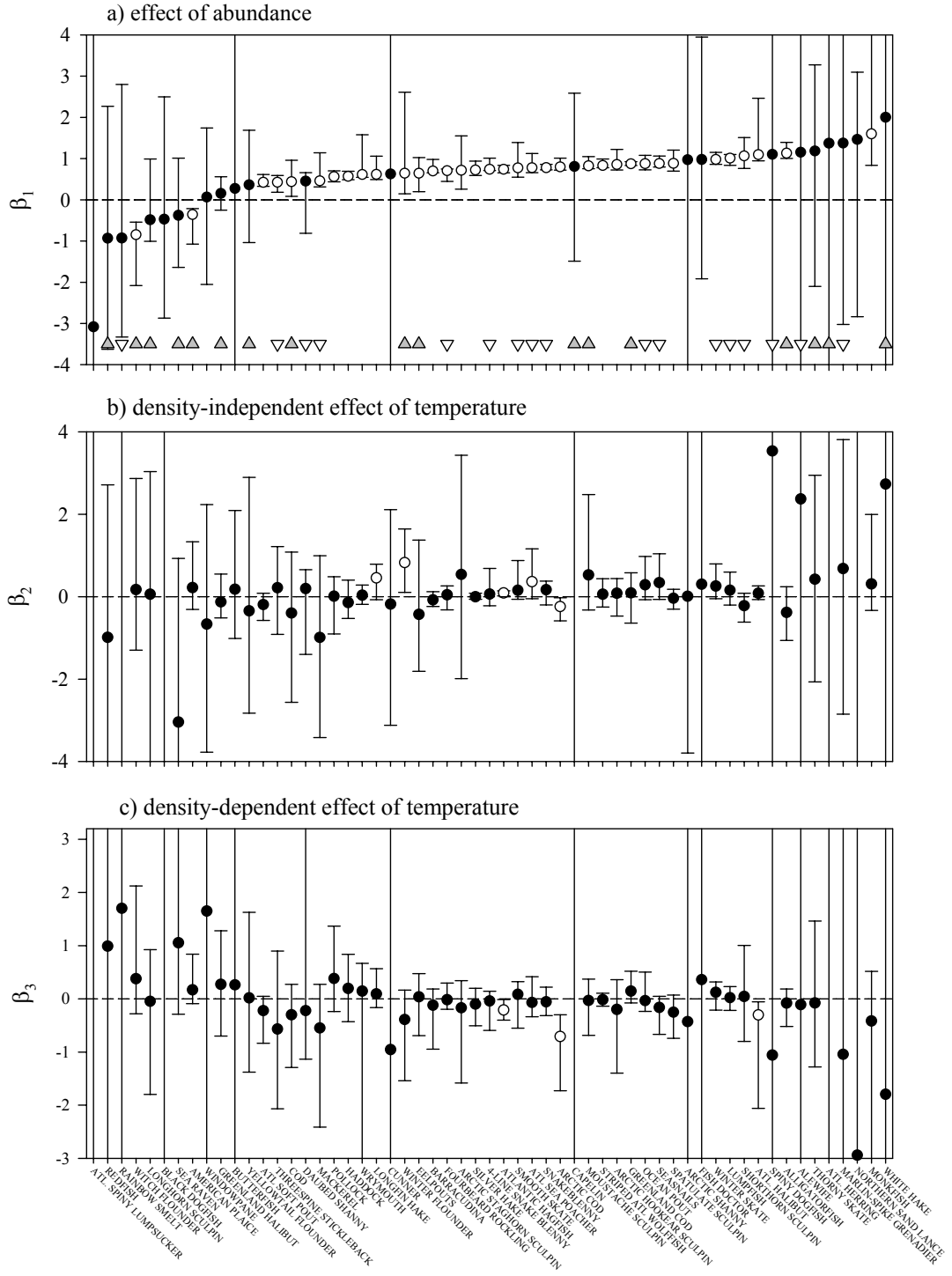


Figure 3.1.2.1. SMA results for the three main parameters of eqn. 20, where N_y (weighted index of core preferred temperature availability) was used as the index of temperature. The figures display the point estimate ±90% confidence intervals. White symbols indicate parameter values that are significantly different from zero at a Type-I error rate of 5%. Results are sorted along the abscissa by the point value of β_1 for the 57 sGSL marine fish species. The triangles in panel (a) indicate the proportion of total survey tows (1971–2006) in which the species was found: >10% of tows grey triangle, 5–10% of tows white triangle.

Significant density-independent effects of temperature were found for 5 species; positive (expected) parameter estimate for four species and negative for the other (Figure 3.1.2.1b). This number of statistically significant cases is approximately what would have been expected by chance alone, at a Type-I error rate of 5%. The results therefore do not suggest a prevalent density-independent effect of available temperatures on abundance-distribution in this fish community. That is not to say that changes in temperature conditions have not affected abundance, which in turn affected distribution, or affected both abundance and distribution, only that they do not seem to have affected them differentially, on average. The same is true for a density-dependent effect of temperature, where statistically significant effects were found for only three species (Figure 3.1.2.1c). The relatively minor effect of temperature change is evident in Figure 3.1.2.2, where the observed and predicted distributions of the six species showing a significant temperature effect are plotted as a function of their abundance. A lack of contrast in environmental conditions does not seem a probable explanation for the paucity of significant results for the availability of preferred-temperature waters (Figure 3.1.2.3). However a lack of long term or low-frequency change, to which species may be more apt to respond, may still be a possible explanation. Alternatively, effects of environmental change may not manifest themselves instantaneously, as assumed here, but rather with a time lag that reflects inertia in species re-distribution.

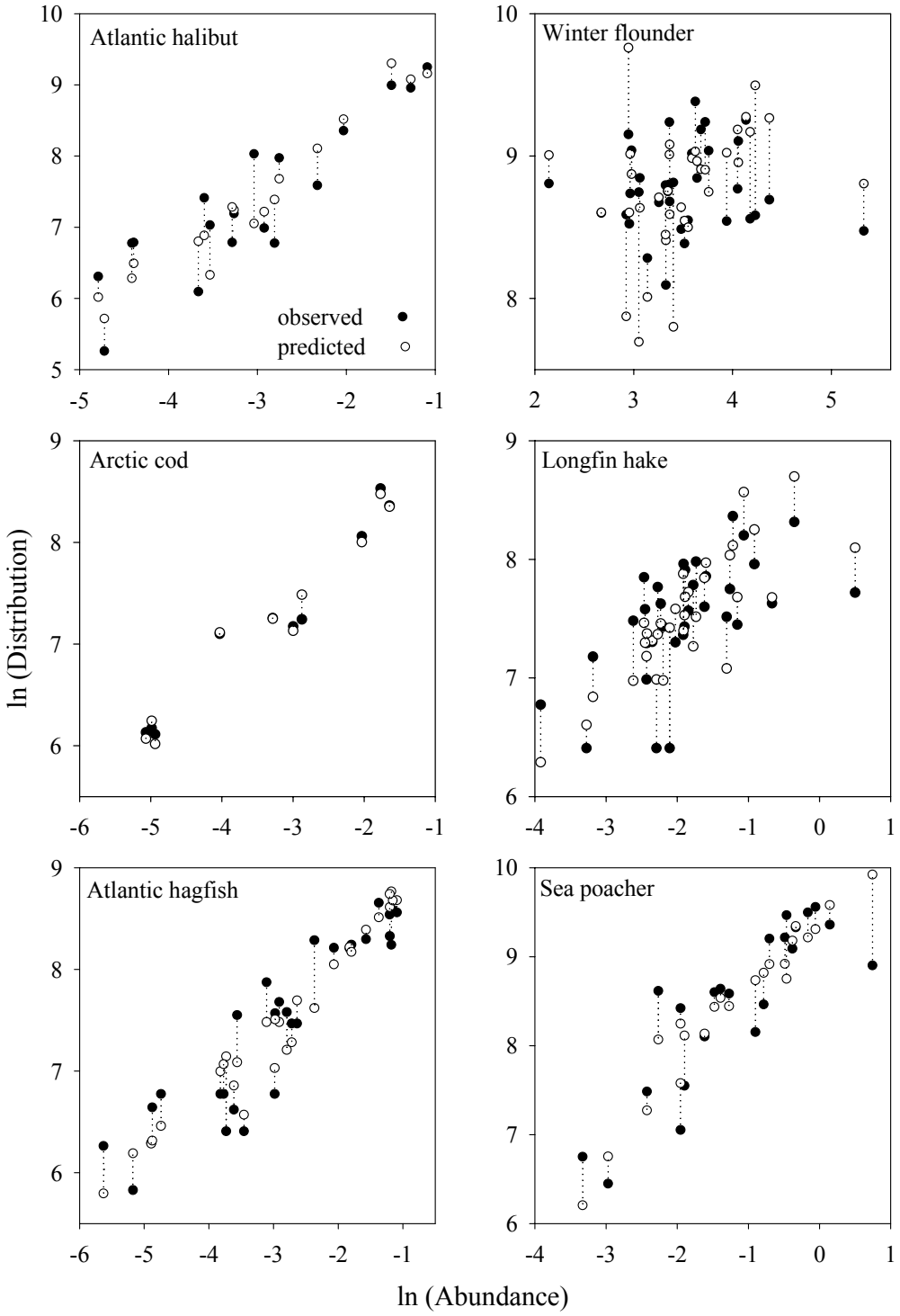


Figure 3.1.2.2. Observed and fitted values of the abundance-D_{95} relationship for the six species for which significant effects of the index of core preferred temperature availability were found. Deviations in predicted values from a strait line relating ln(abundance) and ln(distribution) reflect the magnitude of the predicted temperature effect.

Qualitatively speaking, the preceding analysis appears to have produced comparable results for more common (occurring in >5% of survey tows, 1971–2006) and less common species. This suggests that for the species included in this analysis, there is little evidence for a bias due to inclusion of species for which only a small number of individuals are ever captured in a given survey (Figure 3.1.2.1a).

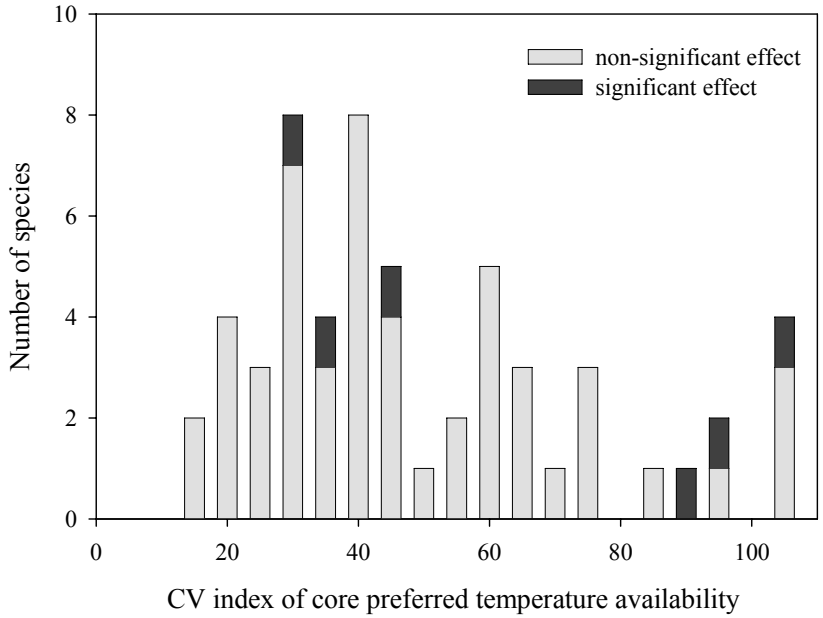


Figure 3.1.2.3. Histogram of the coefficient of variation of the core preferred temperature availability index, separated into species that showed a significant effect of that index on their abundance-distribution relationship (dark grey) and those that did not

The SMA analysis was repeated, excluding the factors related to temperature in eqn. 20., for those species for which neither a significant density-dependent or density-independent effect of core preferred-temperature availability was found. The confidence intervals around the resulting estimates of β_1 were considerably smaller than those in the original analysis (Fig 3.1.3.4a). As a result a number of previously non-significant cases were significant in this new analysis.

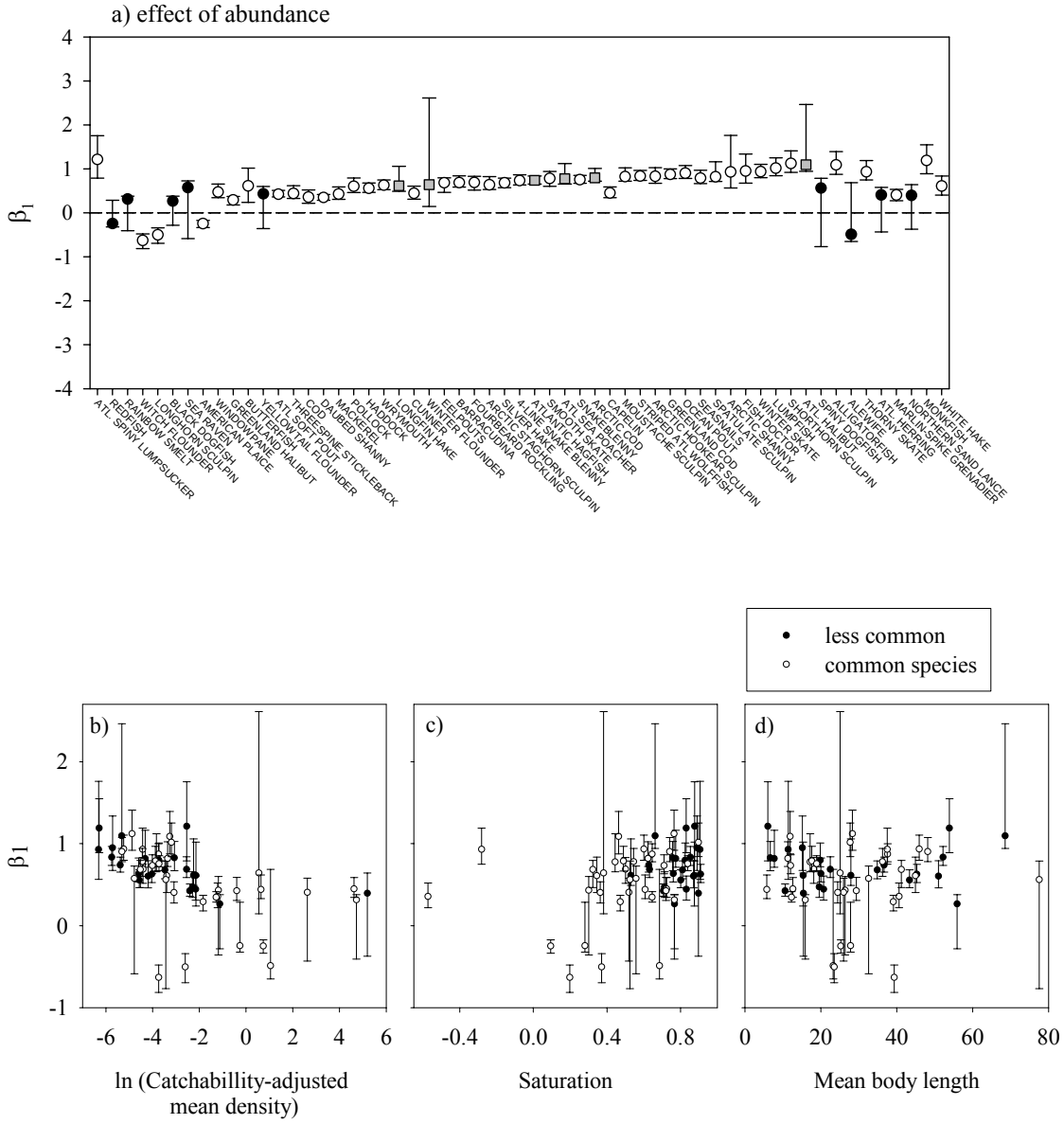


Figure 3.1.2.4. a) SMA results for the effect of abundance on distribution in a model that includes only β_0 and β_1 for those species for which the effect of the index of core preferred temperature availability was not significant (white and black symbols, where the former indicate parameter values that are significantly different from zero at a Type-I error rate of 5%). Grey symbols are the parameter values copied from Fig 2 for those species where a significant effect of core temperature availability was found. The sorting order of results along the abscissa and the range on the y-axis are the ones used in Figure 2. The parameter β_1 was found to be significantly related to b) the catchability-adjusted mean density of fish (million kg·km^{-2}), c) the index of species' habitat saturation and d) the species' mean body length. In panels b-d, species are identified as being relatively common (occurring in >5% of tows 1971–2006) or less common (≤5% of tows).Error bars in all panels are 90% confidence intervals

An exploratory analysis was undertaken to look for factors that may explain interspecies differences in parameter estimates. A general linear model (GLM) was used in each case with β_1 as the dependent variable and (1/Upper-Lower confidence interval) as a weighting factor to downweight cases where the parameter estimate was uncertain. (Note that ability to include this weighting factor was the reason that a GLM was used rather than SMA).Variables potentially affecting the estimate of the effect of abundance (β_1) was the following:

1) The coefficient of variation of $\overline{U_y}$ over all years for the species, since little contrast in abundance would be expected to result in little contrast in distribution, even if an underlying relationship existed between the two;

2) The index of mean preferred habitat saturation ($\overline{O}$);

3) Whether the species was demersal or pelagic;

4) Species body size, as density dependent effects may be predicted to be greater in larger organisms (e.g. Goodwin *et al.*, 2006);

5) Mean species biomass density (kg/km^2), where biomasses are for species and length-specific catchability-adjusted data.

Interspecies differences in the estimated effect of abundance on distribution (β_1) appear to be negatively associated with biomass density and positively related to $\overline{O}$ and body length, though the effect is strongest for the first variable (Table 3.1.2.1, Figure 3.1.2.4b-d). This is true whether the analysis is done on all species or just the more common ones. Jointly, the variables explain about 35–40% of the variability in β_1, depending on whether all species are included in the analysis or not. The result is that densely distributed species are less likely to show strong density dependence or may show negative density dependence (i.e. dispersal at low abundance) (Figure 3.1.2.5). The effect of the two other covariates is weaker, but still consistent with expectation: species that saturate their habitat will show weaker density-dependent expansion since they have little latitude to spread and the strength of density-dependence will be greater in larger-bodied species.

Table 3.1.2.1. Results of GLM analyses aimed at exploring the causes of interspecies differences in the estimated value of the effect of abundance on distribution (β_1). The dependent variable was transformed using exp(β_1) to meet the assumptions of the analysis. Results are for Type III sums of squares.

PARAMETER	*ESTIMATE*	*SE*	*T-VALUE*	*P*
a) all species, all factors				
$\overline{O}$	0.7697	0.2739	2.810	**0.0070**
pelagic	-0.0753	0.2629	-0.286	0.7758
CVabundance	-0.0025	0.0013	-1.857	0.0690
body length	0.0106	0.0049	2.164	**0.0352**
biomass density	-0.0793	0.0322	-2.461	**0.0173**
b) all species, only those factors found to be significant in (a)				
$\overline{O}$	0.4995	0.2347	2.128	**0.0380**
body length	0.0105	0.0046	2.283	**0.0265**
biomass density	-0.1066	0.0277	-3.856	**0.0003**
c) species occurring in >5% of survey tows, all factors				
$\overline{O}$	0.6639	0.3670	1.809	**0.0412**
pelagic	-0.4646	0.4402	-1.055	0.3002
CVabundance	-0.0012	0.0020	-0.592	0.5587
body length	0.0185	0.0083	2.230	**0.0339**
biomass density	-0.1397	0.0498	-2.808	**0.0090**
d) species occurring in >5% of survey tows, select factors				
$\overline{O}$	0.6863	0.3174	2.162	**0.0387**
body length	0.0159	0.0076	2.101	**0.0441**
biomass density	-0.1222	0.0374	-3.269	**0.0027**

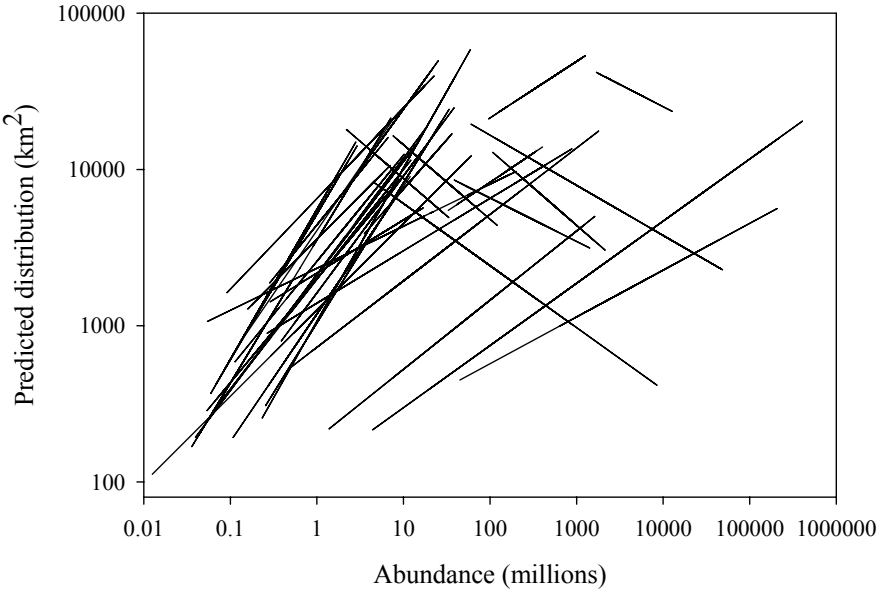


Figure 3.1.2.5. Abundance and predicted distributional area for species occurring in >5% of tows in the sGSL survey 1971–2006.

3.1.3 Discussion

In this analysis considerable evidence was found for an allometric relationship between abundance and distribution across a diversity of marine fishes in a single ecosystem. While this result in itself brings us no further towards understanding the exact mechanism(s) responsible (Shepherd and Litvak, 2004), it does confirm the relevance of the aforementioned consequences as it relates to commercial CPUEs, indices of abundance based on presence-absence, etc.

By including effects of changes in habitat quality as measured by the availability of waters of preferred temperature, an attempt was made to find evidence for density-dependence as might be expected under an IFD. As formulated here, little such evidence was found despite observing temporal variability in habitat quality and finding that changes in community composition in the sGSL over 1971–2005 appear, in part, to be related to changes in bottom-water temperatures (Benoît and Swain, *submitted*). There are several potential reasons for this. Firstly, environmental changes of the magnitude observed may affect species abundance, which in turn affects distribution, or it may affect the two simultaneously, but it does not affect the abundance-distribution relationship. Secondly, other dynamic factors, such as prey availability, may have an overwhelming effect on density-dependence, relative to the direct effect of thermal condition. Thirdly, the model used may be wrong: a wrong assumption regarding the immediacy of environmental change effects, an environmental effect on some but not all demographic classes in the population, an impact on density-dependence that is more complex than the linear one assumed here, etc. Finally, the survey area may not cover the entire population distribution for some species. Given that changes in distribution are predicted to be strongest at the margins of the population's distributional area, environmental effects may not be detected if a substantial portion of that margin is external to the survey area.

In their study of abundance-distribution relationships of marine fishes on the Scotian Shelf (neighbouring ecosystem to the sGSL) Fisher and Frank (2004) found a smaller proportion of species with significant allometric relationships as compared to the present study. This is true even when less frequently captured species are eliminated from the present analysis, as they had done. Two possible reasons are proposed for this discrepancy. The first is statistical.

Fisher and Frank (2004) used as a measure of abundance the mean annual catch of a given species calculated using only tows that captured it. Distribution was quantified such that it was effectively proportional to the minimum number of tows catching 90% of species abundance. If properly interpreted from their paper, this equates to

$$\overline{U_y} = \frac{1}{T_{y,non-zero}} \cdot \sum_{t=1}^{T_y,non-zero} u_{yst} \qquad \text{and} \qquad D_y \propto 0.9 \cdot T_{y,non-zero}$$

An artifact of using these measures is that distribution scales to abundance as $\overline{U_y}^{-1}$ if distribution area is random with respect to abundance. Therefore even if a positive underlying relationship indeed exists between abundance and distribution, this inverse scaling will make it more difficult to detect a statistically significant effect.

The second possible reason explaining the discrepancy between their results and the ones presented here may be the nature of the fisheries in both areas. In the sGSL, many of the commercial finfish fisheries target moving fish, either migratory or spawning aggregations. One could conjecture that such a fishery is less likely to lead to localized depletions as captured fish may originate from a diversity of locations. A region-wide density-dependent relationship may therefore be expected. In contrast, the demersal fish fisheries on the Scotian Shelf are believed to potentially result in spatialized depletion (Shackell *et al.*, 2005) as many of the species in that ecosystem tend to undergo limited migrations compared to those inhabiting the Gulf. If rates of recolonization are low relative to the intensity of fishing and fishing effort is heterogeneously distributed, a species' distribution may reflect a combination of density-dependent changes and local depletion. A region-wide density-dependent relationship is therefore more likely to be obscured or disrupted, or will change over time (as proposed by Fisher and Frank (2004) in a conceptual model). This of course is mere conjecture at this stage but may be an interesting point for further study. A comparative analysis of species abundance-distribution relationship among ecosystems where localized fishery-induced depletion effects are more or less likely may contribute to a better understanding of the mechanisms underpinning the empirical relationships and would constitute a test of the aforementioned conceptual model.

A better understanding of the mechanisms behind the empirical relationships is likely to come only from detailed analyses of vital rates and proxies of fitness (a non-trivial task) or from interspecies/inter-system comparisons. While the latter have provided useful insight for other taxa such as birds (e.g. Webb *et al.*, 2007), species and size-specific differences in catchability of trawl-sampled fishes complicate this task. Some generic corrections to roughly adjust for order of magnitude differences were applied here. The results do provide some consistency with theory and empirical results in other taxa: the slope of the allometric relationship appears to decrease with catchability-adjusted density and with the degree to which a species saturates its available environment. However, because the catchability adjustments are approximate at best, it will be very difficult to robustly undertake these comparisons.

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4 Essential fish habitat

4.1 Introduction

WGFE has examined issues regarding Essential Fish Habitat (EFH) for several years now. The concept of EFH was introduced in the mid 1990s in the USA and, in 1996, the USA Congress added habitat conservation measures to the Magnuson-Stevens Fishery Conservation and Management Act. The USA Congress defined EFH as "those waters and substrate necessary to fish for spawning, breeding, feeding, or growth to maturity", where waters are "aquatic areas and their associated physical, chemical, and biological properties that are used by fish and may include aquatic areas historically used by fish where appropriate"; substrate "includes sediment, hard bottom, structures underlying the waters, and associated biological communities"; necessary means "the habitat required to support a sustainable fishery and the managed species' contribution to a healthy ecosystem"; and "spawning, breeding, feeding, or growth to maturity" covers a species' full life cycle.

In 2003, WGFE was asked to: "Review the concept of essential fish habitat and consider what specify essential fish habitat for individual species or stocks", and suggested that the types of site that may be regarded as EFH for particular species would include

- Breeding, spawning and parturition grounds
- Nursery grounds (for post-larvae, neonates and juveniles)
- Shelter and natural refuges
- Feeding grounds
- Migratory corridors

Furthermore, the grounds utilised by those species that exhibit high habitat specificity or are endemic to restricted locations may also be regarded as EFH (see Section 4 of ICES 2003).

In the following years, the ToRs regarding fish habitat concentrated on the study of habitat requirements of "commercial, threatened or rare species" and has done so by focusing on various gadiform and pleuronectiform fishes (see Section 5 of ICES 2004a) as well as selected deep-water species, the fishes of Le Danois Bank, and the fish communities in Canadian waters (including the Grand Banks and Labrador Shelf) and in the Barents Sea (see Section 3 of ICES 2005a). The following year, WGFE briefly examined fish-benthos interactions, nursery grounds of selected North Sea fishes and the occurrence of a potential spurdog pupping site in the Irish Sea (see Section 5 of ICES, 2006a).

With regards EFH, the 2007 TOR (d) for WGEF was to:

i) study the functional coupling between fish and their biotic and abiotic environment to identify the characteristics of essential habitats for fish species (and life-history stages) of interest. Examine the distributions of demersal and pelagic fish in relation to habitat properties, and identify those ecological, physiological and behavioural components that may affect the distribution of fish (see Section 4.3).

ii) Estimate the cumulative area representing (1) the core abundance of eggs, larvae and nursery areas of commercial species; (2) the survey abundance of all fish species completing their total life cycle within a particular management area as a hypothetical implementation of essential fish habitat (EFH) protection (see Section 4.4).

iii) Explore the utility of using IBTS and other national data to identify the broadscale distribution of nursery grounds of commercial and vulnerable fish species in the ICES area (see Section 4.2).

iv) Overlay fish distribution maps with habitat and environmental layers for available data as an exploratory exercise for developing hypotheses on mechanisms (see Section 4.5).

The concept of EFH is based on those grounds that are "necessary", which to some extent is impractical for management, as this may potentially cover extensive areas of a species' overall distribution. Hence, it is important to recognise that there may be some areas of more critical importance to certain life-history stages of particular species.

Essential fish habitat may be limited in space/time, and should be critical to population dynamics (e.g. through optimal spawning success, or through enhanced growth, survivorship or condition etc.). When there is a concentration of particular life-history stages in a spatially/temporally identifiable location, with important biological processes occurring, and which may be impacted by human activities, then such areas could be considered as important habitats within any spatial management. The spatial/temporal variability in EFH may be linked with the life-history strategy, so some pelagic species may have extensive spawning grounds, of which some areas may be of greater importance to recruitment in various years, and the most important areas may not be reliably identifiable. In contrast, some species may have more defined spawning grounds (e.g. in the case of certain demersal egg-layers), though it should be recognised that although these sites may be important in the short-term, they may also change over time.

For the purposes of the present report, WGFE considered that nursery grounds could be broadly divided into primary nursery grounds and secondary nursery grounds (see Griffiths, 2002; Manderson *et al.*, 2003; Thorpe *et al.*, 2004; Yokota and Lessa, 2006). Primary nursery grounds are those areas in which the earliest, actively free-living life-history stages can successfully recruit to, whether as post-larval stages or pups after parturition. These areas should offer a suitable physical environment and sufficient food availability to promote high growth rates, and should have sufficient shelter/protection from predators so as to increase survivorship. Larger juveniles may expand their habitat and/or move to new areas (secondary nursery grounds) as they attain a larger size before fully recruiting to the adult population.

Fish species have a wide variety of life history patterns, which determine their population dynamics (Petitgas *et al.*, 2006), and these contrasting life-histories will result in different spawning an nursery ground requirements. For example herring *Clupea harengus* has well defined spawning habitats with the eggs deposited on clean gravel etc., whilst sardine *Sardina pilchardus* may spawn over broader areas. Other species may inshore for spawning, for example the garfish *Belone belone* spawn in shallow water and attach its sticky eggs to seagrass or rocks, black gobies *Gobius niger* deposit their eggs on empty shells and are guarded by the male, and the sticky eggs of corkwing wrasse *Symphodus melops* are deposited in a nest made of sea weed, also guarded and cleaned by the adult male. Most fish species have a larval stage that develops in the water column, and are often assumed to be planktonic (i.e. passively dispersed with currents and only capable of vertical behaviour). Leis (2006) has recently concluded that many species of demersal living fishes are nektonic, i.e. having an active horizontal swimming behaviour, during much of their larval life. Long-term variations in the life cycle have been observed in some species (e.g. Salvanes *et al.*, 1994; Vaz and Petitgas, 2002; Petitgas *et al.*, 2006). Additional research on life cycles and functional models are needed for improved understanding of the mechanisms of egg/larval dispersal, post-larval settlement and the biotic and abiotic characteristics of important nursery grounds that may represent EFH.

4.2 Broadscale distribution of nursery grounds

There are several broadscale trawl and beam trawl surveys operating in the ICES area, and these can be used to show the broadscale distributions of juvenile fish of commercial and non-target fish species. Whereas North Sea surveys have standardised gears, which facilitates the

broadscale mapping of juvenile stages (see Section 4.2.2), westerly IBTS surveys use a variety of gears, which will have different catchabilities for many species. In recent years, IBTSWG have mapped the annual distributions and relative abundance of juvenile cod (<23 cm), haddock, hake and whiting (<20 cm), blue whiting (<19 cm), mackerel (<24 cm), horse mackerel (<15 cm), herring (<17.5 cm), and plaice (<12 cm) in western surveys (see Section 4 of ICES, 2005b, 2006b).

Although IBTS surveys cover extensive areas of the North Sea and eastern Atlantic, these surveys often focus on offshore areas, and may not sample some of the shallower inshore and estuarine nursery grounds that are important for some species of fish. It should also be noted that these surveys are typically designed for gadiforms, and so the type of gear, and timing and location of the surveys in relation to recruitment processes may not be optimal for the sampling of juveniles of other species. In the case of pelagic species, as these species can have a high variability in trawl catches, there could be a perception of nurseries occurring over sporadic areas which may not reflect the true situation. Furthermore, it should also be recognised that for some taxa (e.g. *Callionymus*), the accurate identification of juveniles of morphologically similar species can be problematic.

4.2.1 Identification of nursery grounds and other important grounds of pelagic fishes (WGLESP)

The life cycle patterns of several commercial pelagic fishes have been examined by ICES-SGRESP (2004c, 2005c, 2006c). SGRESP recognised the importance of documenting the spatial distribution of life history stages as a first step in understanding potential environmental forcing on fish spatial distributions and recruitment dynamics. SGRESP was concerned by cross-mapping meso-scale oceanographic features with fish habitats. The life cycle spatial organisation was reviewed for following species/stocks: North-east Atlantic mackerel, North-east Atlantic blue whiting; North Sea, Baltic and Irish herring stocks; sardines in Biscay-Iberian waters; North Sea sprat and Biscay anchovy. The template used was as below:

ID card for population

- Life history traits
- Spawning time/habitat
- Adult growth: time/habitat/co-occurring species
- Nursery grounds
- Feeding: predation mode/prey species
- Adult migrations
- Larval drift and nursery areas
- Long term trends
- Current status
- Potential environmental influences
- Knowledge gaps
- References
- Schematic map of the life cycle
- Schematic map of oceanographic features
- Meta information on survey data (for each species a list of recurrent surveys (e.g. bottom trawl, acoustic and ichthyoplankton surveys) is given with a description of the database, contact person and availability of data).

4.2.2 Broadscale distributions of nursery grounds from IBTS surveys

One approach towards identifying the distribution of nursery grounds of commercial and vulnerable fish species in the ICES area is to plot the densities of the recruits on a haul basis. The distributions of juvenile cod, haddock, whiting, herring, plaice, mackerel and sole in the North Sea were illustrated in an earlier report (see Section 5.4 of ICES, 2006a). The present report illustrates the distributions of juveniles of the species listed in Table 4.1, which represent species of commercial and conservation interest. The sizes for which they are illustrated are also given. These sizes were selected after examination of length-frequency histograms. Providing that sufficient data were available, the length cut off should equate with 0-groups, though for more infrequent species, the length cut was to some extent arbitrary and designed to maximise the number of smaller individuals for which the distribution can be mapped. The length-frequency distributions used were the average of the catch numbers by lengths for combined data of the three IBTS-surveys and seasons (Q3 and Q4) as described below.

Data were extracted as catch numbers per hour on a haul basis from the ICES DATRAS database and originates from all IBTS-surveys that are stored in this database: The North Sea International Bottom Trawl Survey (NS-IBTS), covering the North Sea, Skagerrak and Kattegat, the Scottish West Coast Survey (SGFS), carried out on the Hebridean shelf, and the French EVHOE Bottom Trawl Survey that covers the Celtic Sea and the Bay of Biscay. These three surveys together cover a large part of the European Continental Shelf, however, it is highly desirable that data of more IBTS-surveys soon become available through DATRAS in order to fill in gaps (e.g. English Channel, Irish Sea, Iberian shelf) and to expand the plotted area. A preliminary examination of some of the data from the English westerly IBTS survey is discussed in Section 4.3.5.

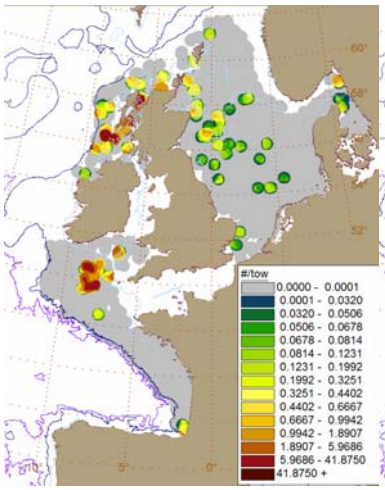
To avoid seasonal variation between areas within the plots, only data were used from the season when all surveys were carried out, i.e. the 3rd and 4th quarter. The used time series were 1991–2006 for the NS-IBTS and the SGFS, and 1997–2006 for EVHOE.

The density distribution of 'young fish' was plotted as the catch numbers per hour of all hauls taken during the different IBTS-surveys during 3rd and 4th quarter for the years 1991–2006 (see Figure 4.1).

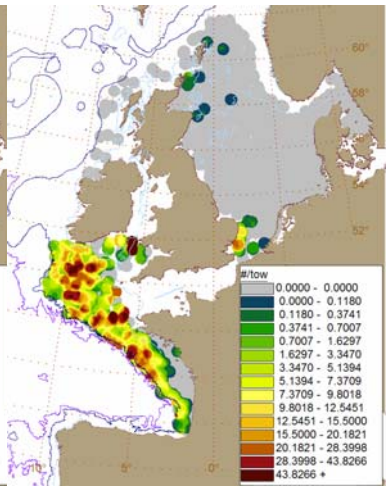
Table 4.1. Size of the length split to select 'young fish'.

SPECIES		LENGTH SPLIT	LESS THEN (CM)
Squalus acanthias	spurdog	Arbitrary	45
Scyliorhinus canicula	lesser spotted dogfish	Arbitrary	30
Clupea harengus	herring	0-group	24
Sardina pilchardus	pilchard	0-group	17
Sprattus sprattus	sprat	0-group	9
Engraulis encrasicolus	anchovy	0-group	11
Lophius piscatorius	anglerfish	0-group	25
Gadus morhua	cod	0-group	23
Melanogrammus aeglefinus	haddock	0-group	20
Merlangius merlangus	whiting	0-group	20
Pollachius virens	saithe	0-group	22
Trisopterus luscus	bib	0-group	13
Merluccius merluccius	hake	0-group	20
Zeus faber	john dory	0+1-group	21
Trachurus trachurus	horse mackerel	0-group	14
Mullus surmuletus	striped red mullet	0+1-group	18
Pleuronectes platessa	plaice	0-group	18
Solea vulgaris[1]	sole	0-group	16

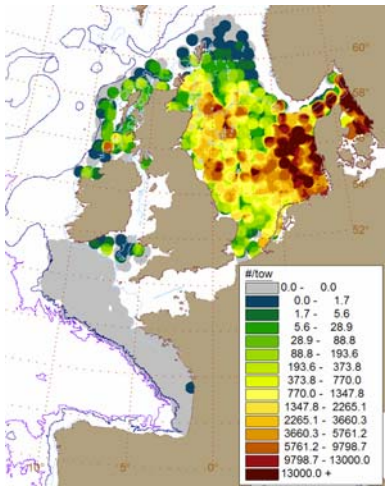
[1] **Data for sole was extracted from ICES-DATRAS by TSN-code 173001 (= *Solea vulgaris*), which means that data for the same species that is recorded differently as *Solea solea* with TSN-code 173002 were not included in the analysis.**



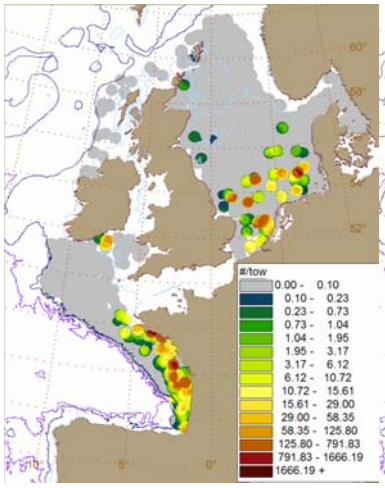
Scyliorhinus canicula



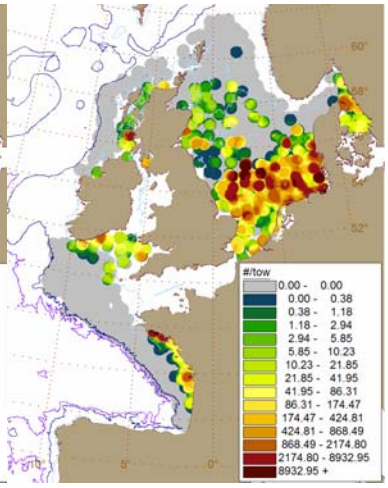
Squalus acanthias



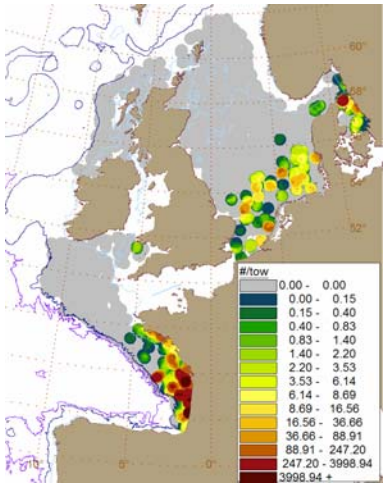
Clupea harengus



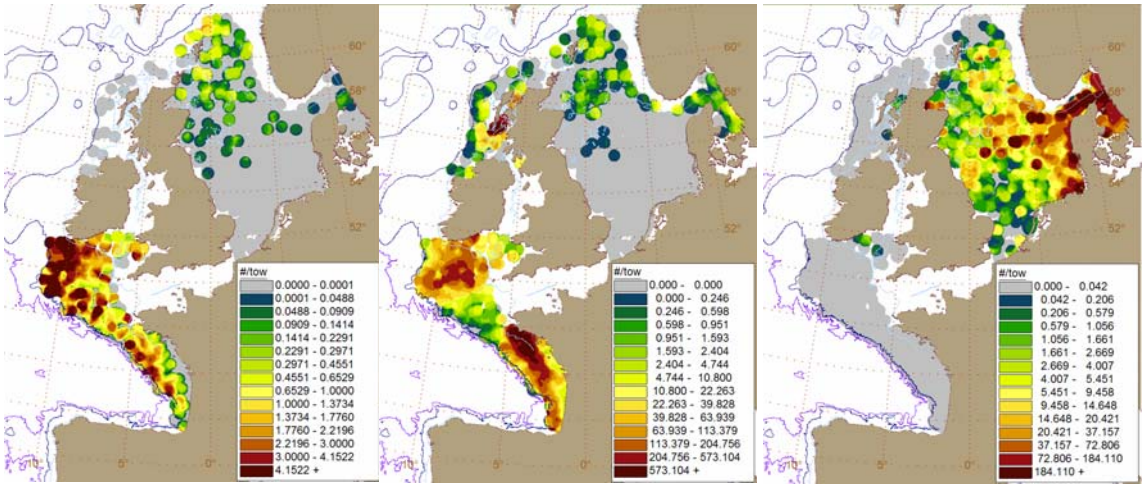
Sardina pilchardus



Sprattus sprattus

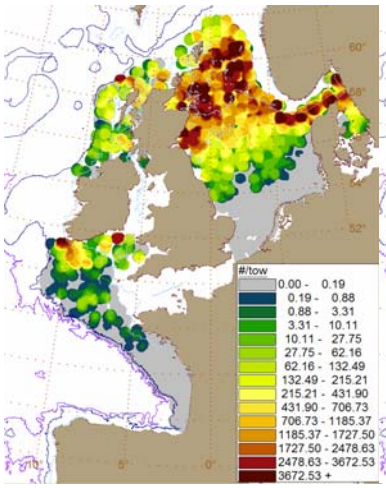


Engraulis encrasicolus

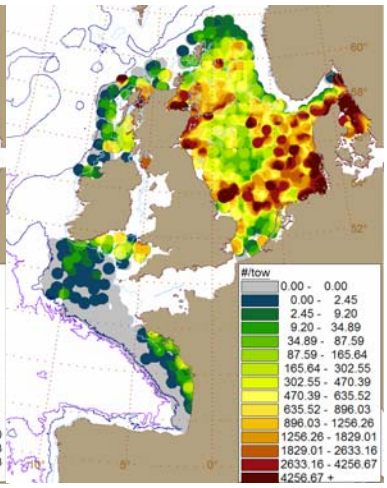


Lophius piscatorius *Merluccius merluccius* *Gadus morhua*

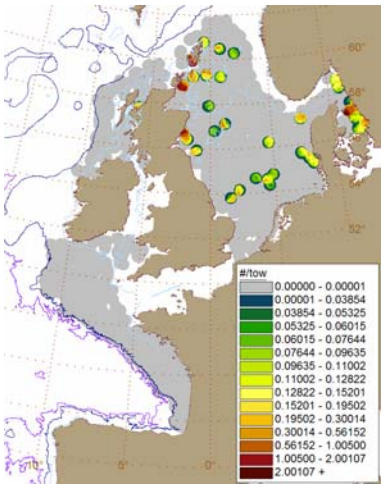
Figure 4.1. Numbers of young fish by haul. Data are derived from several IBTS surveys during Q3 and Q4 1990–2006 (see text). (Grey area: surveyed area.)



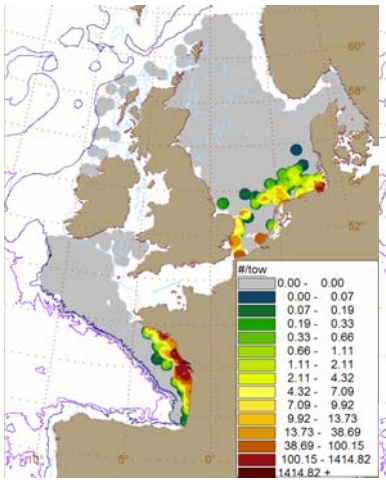
Melanogrammus aeglefinus



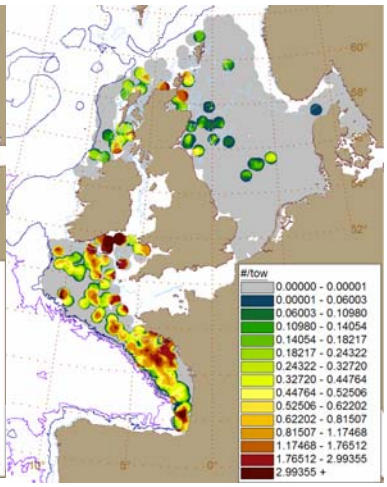
Merlangius merlangus



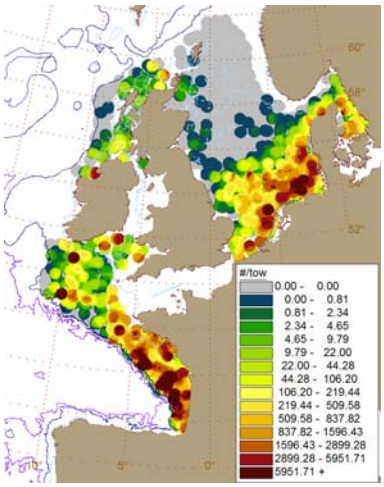
Pollachius virens



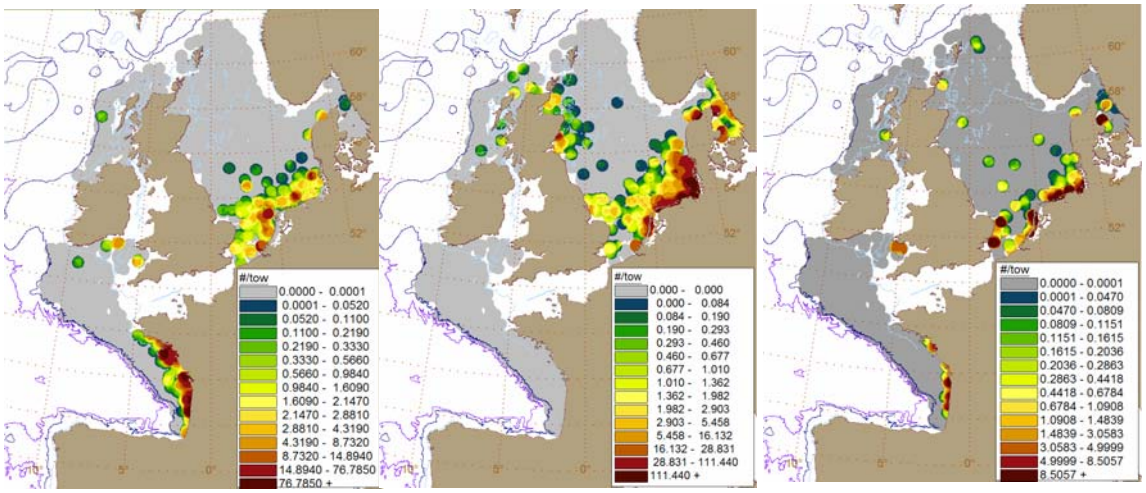
Trisopterus luscus



Zeus faber



Trachurus trachurus



Mullus surmuletus *Pleuronectes platessa* *Solea vulgaris*

.

Figure 4.1 (continued): Numbers of young fish by haul. Data are derived from several IBTS surveys during Q3 and Q4 1990–2006 (see text). (Grey area: surveyed area.)

4.2.3 The comparative utility of IBTS surveys and beam trawl surveys for the identification of nursery grounds: A case study of the Bay of Biscay

In this case study, the use of IBTS data for identifying nursery area for a selection of species that are known to have coastal nurseries was investigated empirically for the Bay of Biscay. In this area, the Western IBTS survey is carried out in October-November every year. Simultaneously, France has been carrying out a number of dedicated coastal nursery surveys, mainly in river estuaries encompassing a variety of coastal habitats ranging from open shallow muddy estuarine areas under the direct influence of freshwater inflows to semi-enclosed sheltered muddy marsh areas with shellfish-farming, lightly affected by rivers (Gilliers *et al.*, 2006). The nursery locations identified by both types of surveys were compared. Nursery grounds were defined here as the area containing the upper 25% of average density across years for individuals representing the first peak in length-frequency data. Potentially these might not correspond to the same age in the two types of surveys, as the Western IBTS survey does not cover areas shallower than about 30 m and there will be some differences in the size selectivity of the gears. The list of species had to be limited to those caught in reasonable numbers in both types of surveys, thus plaice and flounder had to be excluded.

4.2.3.1 Data

Offshore dataset. The western IBTS survey has been carried out annually in the Bay of Biscay in the fourth quarter (October-November) since 1987 with gaps in 1991, 1993 and 1996 (Poulard *et al.*, 2003). The sampling design is stratified according to latitude and depth (30–600) and a 36/47 GOV trawl with a 20 mm mesh codend liner is used. Haul duration is 30 minutes at a towing speed of about 4 knots.

Coastal dataset. The coastal trawl surveys for juvenile flatfish species have been conducted in the Bay of Biscay from 1980 to 2004 (Le Pape *et al.*, 2003). These nursery-dedicated surveys were carried out from the end of August to the end of October. Early studies have found that this period coincides with the end of the growth phase of juvenile flatfish and that it was a suitable period for their collection, providing consistent estimates for notably 0-group fish

(Dorel *et al.*, 1991). Surveys were carried out in depths ranging from 5 to 25 m using a 2.9 m wide and 0.5 m high beam trawl with a 20-mm stretched mesh net in the codend. Hauls were conducted on homogeneous sediment and depth and lasted 20 min covering a mean area of 4500 to 5000 m^2. All commercial species caught were counted and measured for total length.

4.2.3.2 Methods

In order to identify the youngest age group present in a given survey, length-frequency distributions were investigated. Normal distributions were fitted to numbers-at-length cumulated across years. The selection of the number and location in terms of length of the normal distributions as well as their standard deviation was based on optimal fit using Bayesian Information Criterion (BIC) as described in Fraley and Raftery (2002a and b). All measured individuals were then categorised to a presumed year class, based on the normal distribution they belonged to. Individuals belonging to the first peak were used for deriving average densities on a spatial grid (0.25° x 0.125°) by year. Each grid cell was then averaged across years and maps of quartiles (<25%, 25–50%, 50–75% and >75%) of those average densities were plotted by species.

4.2.3.3 Results and conclusion

Comparison of length-frequency distributions for the IBTS and coastal surveys show that in general the same size range is sampled. However, in IBTS the magnitude of the first peak is smaller than the second for a number of species, while for the same species the opposite is true for the coastal surveys (Figure 4.2). The distributions of 15 species are shown in Figure 4.3.

For common dragonet (*Callionymus lyra*: CALMLYR), however, the size range studied is probably not represented by juveniles only, and may also include some misidentified *C. reticulatus,* which are common in the area. This indicates that for those species the IBTS survey either covers less of the nursery habitat and/or has lower catchability. IBTS surveys also seem to be overlooking the first peak for scaldfish (*Arnoglossus laterna*: ARNOLAT), thereby catching only the larger-sized groups. Note also that the identification of second or third peaks in the length distributions is not always successful. For hake (*Merluccius merluccius*: MERLMER) and whiting (*Merlangus merlangus*: MERNMER) the second peak is close to the first, which means that recruitment density is somewhat underestimated.

Both surveys agree in the location of the nurseries for three out of four flatfish species that are known to have 0+ juveniles at the time of the survey: solenette (*Buglossidium luteum*: BUGLLUT), wedge sole (*Dicologlossa cuneata*: DICOCUN) and common sole (*Solea solea*: SOLESOL). For scaldfish, as the two surveys did not sample the same size lengths, it is likely that they give complementary information on the essential habitats for two age groups. The distribution maps of some other inshore fish species commonly known to have coastal and to some extent estuarine nurseries, also displayed good matches between the two surveys. These species are sand smelt (*Atherina presbyter:* ATHEPRE), sand goby (*Pomatoschistus minutes*: POMOMIN), sprat (*Sprattus sprattus*: SPRASPR) and bib (*Trisopterus luscus*: TRISLUS). Other species shown are sardine (*Sardina pilchardus*: SARDPIL), mackerel (*Scomber scombrus*: SCOMSCO) and red mullet (*Mullus surmuletus*: MULLSUR).

Some additional analyses were carried out to investigate the spatial stability of nursery areas for hake and anchovy (Figure 4.4). While for hake the identified nursery areas are stable between years, corresponding mainly to a muddy area in the centre of the Bay of Biscay, the annual maps for anchovy (*Engraulis encrasicolus:* ENGRENC) clearly show that 0-group individuals are found in different areas in different years. Thus this might indicate that for anchovy the water conditions rather than the bottom habitat type determines nursery locations, or it may simply be a sampling artefact.

In conclusion, this empirical study of the usefulness of the IBTS data for identifying nursery areas points shows that indeed, at least in the Bay of Biscay, nurseries can be identified which are in agreement with those determined using more coastal surveys. As the Western IBTS survey covers the whole shelf it allows to delimit the extent of the nursery areas and to see that in certain cases a large part of the shelf is actually concerned.

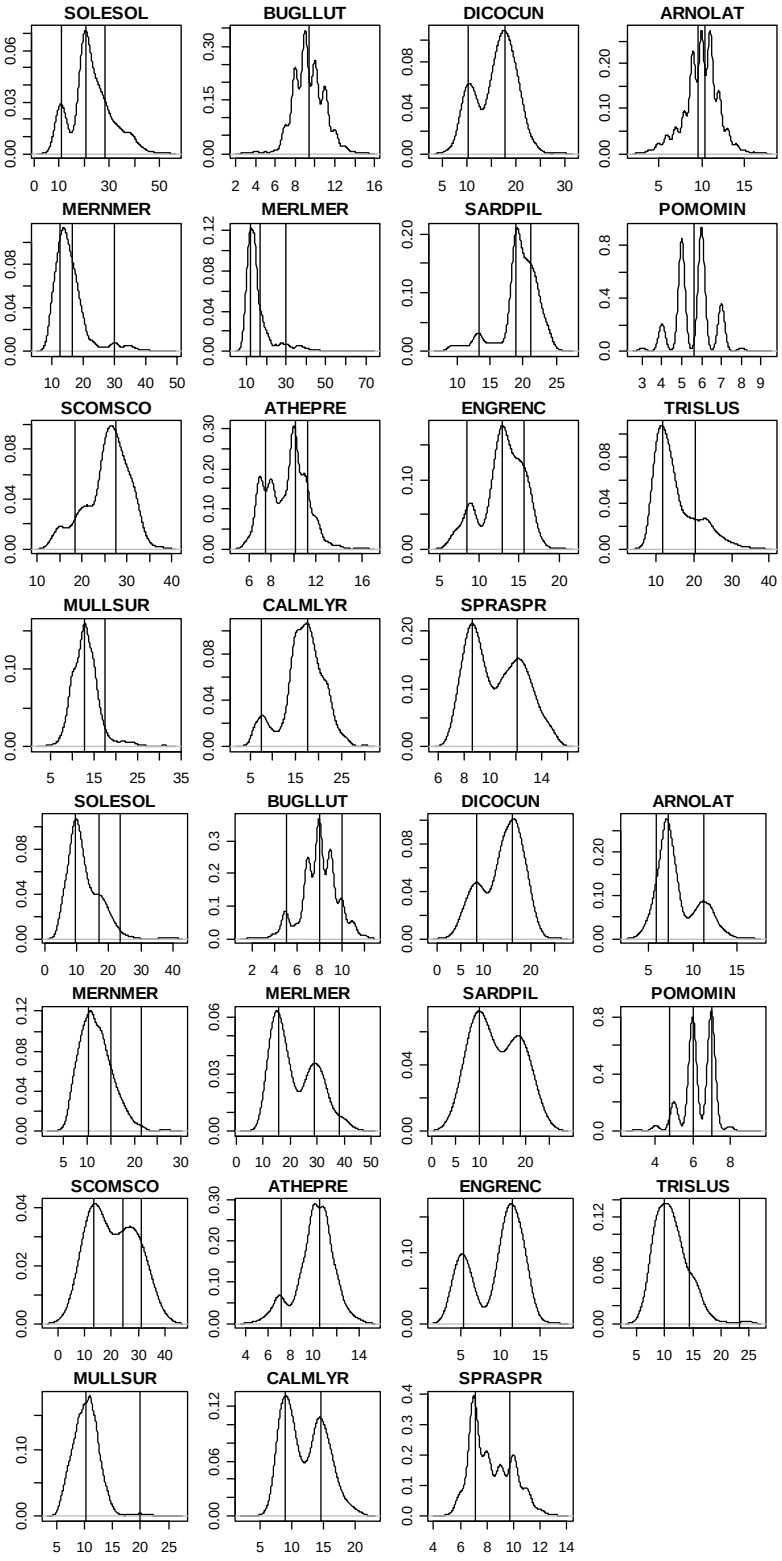


Figure 4.2. Cumulated length frequency distributions in the Bay of Biscay in the Western IBTS (upper) and coastal surveys (lower); Lines indicate identified modes fitting normal distributions. Species (from the top left) are: *Solea solea: SOLESOL; Buglossidium luteum: BUGLLUT; Dicologlossa cuneata: DICOCUN; Arnoglossus laterna: ARNOLAT; Merlangus merlangus: MERNMER; Merluccius merluccius: MERLMER; Sardina pilchardus: SARDPIL; Pomatoschistus minutes: POMOMIN; Scomber scombrus: SCOMSCO; Atherina presbyter: ATHEPRE; Engraulis encrasicolus: ENGRENC; Trisopterus luscus: TRISLUS; Mullus surmuletus: MULLSUR; Callionymus lyra: CALMLYR*; and *Sprattus sprattus: SPRASPR.*

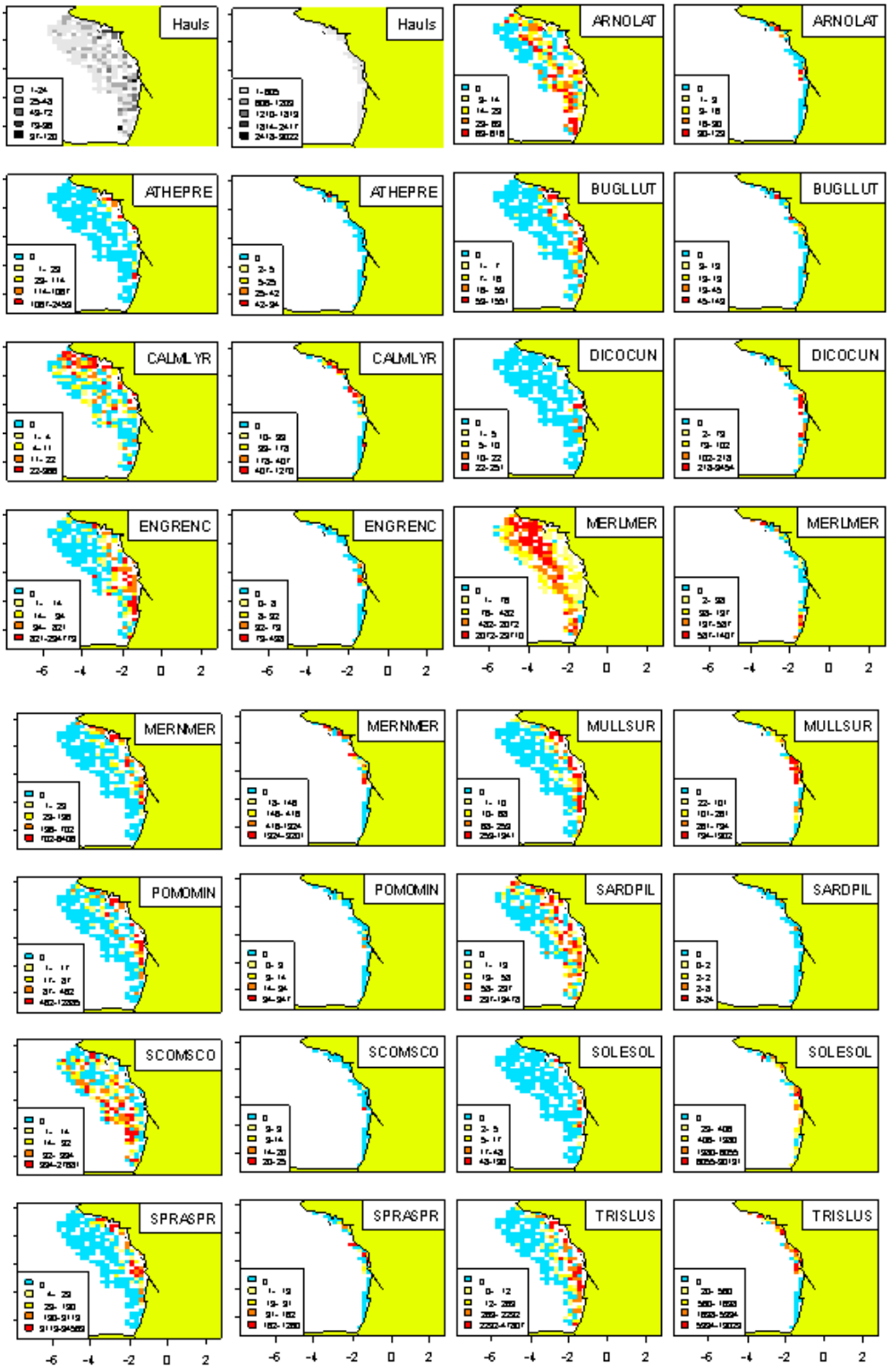


Figure 4.3. Maps of nursery areas in Bay of Biscay using Western IBTS (first and third column) and coastal survey data (second and fourth column). Species (from the top) are: *Arnoglossus laterna: ARNOLAT; Atherina presbyter: ATHEPRE; Buglossidium luteum: BUGLLUT; Callionymus lyra: CALMLYR; Dicologlossa cuneata: DICOCUN; Engraulis encrasicolus: ENGRENC; Merluccius merluccius: MERLMER; Merlangus merlangus: MERNMER; Mullus surmuletus: MULLSUR; Pomatoschistus minutes: POMOMIN; Sardina pilchardus: SARDPIL; Scomber scombrus: SCOMSCO; Solea solea: SOLESOL; Sprattus sprattus: SPRASPR* and ***Trisopterus luscus: TRISLUS.***

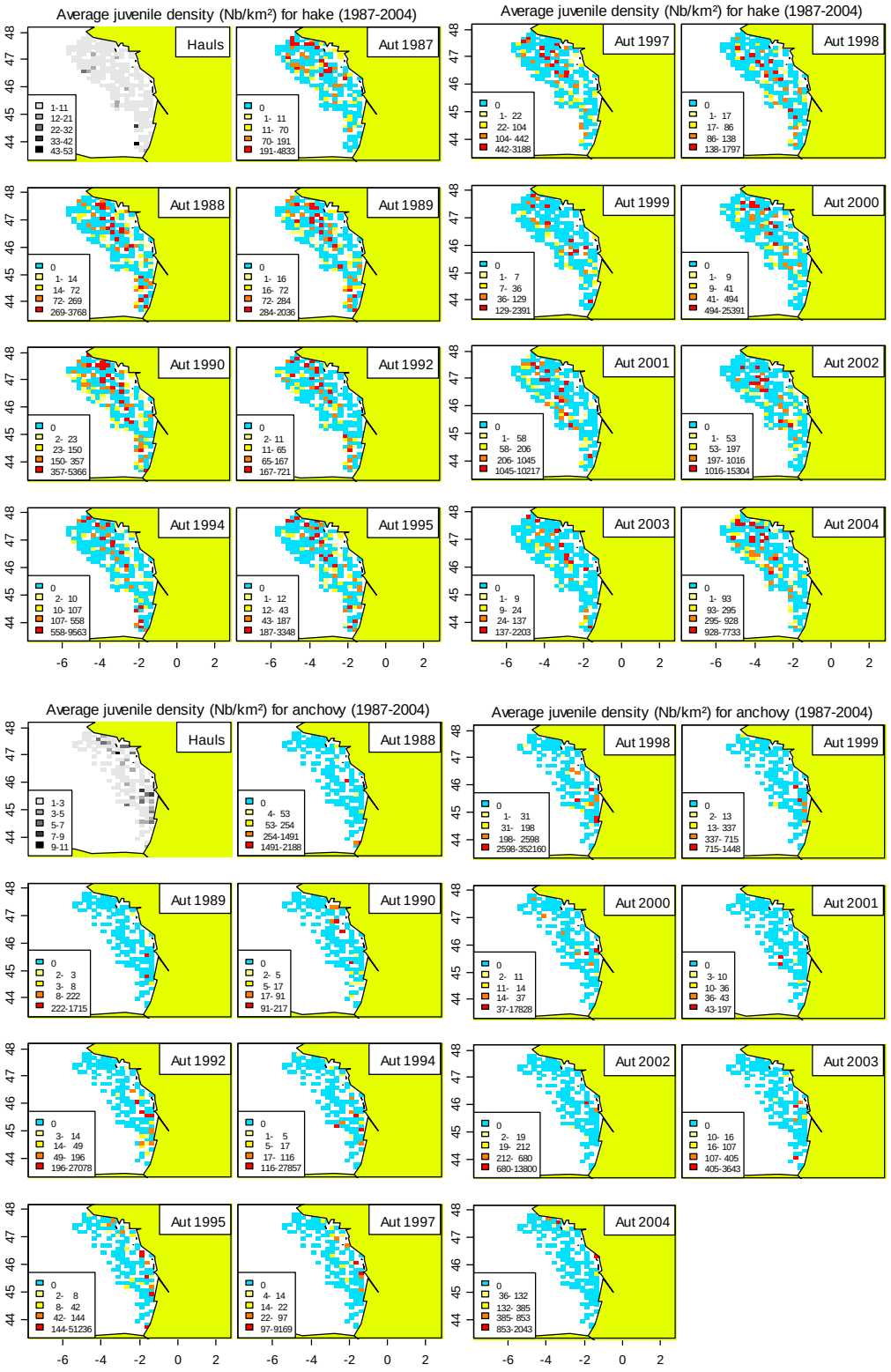


Figure 4.4. Annual maps for nursery areas based on Western IBTS data, hake (top) and anchovy (bottom).

4.3 The relationship between fish and their biotic and abiotic environment

4.3.1 Relationship between fishes and the physical and biotic environment in the eastern English Channel

Valuable marine habitats and living resources can be found in the eastern English Channel (ICES Division VIId) and in 2003 a Franco-British Interreg IIIA two-year project, 'Eastern Channel Habitat Atlas for Marine Resource Management' (CHARM-Phase I), was initiated. This project assessed the status of key commercial fish species as well as describing and modelling their habitats in relation to the environment of the Dover Strait and adjacent waters to support decision-making for the management of essential fish habitats.

Available data and related project's output were compiled into an atlas accompanied with a review of the policy and legal framework for the protection of living marine resources and their habitats in the study area. Fish habitat corresponds to geographical areas within which ranges of environmental factors defined the abundance of a particular species. Habitat modelling was used to relate the spatial distribution of selected fish species to various environmental factors, hence delineating their optimum habitat. This study was based on data obtained from the IFREMER Channel Ground Fish Survey (1988–2004) and the Cefas 4m-Beam Trawl Survey (1989–2004), and included both species abundance and environmental data.

A generic methodology allowing for the modelling of several fish species was developed, including measures of fit and model validation techniques. In brief, habitat suitability modelling based on non-parametric multi-linear quantile regressions was used to relate species abundance to depth, temperature, salinity, seabed stress and sediment type. Backward selection resulted in distribution models that described species affinity with a subset of significant environmental variables that were used to map fish optimum habitats using GIS.

CHARM phase II (a two year project that started in 2006) aims to extend this initial effort to the wider eastern English Channel and will use a process-oriented approach to further explain the results obtained in Phase I, and develop predictive tools for assessing management options. In Phase II, marine species habitat modelling will be undertaken for both juveniles and adults. Then, having gathered relevant fisheries statistics on commercial fish stocks, both habitat models and statistics will serve as forcing variables and inputs to an integrative spatially explicit modelling approach of the marine ecosystem of the area. Two types of models are foreseen: (i) a model of the eastern English Channel ecosystem functioning using mass-balance food web models (Ecopath with Ecosim) and the habitat models previously developed (Ecospace) will be built in order to evaluate management scenarios; and (ii) a conservation plan for the Eastern English Channel using the MARXAN spatial planning software will be developed and should enable the project to identify important sites for conserving biodiversity, whilst using anthropological, economic and legal data to minimise potential opportunity costs. The outputs from the modelled scenarios in Ecopath and MARXAN may ultimately be used to develop a draft management strategy of the eastern English Channel. This work will help elaborate guidelines for the conservation and protection of natural habitats of marine living resources in relation to, for example, climate change scenarios and anthropogenic disturbance.

The CHARM project has resulted in a number of papers (see: Eastwood *et al.*, 2003; Vaz *et al.*, 2004a,b, 2005a,b, 2006, 2007; Martin *et al.*, 2005; Koubbi *et al.*, 2006).

4.3.2 Mapping habitats in the Baltic Sea (BALANCE project)

The EU Interreg III B project BALANCE ("Baltic Sea Management – Nature Conservation and Sustainable Development of the Ecosystem through Spatial Planning") is aiming towards

developing marine spatial planning tools for the Baltic Sea area. One of the main objectives is to provide large-scale maps at landscape and habitat level, to be used in marine management. The project is focused on four pilot areas and is represented by partners from ten countries (www.balance-eu.org).

A central part of the habitat level mapping is to develop methods for identifying essential fish habitats in physically complex coastal areas, which sets high demands on the spatial resolution of the models. This is done for identifying nursery areas of a number of fish species in the northern Baltic Proper, which is characterized by a vast archipelago. Most fish species in the area are dependent on shallow coastal areas for spawning. The models are developed for dominating coastal fishes that are dependent on shallow coastal areas for spawning, such as perch (*Perca fluviatilis*), pike (*Esox lucius*), zander (*Sander lucioperca*) and roach (*Rutilus rutilus*). The essential fish habitats identified will be used within Balance in the assessment of ecological coherence of the Natura 2000 network. The models will also be used in national and regional management, for identifying areas in need of protection or habitat restoration.

The models are developed by estimating the statistical relationship between young (0+) fish occurrence and environmental variables, based on field data collected along major environmental gradients. Generalized additive models (GAM) are used for constructing the statistical explanation models. Habitat maps are produced by projecting the results to the whole study area using full coverage maps of the identified key environmental variables in GIS. The most important variables for predicting habitat quality have been identified as vegetation coverage, wave exposure, water depth and water transparency. The models spatially identify the realized ecological niche of the species with respect to the parameters studied. The precision of the predictions will further on be validated using external ground-truthing data (Figure 4.5).

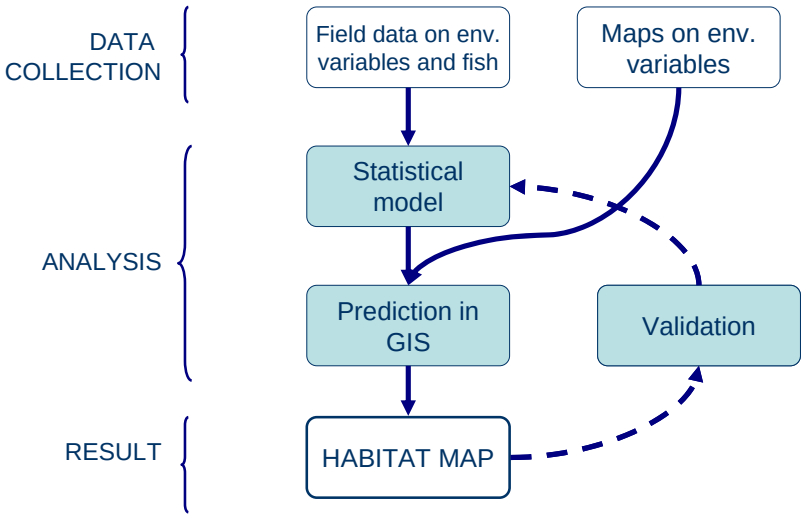


Figure 4.5. Principal steps in mapping of essential fish habitats, as applied within the Interreg IIIB project Balance.

There is an increasing demand for maps describing the spatial distribution of sensitive coastal areas to be used in marine planning, and for many species/habitats GIS modelling is the only feasible way of obtaining maps with large spatial coverage. In many areas, existing survey data might be useful for producing coarse GIS models. However, it is worth pointing out that the data requirements for spatial models differs in a number of ways from that in traditional ecological studies, and a few aspects should be considered before applying the concept to existing data sets. It should be assured that the data set to be used represents the proper

environmental variables, so that the relevant potential mechanisms are captured in the statistical explanation models. When producing large-scale maps the possibility of regional differences in species-environment relationship should also be considered. Also, the sampling design should comprise entire environmental gradients of governing variables, rather than only the optimal parts with regard to the target species and life stage. Many of the datasets collected in current monitoring programs do not meet this criterion, as they are often targeted at core areas of the distribution. In other cases, only certain habitat types have been included in the survey, as the sampling methods used are restricted for example to certain substrate types.

In order to make satisfying predictions of species/habitat distributions, it is essential that available maps of the explanatory environmental variables have an adequate spatial resolution as well as a spatial extent covering the whole areas of interest. A lack of high quality maps of environmental variables, and in some cases public availability to existing maps, is currently a major impediment to successful marine habitat mapping, and an important focus area for future development.

4.3.3 Distribution of demersal fish in the Celtic Sea and Bay of Biscay with respect to temperature and depth

4.3.3.1 Long-term hydrography of the Bay of Biscay

Currently, only an interannual hydrography of the Bay of Biscay is available, although an estimation of annual trends by month and for the whole year and by depth are in process. It covers the southern Celtic Sea up to 50°N. This hydrography, shows that the western IBTS survey in the Bay of Biscay is carried out at a period of high temperature. In other words, the November bottom temperature in the Bay of Biscay is higher than the annual mean (Figure 4.6). In November, bottom temperatures are higher in the south, and a lens of colder water occurs in the southern Celtic Sea.

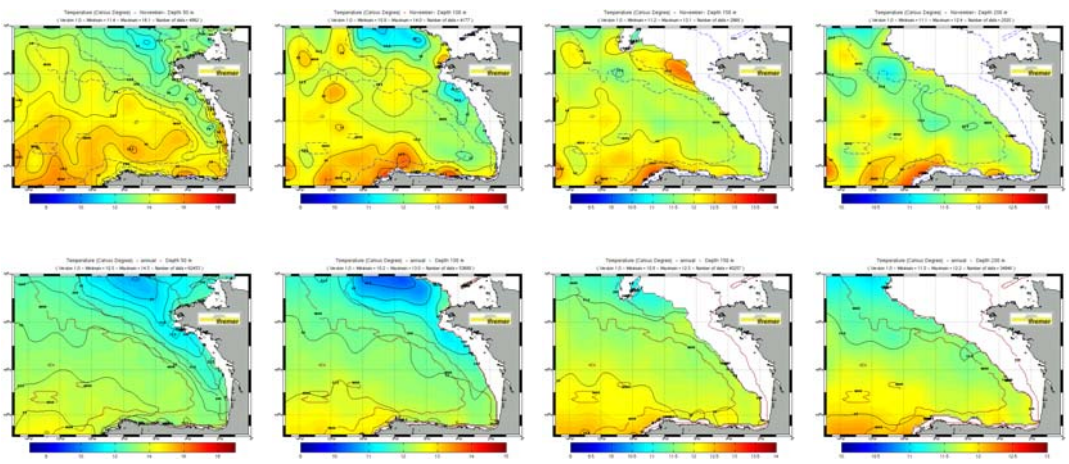


Figure 4.6. Hydrography of the Bay of Biscay, long term average (mean over 1862–2006, most data being from 1960–2006), at the time of Western IBTS survey (November, top) and annual mean (bottom) at standard depths 50, 100, 150 and 200 m. Maps are available from IFREMER (http://www.ifremer.fr/climatologie-gascogne/).

4.3.3.2 Environmental conditions occurring during the western IBTS survey

The Bay of Biscay is defined as the area south of 48°N (ICES Division VIIIa,b) and the Celtic Sea as the area North of 48°N (ICES Divisions VIIg,h,j). Time series of bottom temperature and salinity per depth were compiled based on CTD data from the survey for the years 1992–2005 (Figures 4.7–4.8). Data from shallow areas were removed due to data scarceness.

In the Bay of Biscay, at 30–75 m, temperature observed during the survey has varied without trends. At 75–125 m and 125–175 m, there is a trend for temperature to be higher in the middle of the time series (1998–2002) and lower at either end (Figure 4.7). Salinity displays a more complex pattern with higher salinity at all depths in 1992, 1997, and 1998 and, to a lesser extent, at the end of the time series (Figure 4.8). In the Celtic Sea, the increase observed at the start of the time series, might represent actual mean temperature of the tows carried out but it is more likely to be an effect of change in the geographical area of the survey than an actual feature. From 1997, the recorded temperature and salinity have varied with a consistent pattern over depth ranges and there is correlation between temperature and salinity.

4.3.3.3 Fish distribution *versus* depth

Species distributions with respect to depth have been analysed for the period 1987–2006 (Figure 4.9). Note that the sampling scheme did not cover the Celtic Sea before1997. The study was restricted to the shelf (ca. 30–200m deep). Outside this range sampling intensity has varied over time; however, for some species the depth distribution is truncated. For example, this effect is strong for bluemouth (*Helicolenus dactylopterus*). The depth distribution of some species varies between the Bay of Biscay and the Celtic Sea. For example, hake has a shallower distribution in the Bay of Biscay (median at about 75 m compared to >100m in the Celtic Sea), due to the large nursery area on the Biscay shelf. The deeper distribution in the Celtic Sea might result from a higher proportion of larger individuals. No temporal trend in the depth distribution of hake is visible. Contrary to hake, haddock is distributed deeper in the Bay of Biscay (median over 100 m compared to 60–70 m in the Celtic Sea). In the Bay of Biscay, haddock is limited to the Northern part and small numbers are caught by the survey. In the Celtic Sea, the depth distribution may have increased from 1997 to 2001 and decreased afterwards. Bottom temperatures recorded during the cruise were low in 2000–2001 so that the changes in depth distribution may be due to other factors. For whiting and bib (*Trisopterus luscus*), only the distributions in the Bay of Biscay should be considered. No clear trend appears for these species, however, the depth range of both may have expended toward deeper areas over the time series. Poor cod (*Trisopterus minutus*) is abundant in both the Bay of Biscay and Celtic Sea. The depth distribution seems to have varied over time in both areas without trend nor consistency between the two areas. Bluemouth (*Helicolenus dactylopterus*) appears in significant numbers in both areas. John dory (*Zeus faber*) is distributed over the full shelf in both areas with no time trend in depth distribution.

For the pelagic species (anchovy, herring, sprat, mackerel and sardine) numbers caught in the Celtic Sea are too small for meaningful analysis. In the Bay of Biscay both mackerel and sardine have been caught deeper after 1997. For mackerel this coincides with an increase in estimated population abundance (Figure 4.10). However, trends based on a survey using bottom gear may not be representative for pelagic species.

The distribution of red mullet (*Mullus surmuletus*) seems to have expanded to deeper waters since 1998–1999, which might be have been caused by increasing population abundance, though no clear trend appeared (Figure 4.10). No variation in depth distributions is observed from this graphical analysis for cod, white-bellied anglerfish (*Lophius piscatorius*), lesser spotted catshark (*Scyliorhinus canicula*), spurdog (*Squalus acanthias*), thornback ray (*Raja clavata*), bluemouth (*Helicolenus dactylopterus*) and boarfish (*Capros aper*). For the later species, the abundance has been increasing over the survey time series (Blanchard and Vandermeirsch 2005).

4.3.3.4 Fish distribution *versus* bottom temperature

Some clear results appear for temporal changes in distributions with respect to bottom temperature (Figure 4.11). Several benthic (monkfish, lesser spotted catshark, spiny dogfish, thornback ray), but also hake and John dory were caught in warmer temperatures in warmer years and in colder temperatures in colder years. This suggests that these species kept the same geographical distribution over time and did not move in response to changing environmental conditions. The trend for boarfish seems to be slightly out of phase compared to the other species. Boarfish was caught in water about 1°C warmer in 2000–2003 than at the start and at the end of the time series. Shallow water species (e.g. whiting, bib, anchovy, sprat) were caught in more varying water temperatures. Mackerel in the Bay of Biscay in November seems to be restricted to a much narrower temperature range than other pelagic species. Furthermore detailed analysis is required to confirm these preliminary results.

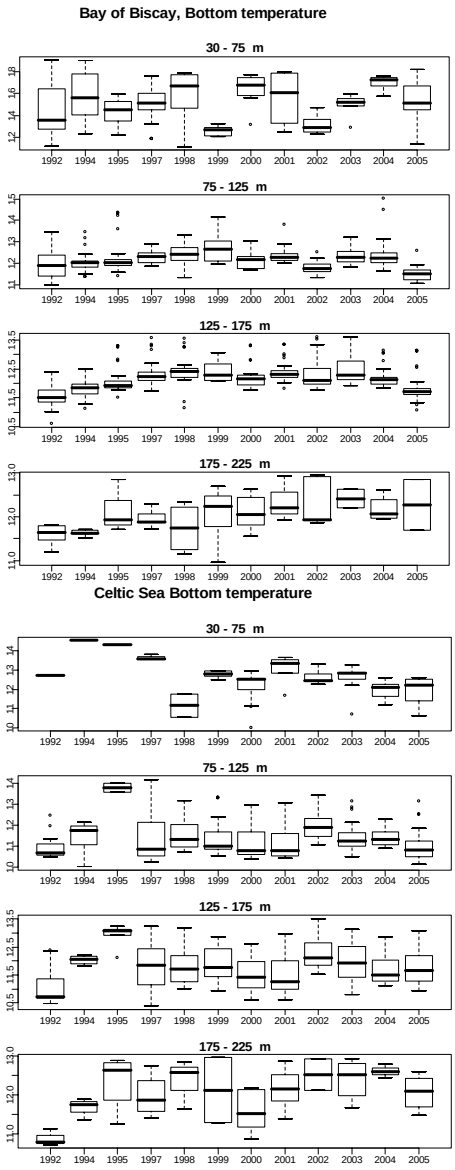


Figure 4.7. Variability of bottom temperature at the location of survey hauls (Western IBTS) in the Bay of Biscay and Celtic Sea in November-December from CTD data.

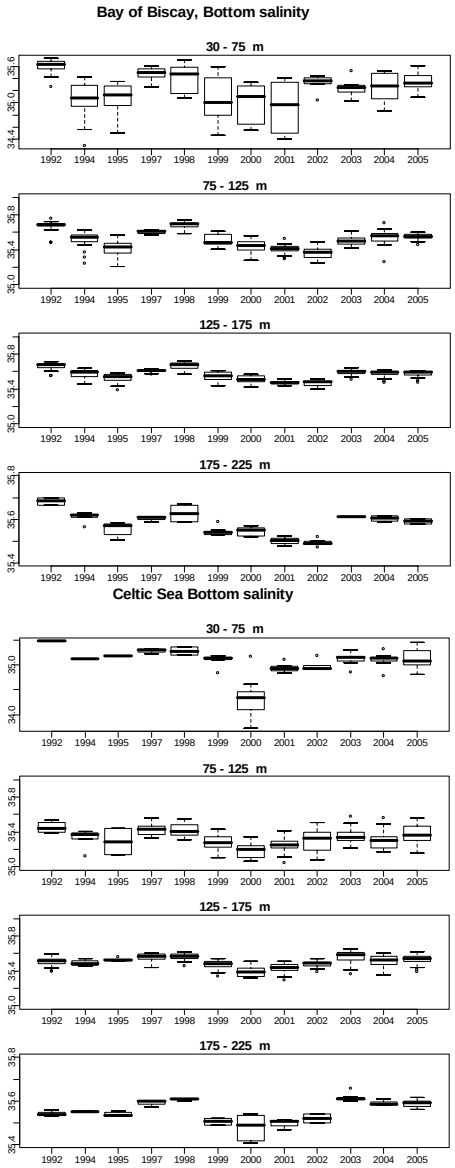


Figure 4.8. Variability of bottom salinity at the location of survey hauls (Western IBTS) in the Bay of Biscay and Celtic Sea in November-December from CTD data.

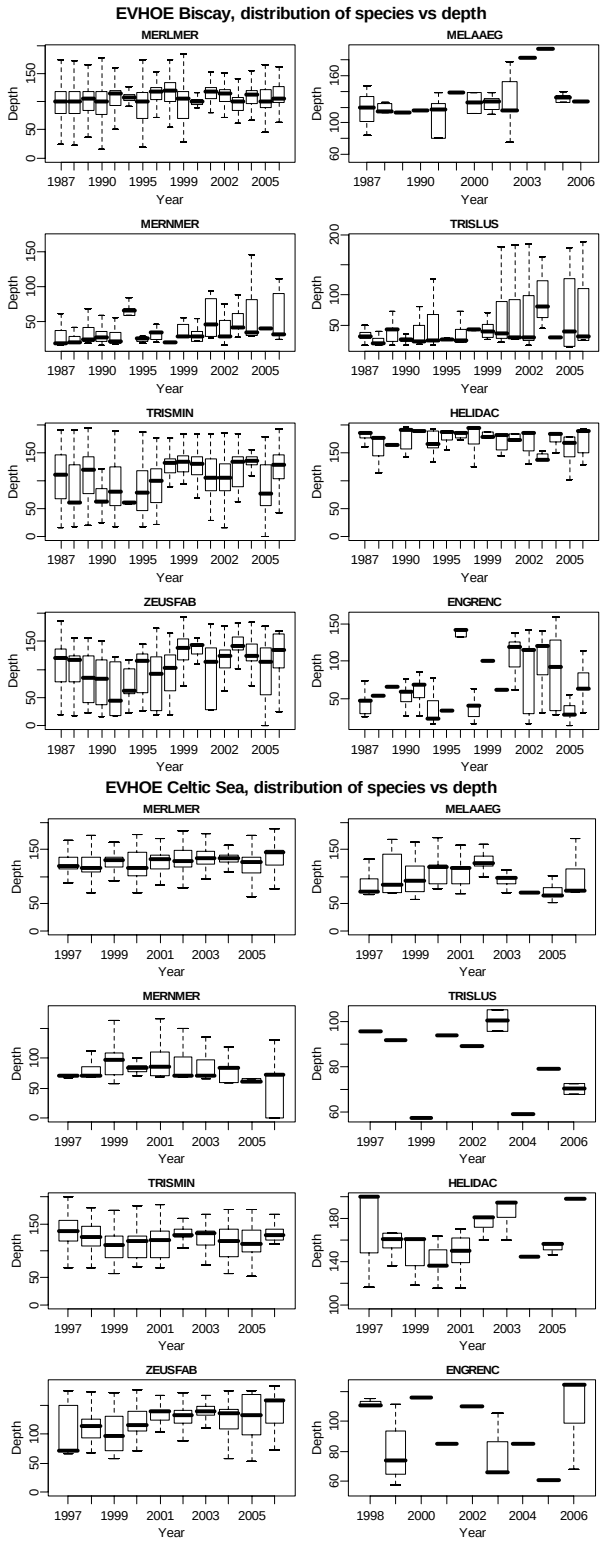


Figure 4.9. Depth distribution by species (from top left: *Merluccius merluccius, Melanogrammus aeglefinus, Merlangius merlangus, Trisopterus luscus, Trisopterus minutus, Helicolenus dactylopterus, Zeus faber* and *Engraulis encrasicolus*) from the western IBTS survey in the Bay of Biscay and Celtic Sea. For this analysis each caught individual was assigned the depth of the given haul.

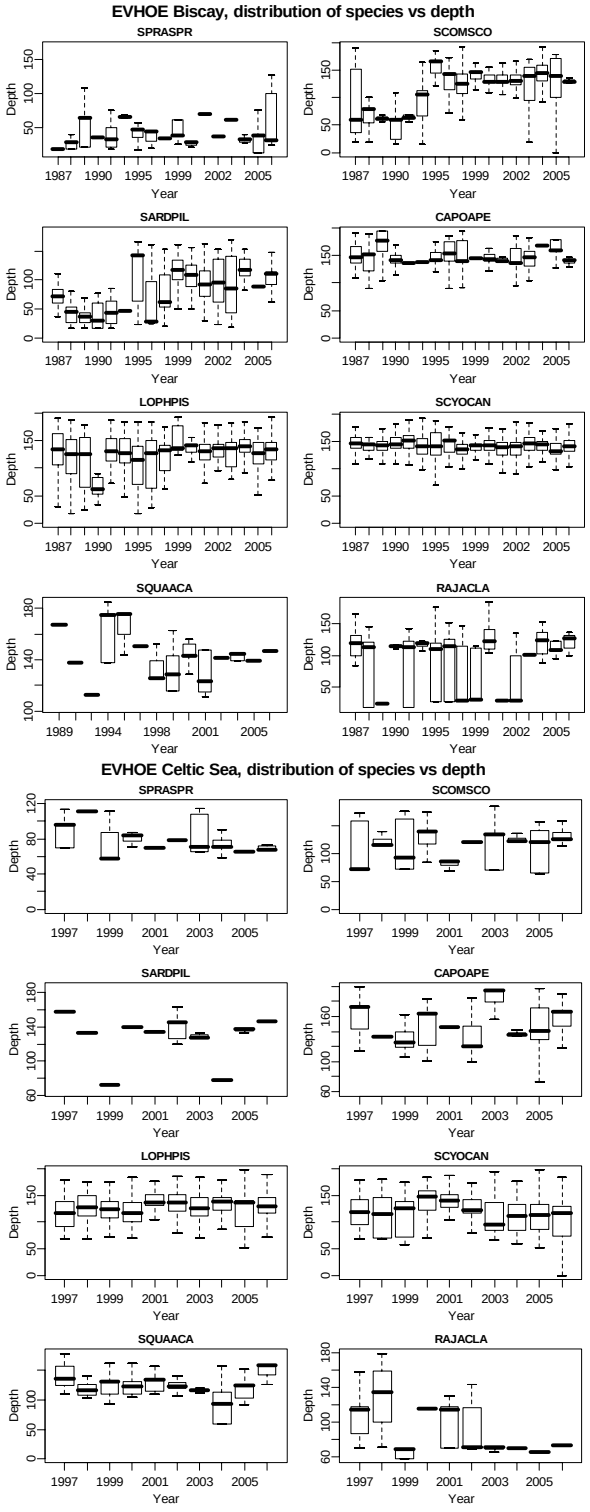


Figure 4.9. continued. Depth distribution by species (from top left: *Sprattus sprattus, Scomber scombrus, Sardina pilchardus, Capros aper, Lophius piscatorius, Scyliorhinus canicula, Squalus acanthias* and *Raja clavata*) from the western IBTS survey in the Bay of Biscay and Celtic Sea. For this analysis each caught individual was assigned the depth of the given haul.

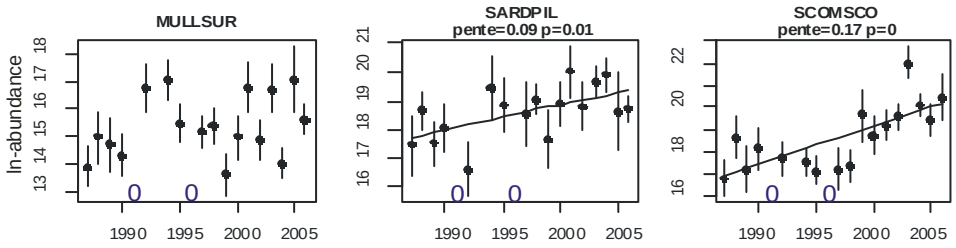


Figure 4.10. Estimated total abundance of red mullet, sardine and mackerel in the Bay of Biscay based on western IBTS data. 0 no survey.

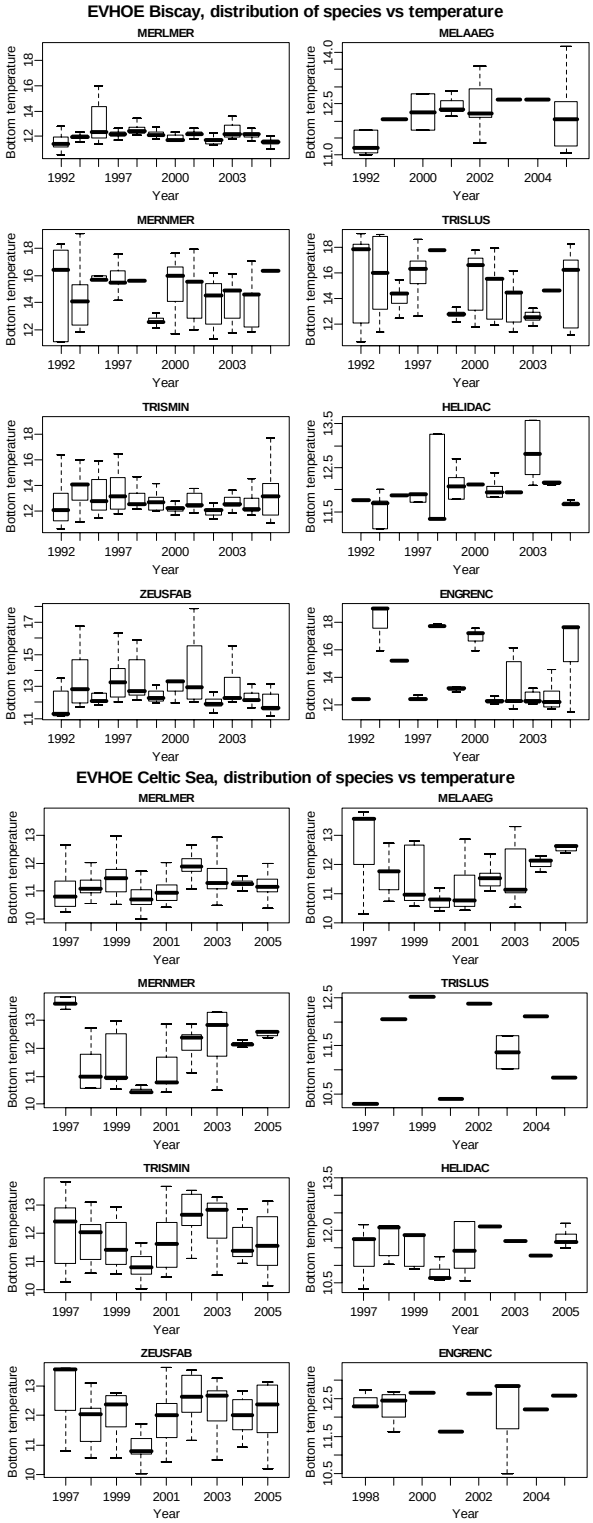


Figure 4.11. Bottom temperature distribution by species (from top left: *Merluccius merluccius, Melanogrammus aeglefinus, Merlangius merlangus, Trisopterus luscus, Trisopterus minutus, Helicolenus dactylopterus, Zeus faber* and *Engraulis encrasicolus*) from the western IBTS survey in the Bay of Biscay and Celtic Sea. For this analysis each caught individual was assigned the temperature of the given haul.

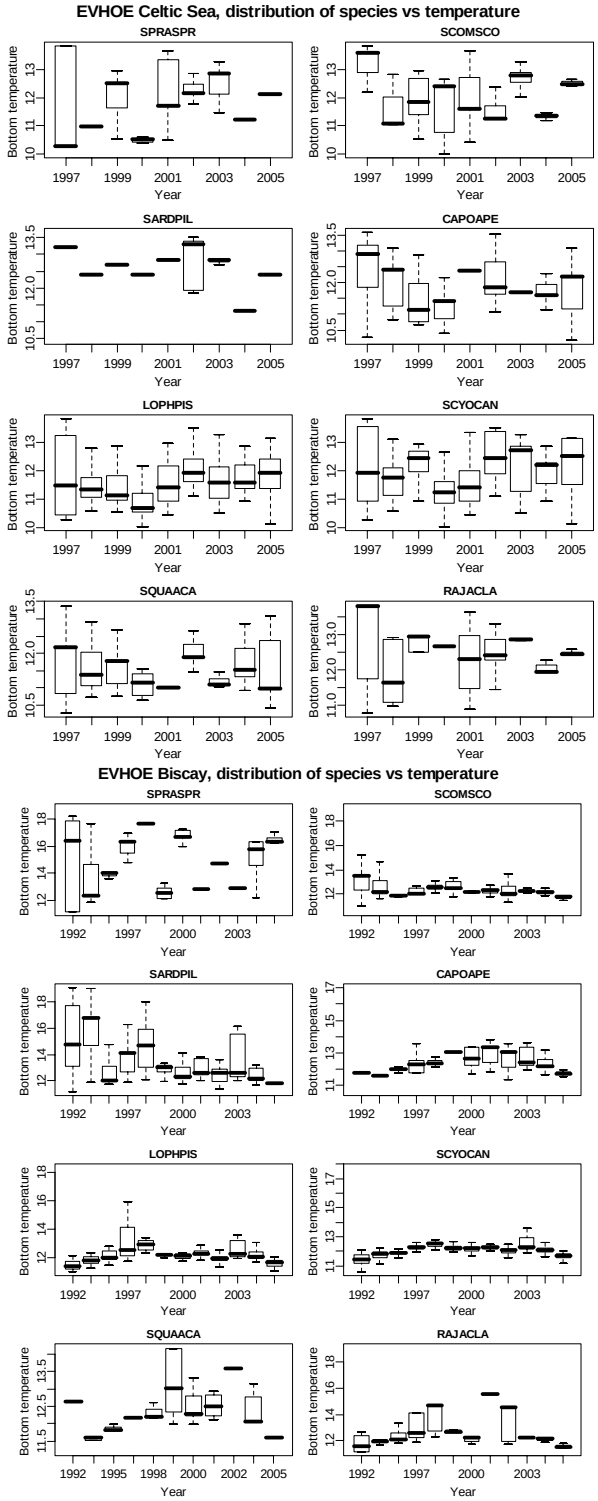


Figure 4.11 continued. Bottom temperature distribution by species (from top left: *Sprattus sprattus, Scomber scombrus, Sardina pilchardus, Capros aper, Lophius piscatorius, Scyliorhinus canicula, Squalus acanthias* and *Raja clavata*) from the western IBTS survey in the Bay of Biscay and Celtic Sea. For this analysis each caught individual was assigned the temperature of the given haul.

4.3.4 Relationships between fish assemblages and the physical environment in the Irish Sea

Cefas have recently begun to participate in the fourth quarter IBTS survey for western areas, fishing at fixed stations in the Irish Sea, Bristol Channel, western English Channel and Celtic Sea (ICES, 2005b, 2006b). The gears used in this survey are a modified rockhopper GOV on hard grounds (see Section 3.1.2 of ICES, 2004b), and GOV trawl with standard ground gear on finer grounds. In addition to data on fish catches, data on the surface and bottom temperature and salinity are collected with a mini-CTD mounted on the trawl, and sediment samples from the stations have been collected with Shippek grab, though it should be noted that currently only 1–2 sediment samples per station have been collected and processed for particle size analysis (PSA). It is intended that further sediment samples will be taken in future surveys.

Preliminary analyses of catch data (numbers of fish and shellfish per hour) from 2005 have been compared with the following physical parameters: water depth, bottom temperature, bottom salinity, % silt, % fine and medium sand, % coarse sand, and % gravel and tidal stress. Catch data for those stations at which all physical parameters were available (Figure 4.12) were fourth root transformed using Primer (version 6.1.6), and the Bray-Cutis similarity of stations calculated and illustrated using MDS (Figure 4.13). It should be noted that these catches come from two gears, which vary in ground gear and bridles/sweeps.

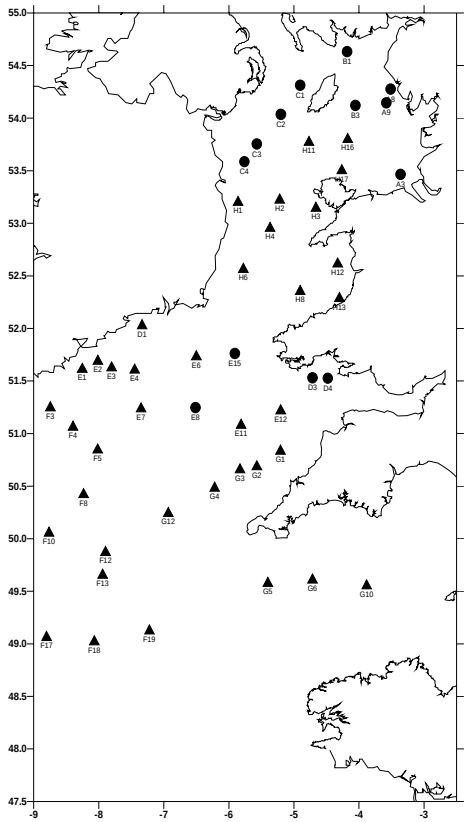


Figure 4.12. Map of survey stations included in preliminary data analysis showing stations fished with rockhopper GOV (triangles) and standard ground gear (circles).

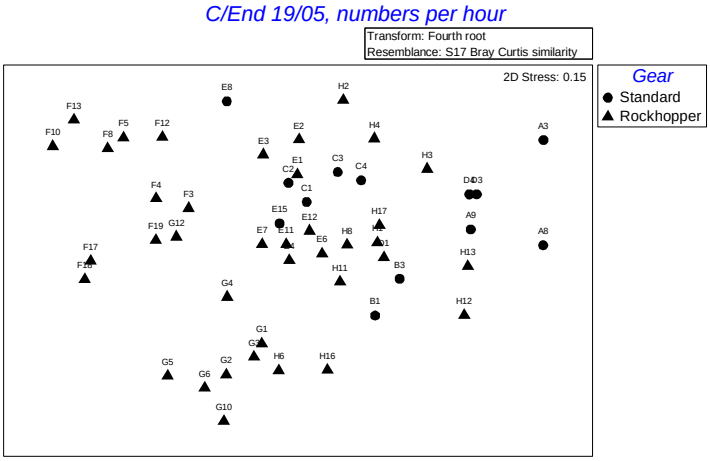


Figure 4.13. MDS plot of the relative abundance of fish and shellfish (fourth root transformed) at stations in the Irish Sea and Celtic Sea, by gear.

BIOENV analysis was used to determine which physical parameters best correlated with fish catches, using the weighted Spearman correlation (ρw). These analyses are intended to illustrate the relative importance of selected physical parameters on the broadscale distribution of fish assemblages, and it is should be noted that further data (both catch data and physical parameters) are required to better examine the relationships between fish communities and particular fish species and the biotic/abiotic environment. Preliminary studies indicated that depth was the single physical parameter that correlated best with the trawl data (ρw=0.46), followed by bottom salinity (ρw=0.39) and temperature (ρw=0.18). The best correlations between multiple variables and the catch data indicated that depth and temperature were the most important two factors (ρw=0.56) and depth, temperature and salinity the most important three factors (ρw=0.61). Though tidal stress and sedimentary data less well correlated as single variables, they were relatively important when considered in conjunction with depth and temperature etc. Figure 4.14 illustrates the MDS of transformed catch data overlain with the various physical parameters, indicating the associations that can exist between fish assemblages and the environment.

Obviously, this preliminary analysis shows how broad physical parameters can affect fish assemblages as a whole, and more detailed studies to examine how such parameters can affect particular species and life-history stages are required. Although water depth and temperature are two of the more important environmental drivers affecting the distribution of fish assemblages, sediment type may have more localised effects on the distribution of fish and fish assemblages on a finer scale (Figure 4.15), and further studies examining the fine scale distribution of fish in relation to seabed sediments and topography are required.

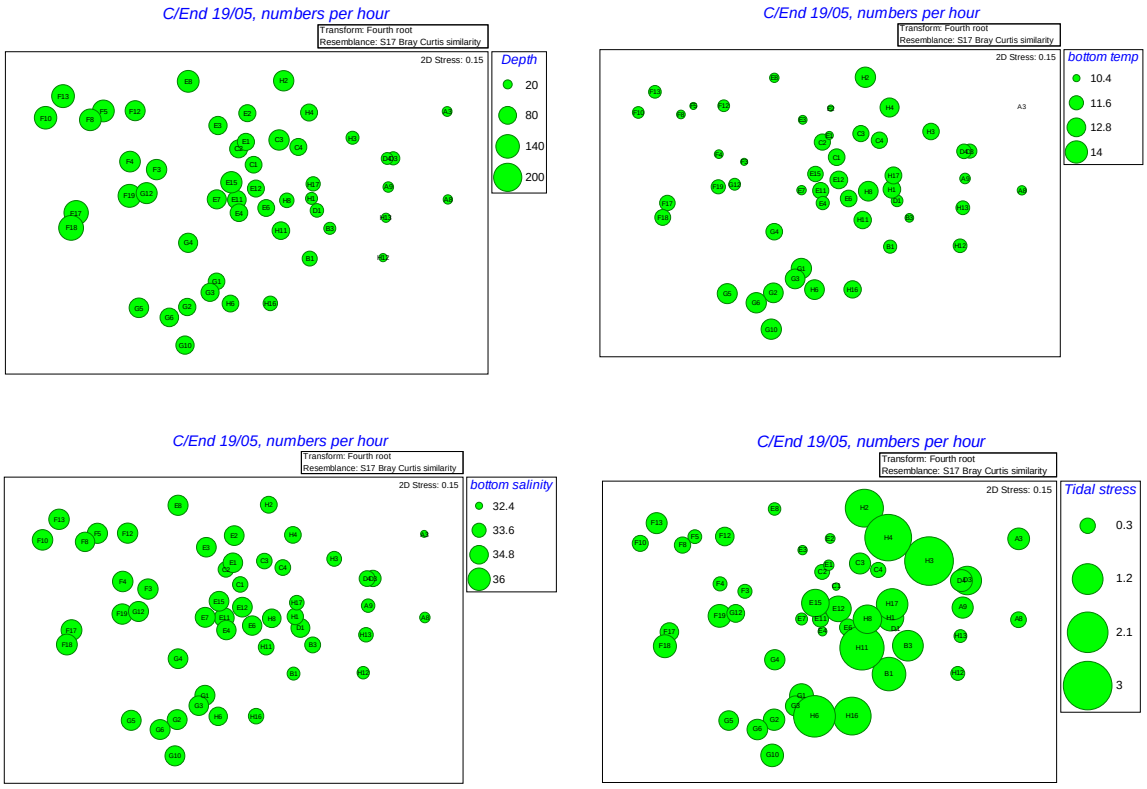


Figure 4.14. MDS plots of the relative abundance of fish and shellfish (fourth root transformed) at stations in the Irish Sea and Celtic Sea overlain with physical parameters.

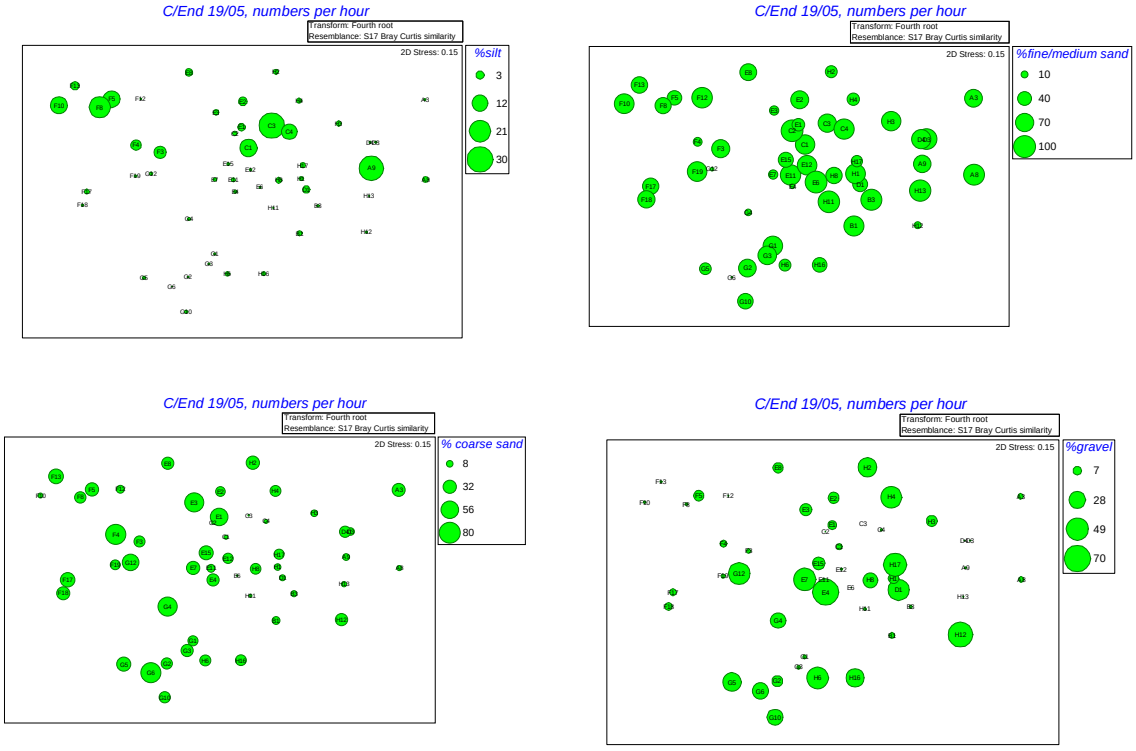


Figure 4.15. MDS plots of the relative abundance of fish and shellfish (fourth root transformed) at stations in the Irish Sea and Celtic Sea overlain with sedimentary parameters.

4.3.5 Physical and biological characteristics of selected gadiform nursery grounds

Preliminary examination of survey data from the UK (England and Wales) westerly IBTS survey were used to provide a preliminary description of the biological characteristics of selected gadiform nursery grounds. Comparatively large numbers of juvenile gadoids (O-groups) are taken in some of the inshore bays in the western Irish Sea, including Dundrum Bay, which can have high catch rates of cod, haddock and whiting (Figures 4.16–4.17). Other abundant fish species at this site include dab, sprat and plaice. Catches at this site (which is in waters 22 m deep) suggest that there are some fine grounds, but also some coarser grounds, as large quantities of the ascidian *Ascidia mentula* are taken. Other benthic species indicative of coarse ground include *Echinus esculentus, Cancer pagurus, Crossaster pappossus, Ophiothrix fragilis* and *Ophiocomina nigra* and a variety of hydroids, whereas the presence of, *Astropecten irregularis* and *Ophiura ophiura* would suggest there are areas of fine ground along the tow. Further south, Dundalk Bay (26 m deep) can also have good catches of juvenile gadoids, especially cod, with a catch rate of >500 ind.h^{-1} in 2004 (Figure 4.16). The overall community here is relatively similar, and once again comprising fauna that typically associates with soft ground (e.g. *Nephrops norvegicus, A. irregularis, Cepola rubescens, Goneplax rhomboides*) as well as fauna that is more characteristic of coarse grounds (e.g. *C. pagurus,* ascidans, *Necora puber, Alcyonium digitatum*).

Just offshore from these bays, very high catch rates of juvenile whiting can be made north-east of Dublin in waters 66 m deep. This site is a muddy ground with good catch rates of *Nephrops,* as well as good catch rates of other decapod crustaceans, notably the shrimps *Dichelopandalus bonnieri, Crangon allmanni* and *Pasiphaea* spp.

Juvenile hake can be very abundant in parts of the northern Celtic Sea (Figures 4.16 and 4.18), with seven stations accounting for >60% of juvenile (<20 cm) hake caught between 2004 and 2006. Two stations south of Ireland have high catch rates of hake (stations E1 and E2 in Figure 4.18, 55–65 m deep), and these sites are relatively muddy, as evidenced by the presence of *Nephrops norvegicus* and *Cepola rubescens*, though coarse ground fauna (e.g. *Marthasterias glacialis* and *Pagurus prideaux*) also occur, suggesting a relatively heterogeneous environment. The associated fauna at stations further offshore that have high catch rates of juvenile hake (F3 and F5, ca. 100 m deep) also have a combination of fine ground fauna (*A. irregularis, Actinauge richardi*, as well as coarse ground fauna (e.g. *P. prideaux* and *Porania pulvillus*). Further south, high catch rates of juvenile hake (F10, F13 and F14, 120–135 m deep) are also typified by a combination of fine ground fauna (e.g. *N. norvegicus*, *G. rhomboides*) and hard ground fauna (*Echinus esculentus, P. pulvillus*). Catches at the offshore stations typically contain a variety of natantid shrimps (e.g. *D. bonnieri, Pontophilus spinosus, C. allmanni* and *Processa* spp.). Additional data on the epifauna of some of these stations are also available from 2m-beam trawl surveys (Table 4.2), which indicate that small crustaceans are abundant in the area, and that there are also several species of sessile invertebrate that may provide habitat (e.g. *Nemertesia ramosa*, *Lytocarpia myriophyllum* and other hydroids, Devonshire cup coral, *Alcyonium digitatum, Actinauge richardi* and the tube-dwelling polychaete *Hyalinoecia tubicola.*

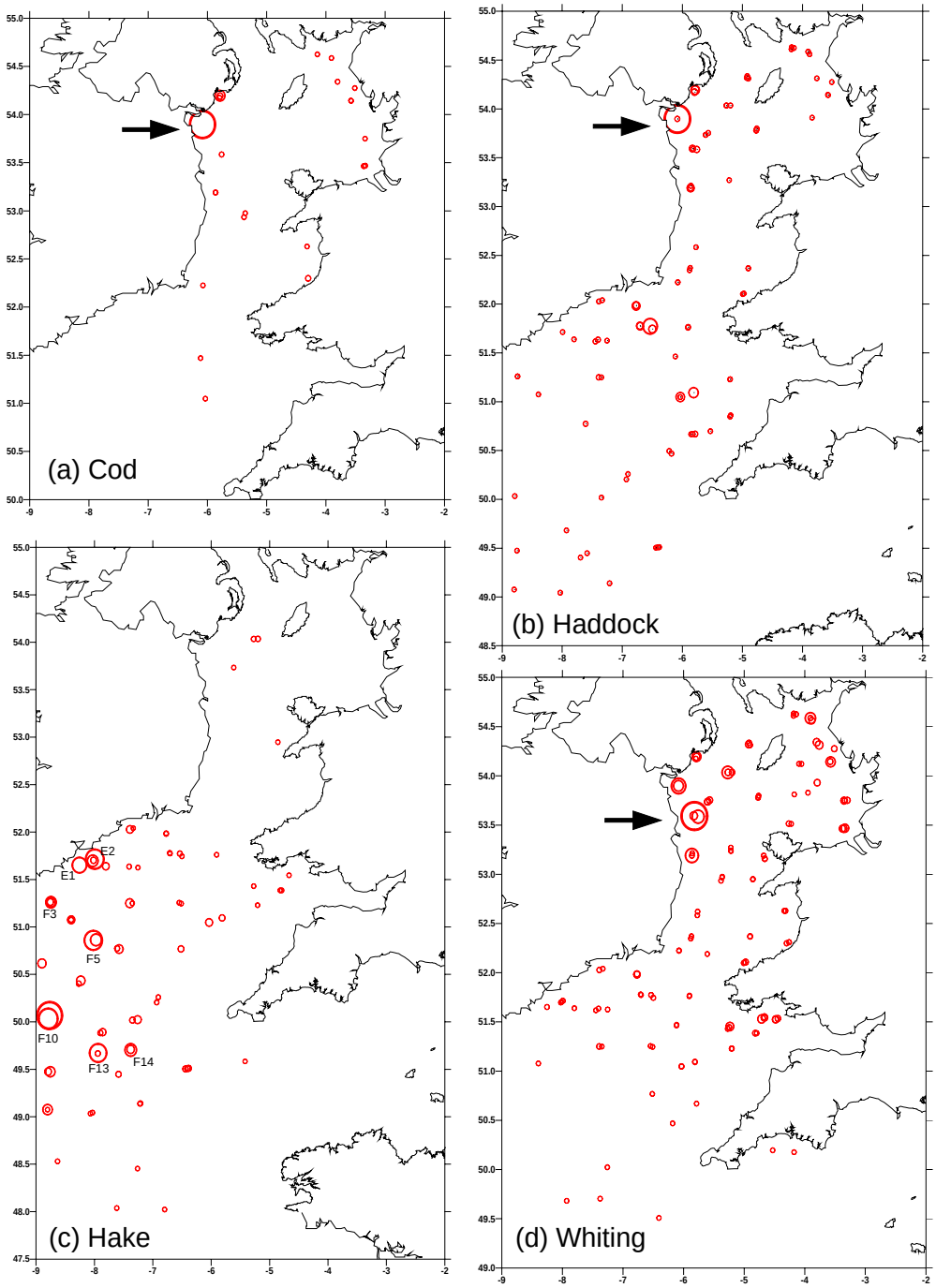


Figure 4.16. Occurrence and relative abundance of juvenile cod (<23 cm) and haddock, hake and whiting (<20 cm) from the UK westerly IBTS survey (2004–2006 combined, catch numbers raised to numbers per hour). Maximum bubble sizes are 504, 14637, 482 and 34834 for cod, haddock, hake and whiting respectively.

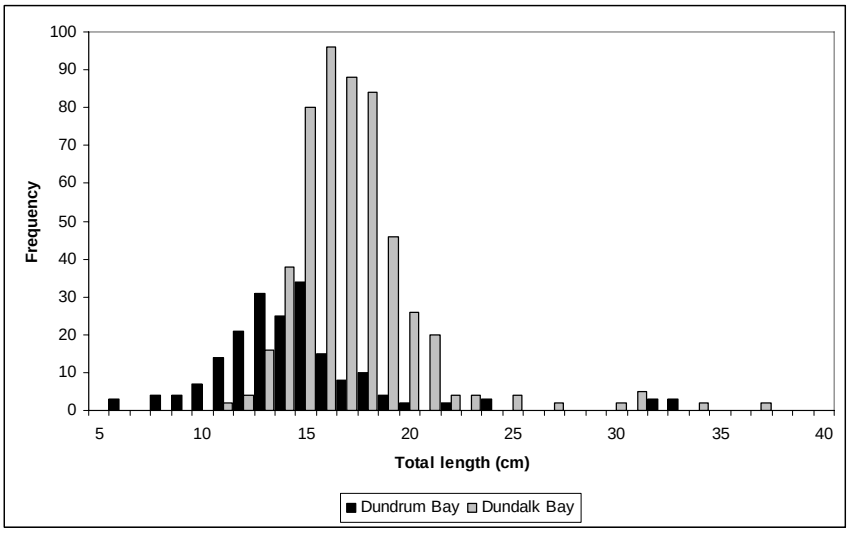


Figure 4.17. Size-distributions of cod in Dundrum and Dundalk Bay in westerly IBTS survey (2004–2006 combined, catch numbers raised to numbers per hour).

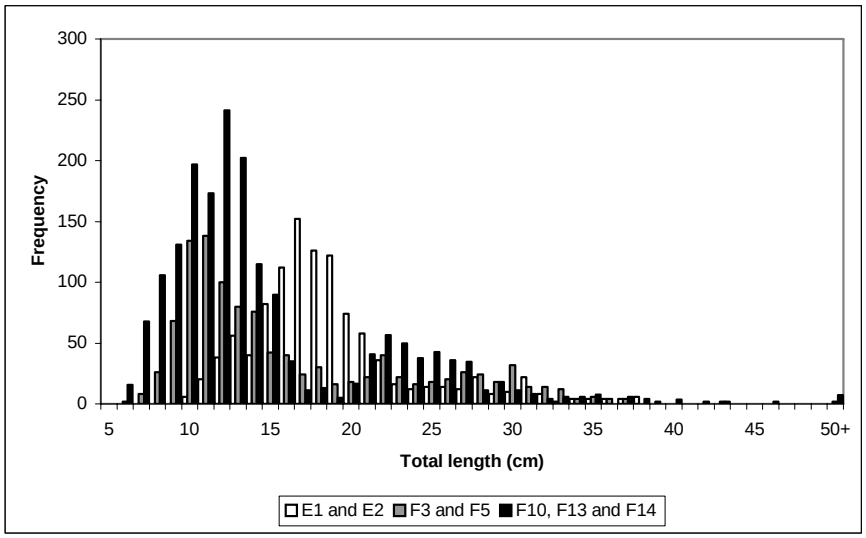


Figure 4.18. Size-distributions of hake at selected stations in the Celtic Sea from the UK westerly IBTS survey (2004–2006 combined, catch numbers raised to numbers per hour).

Table 4.2. Epibenthic fauna (mean catch per tow from 2m-beam trawl sampling) on grounds in the northern Celtic Sea associated with high catch rates of juvenile hake.

STATION		F3		F5		F10	
No. of tows		3		1		2	
Species		biomass (g)	nos	biomass (g)	nos	biomass (g)	nos
Crustacea	*Scalpellum scallpellum*	0.7	2.0	-	-	-	-
	Epimeria cornigera	-	-	-	-	1.0	4.0
	Amphipoda (indet.)	0.3	0.7	-	-	6.3	22.0
	Cirolana cranchii	-	-	-	-	4.0	7.5
	Eurydice sp.	0.3	0.3	-	-	-	-
	Solenocera membranacea	-	-	-	-	0.5	0.5
	Processa canaliculata	24.0	23.3	22.0	36.0	185.5	265.0
	Crangon allmanni	151.7	214.3	58.0	78.0	210.7	303.9
	Philocheras echinulatus	4.0	13.0	8.0	27.0	56.3	253.2
	Pontophilus spinosus	10.7	14.7	16.0	15.0	6.0	5.5
	Alpheus glaber	1.7	2.0	7.0	6.0	43.5	33.5
	Dichelopandalus bonnieri	22.7	15.7	25.0	13.0	29.8	21.2
	Pandalina brevirostris	0.3	2.3	-	-	-	-
	Pandalus propinquus	0.3	0.3	-	-	-	-
	Spirontocaris lilljeborgi	0.7	1.3	-	-	0.5	0.5
	Anapagurus laevis	1.0	2.0	-	-	13.2	65.3
	Pagurus prideaux	193.0	12.7	21.0	1.0	5.0	0.5
	Pagurus variabilis	17.7	1.0	-	-	-	-
	Galathea spp.	3.3	5.3	1.0	1.0	1.5	1.5
	Munida rugosa	29.3	20.0	1.0	1.0	26.0	45.0
	Nephrops norvegicus	7.7	1.0	45.0	31.0	65.0	14.5
	Ebalia tuberosa	2.3	1.7	-	-	-	-
	Ebalia granulosa	1.3	3.0	-	-	-	-
	Ebalia tumefacta	0.7	1.0	-	-	-	-
	Macropodia tenuirostris	1.7	3.7	-	-	1.0	0.5
	Goneplax rhomboides	0.7	1.0	9.0	2.0	9.5	3.0
	Liocarcinus depurator	127.3	30.0	30.0	8.0	33.0	15.5
	Liocarcinus holsatus	0.3	0.3	-	-	2.0	0.5
	Liocarcinus pusillus	0.7	1.7	-	-	-	-
	Macropipus tuberculatus	-	-	-	-	2.5	4.0
	Atelecyclus rotundatus	-	-	-	-	0.5	1.0
Molusca	*Turitella communis*	25.0	35.0	1.0	2.0	309.2	927.7
	Polinices fusca	3.7	0.7	14.0	1.0	17.5	4.0
	Aporrhais pespelecani	1.0	0.3	-	-	-	-
	Calliostoma papillosum	1.0	0.7	-	-	-	-
	Colus gracilis	10.7	0.7	-	-	-	-
	Buccinum undatum	23.3	0.3	-	-	-	-
	Neptunea antiqua	79.3	0.7	-	-	-	-
	Turridae	0.3	0.3	-	-	-	-

STATION		F3		F5		F10	
No. of tows		3		1		2	
Species		biomass (g)	nos	biomass (g)	nos	biomass (g)	nos
	Nucula sulcata	-	-	13.0	12.0	1.5	3.0
	Astarte sulcata	50.3	7.0	-	-	-	-
	Circumphalus casina	14.0	0.3	-	-	-	-
	Palliolum tigerinum	0.7	0.7	-	-	-	-
	Cuspidaria cuspidata	-	-	-	-	0.5	1.0
	Bivalvia indet.	-	-	-	-	1.5	0.5
	Sepiola atlantica	-	-	-	-	0.5	0.5
	Rossia macrosoma	-	-	-	-	6.5	0.5
	Eledone cirrhosa	16.3	0.3	-	-	-	-
Echinodermata	*Antedon bifida*	1.7	4.0	-	-	-	-
	Astropecten irregularis	53.7	20.3	18.0	3.0	6.0	1.0
	Asterias rubens	84.7	18.0	-	-	3.5	0.5
	Luidia sarsi	27.3	2.3	-	-	-	-
	Porania pulvillus	22.7	0.7	-	-	-	-
	Amphiura sp.	-	-	1.0	1.0	5.8	5.8
	Ophiura affinis	-	-	2.0	10.0	-	-
	Ophiura albida	1.3	5.3	-	-	-	-
	Ophiura ophiura	15.7	6.0	-	-	-	-
	Psammechinus miliaris	0.3	0.3	-	-	-	-
	Echinocardium sp.	-	-	-	-	7.5	2.5
	Brissopsis lyrifera	-	-	-	-	29.5	0.5
	Holothuroidea	0.3	0.3	-	-	-	-
Cnidaria	*Nemertesia ramosa*	-	-	-	-	0.5	-
	Lytocarpia myriophyllum	6.0	-	-	-	0.5	-
	Hydroids	10.7	-	1.0	-	0.5	-
	Devonshire cup coral	332.3	86.0	-	-	-	-
	Alcyonium digitatum	6.0	-	-	-	-	-
	Actinauge richardi	67.7	1.0	-	-	64.5	1.0
	Urticina felina	7.3	0.7	-	-	15.5	0.5
	Anemone indet.	6.7	1.0	-	-	-	-
Other	Shell debris	333466.7	-	350.0	-	314.9	-
	Aphrodita aculeata	2.7	0.3	-	-	-	-
	Ditrupa arietina	0.3	1.7	-	-	-	-
	Hyalinoecia tubicola	11.0	15.0	-	-	-	-
	Echiuroidea	1.7	0.3	-	-	-	-
	Alcyonidium diaphanum	0.7	-	-	-	-	-

4.3.6 Research needs to identity functional links

Although the case studies undertaken by WGFE in recent years have highlighted the influence that physical parameters have on the distributions of fish and fish assemblages, and there have been several recent studies on this topic (e.g. Methratta and Link, 2006; Chang *et al.*, 2005; Hinz *et al.*, 2006), such correlations and associations do not identify causal relationships. The

identification of functional links will require a more thorough review of the scientific literature in order to gauge the importance of various habitat features on fish at a population level (e.g. growth models with temperature, survivorship by habitat etc.).

4.4 Areas of core abundance for life history stages

The lack of available ichthyoplankton data has precluded the completion of this TOR, though case studies on assessing the core abundance of other life-history stages are presented.

4.4.1 Availability of ichthyoplankton data

Ichthyoplankton surveys have been undertaken in many parts of the ICES area, including the North Sea (Woehrling and Le Fevre-Lehoerff, 1998; Greve *et al.*, 2005), Irish Sea (Nichols *et al.*, 1993; Fox *et al.*, 1997, Bunn *et al.*, 2004; Bunn and Fox, 2004), eastern English Channel (Korotenko and Sentchev, 1994, 2004; Grioche and Koubbi, 1997; Grioche *et al.*, 1999), Celtic Sea (Horstman and Fives, 1994), Cantabrian Sea (Sola and Franco, 1985; Rodriguez and Rubin, 1991) and Portuguese shelf (Afonso, 1995). Collation of available data is required before the distribution of the egg and larval stages can be mapped and the core areas identified. Although there should be appropriate species-specific data for some marine species (e.g. mackerel), the eggs and larvae of some other taxa (e.g. some gadoids) can be difficult to identify to species level, and are therefore sometimes recorded by genus or family.

Several ICES WGs are involved in ichthyoplankton surveys, including the Planning Group on North Sea Cod and Plaice Egg Surveys in the North Sea (PGEGGS), Working Group on Acoustic and Egg Surveys for Sardine and Anchovy in ICES Areas VIII and IX (WGACEGG), and the Working Group on Mackerel and Horse Mackerel Egg Surveys (WGMEGS).

There are also dedicated programmes for monitoring herring larvae. The ICES programme of international herring larval surveys in the North Sea and adjacent areas has been in operation since 1967. The main purpose of this programme is to provide quantitative estimates of herring larval abundance, which are used as a relative index of changes of the herring spawning stock biomass in the assessment. The larval surveys are carried out in specific time periods and areas, following the autumn and winter spawning activity of herring from north to south: the Orkney/Shetland area, the Buchan region, the Central North Sea and the Southern North Sea. All other locations, e.g. IVa North and IVa South and VIIb, which were part of the surveys up to the 90s, have not been sampled since 1994.

The aim is to sample the major herring spawning grounds in the appropriate areas in an approximate 10*10 nm rectangle grid. Sampling takes place with a GULF III or GULF IV sampler, which is fitted with a 280 or 300 µ mesh size net. The International Herring Larvae database contains information about the surveys conducted since 1972 and is held at the Leibniz Institute of Marine Sciences in Kiel, Germany. More information can be found in the Manual for the International Herring Larvae Surveys south of 62° North – Version 24.01.06, which is an appendix in ICES (2006d)

Also during IBTS 1st quarter, herring larvae are sampled during the night by small, fine-meshed nets. From 1977 to 1991 the gear was a small mid-water trawl (IKMT), but due to poor catchability of this gear, the standard gear was changed to a 2-metre ring net (MIK), used since the 1991 sampling. The total abundance of herring larvae in the survey area is used as an estimate of 0-ringer abundance of the stock.

4.4.2 Availability of data on post-larval fishes

Our current knowledge of the settlement habitats for larval fish and the habitats occupied by post-larval stages is currently limited for most fish species in the ICES area (but see Beyst *et al.*, 1999; Wennhage and Pihl, 2001). The role of microhabitats for recently settled fish are

better studied elsewhere in the world (e.g. Auster *et al.*, 1997; Tunesi *et al.*, 1997; Tolimieri, 1998; Lindholm *et al.*, 1999; Steves *et al.*, 2000; Steves and Cowen, 2000; Jenkins and Hamer, 2001; Johnson et al, 2001; Manderson *et al.*, 2002; Rooker *et al.*, 2004; Able *et al.*, 2006; Sullivan *et al.*, 2006), and these studies indicate that several habitat features, such as complex habitats (e.g. the crests of sand waves, shell banks, amphipod tubes, emergent biogenic fauna), presence of con-specifics, lack of predators, and abundance of suitable prey are important characteristics of juvenile habitat, and that settlement habitat can be more specific than juvenile habitat.

4.4.3 Juvenile fish surveys (e.g. IBTS)

The distribution and abundance of juveniles of selected species are presented in Sections 4.2.2 and 4.2.3.

4.4.4 Adult fish surveys

Further analyses of DATRAS data are required to examine the distributions of mature fish, though the timing and locations of the surveys and the sampling gear and protocols will limit the utility of the IBTS surveys for identifying some of the important adult habitats (e.g. feeding grounds, spawning grounds etc.).

4.5 Broadscale mapping of fish distributions and environmental parameters

Although some basic physical data are available for broadscale analyses (e.g. water temperature and salinity), and other oceanographic data are available from oceanographic models (e.g. tidal stress), the lack of integrated electronic data for seafloor sediments, seabed topography and benthic communities is an issue that will limit the interpretation of the broadscale mapping of demersal species.

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5 Development and testing ecosystem quality objectives (Eco QOs) for marine fish communities

TOR c) EcoQOs: continue analyses of the sensitivity, response and specificity of fish community indicators using simulation approaches and supporting empirical analyses

5.1 Objectives

Indicators are needed to evaluate how well a fishery is managed, in relation to specified objectives (Hilborn and Walters, 1992). In recent years fisheries research has focused primarily on developing indicators; much less effort has gone into developing approaches for actual fisheries management in an ecosystem context based on indicators. Clearly if appropriate management performance indicators are monitored, they will reveal if something is wrong, but do not necessarily tell what is wrong, nor what management action should be implemented to mitigate effects. In the context of single stock assessments, the dual role of indicators as triggers for management measures ('control' function) and elements of management performance reporting ('audit' function) has emerged for various practical and historical reasons (Rice and Rivard, 2006), where the second role corresponds to the function mentioned by Hilborn and Walters cited above. The dual role of stock indicators seems to have pervaded to ecosystem indicators.

Whereas indicators related to specified objectives have to be agreed on for the audit function, metrics could be used in a more comprehensive way for the control or other functions. The size-based indicator issue is addressed in Section 6 (ToR f and g) below. Here ToR c) is addressed, understanding indicators as metrics in a broader sense.

First, several candidate metrics for changes in size structure in exploited communities are examined based on empirical analyses. The questions addressed are:

- Which size-based community metrics are sensitive and responsive to changes in the size-structure of the community, owing to fishing impacts or other environmental changes?
- How do size-based community metrics reflect documented changes in populations?
- Which size-based community metrics are to be used for what purposes?

Second, ways to link diversity metrics with size are explored. Slope of diversity size-spectrum has been examined as a potential metric with MSVPA and empirically, a further application and critical analysis of other potential size based diversity metrics is presented.

5.2 Sensitivity and response of fish community metrics based on empirical analyses: French bottom surveys

5.2.1 Materials and methods

5.2.1.1 French surveys

Six continental shelf communities supporting mixed fisheries were monitored by bottom-trawl surveys and three estuarine communities identified as nursery areas for commercially important stocks exploited elsewhere in mixed fisheries were sampled by beam-trawl surveys. In all surveys, all fish are identified and counted, and most or all are measured. In all surveys, the sampling design is stratified according to depth and some other criterion (*e.g.* North/South in the Bay of Biscay, bottom substrate in the Vilaine estuary). Survey trawls do not sample all species equally well. Population metrics were estimated only for the fish species for which a reasonable precision could be achieved (Table 5.2.1). The criteria for selecting a species were

a sufficient occurrence (proportion of hauls with the species present being larger than 5%), a sufficient density and/or commercial interest, and availability of length data (*e.g.* in the MEDITS surveys, only a subset of species are measured).

Table 5.2.1. List of the survey data used in the analysis.

COMMUNITY	SURVEY	TIME SERIES AVAILABLE	SEASON	NUMBER OF HAULS / YEAR	TOTAL AREA COVERED (KM²)	DEPTH RANGE (M)	NUMBER OF FISH SPECIES CAUGHT
Seine Estuary	Seine	1995–2002	Autumn	45	550	0–20	56
Somme Estuary	Somme	1998–2004	Autumn	48–54	718	0–20	41
Vilaine Estuary	Vilaine	2000–2004	Autumn	19–46	330	0–20	37
East Corsica	MEDITS(1)	1995–2004	Spring	13–25	4 562	10–800	163
Gulf of Lions	MEDITS(1)	1995–2004	Spring	64–76	13 860	10–800	179
Southern North Sea	IBTS(2)	1990–2005	Winter	143–210	252 813	15–100	106
Eastern Channel	CGFS(3)	1997–2004	Autumn	83–109	30 672	7–82	76
Celtic Sea	Western IBTS(4)	1997–2004	Autumn	53–82	150 000	20–400	103
Bay of Biscay	Western IBTS(4)	1987–2005	Autumn	56–113	72 500	20–600	194

(1) International bottom trawl surveys in the Mediterranean (Anonymous, 1998).

(2) International Bottom Trawl Survey (ICES, 1996).

(3) Channel Ground Fish Survey (Carpentier *et al.*, 1989).

5.2.1.2 Metrics

The following size-based community metrics were used (Table 5.2.2): average weight and length in the community, the proportion of large fish above fixed thresholds (15, 20, 25, 30 cm), and population length percentiles (0.05, 0.25, 0.75 and 0.95) averaged across populations. In addition, a diversity index Δ_1, interpreted as the probability that two individuals randomly picked up in the community belong to different species (Hurlbert, 1971), was used as a crude measure of dominance in the community.

Time series of these metrics were plotted and linear trends tested using standard regression with $\alpha = 0.05$.

Table 5.2.2. Definition of the community size-based metrics used in the analysis.

POPULATION METRICS	DEFINITION	REQUIRED DATA	ESTIMATOR
N_i	Abundance index for species i	Catch haul k stratum j $y_{k,j}$ Swept area $a_{k,j}$ Stratum area A_j	$N_i = \sum_j N_{i,j} = \sum_j A_j \sum_{k=1}^{n_j} y_{k,j} \Big/ \sum_{k=1}^{n_j} a_{k,j}$ $Var(N_i) = \sum_j \frac{A_j^2}{n_j - 1} \sum_{k=1}^{n_j} \left(\frac{y_{k,j}}{a_{k,j}} - \frac{\sum_{k=1}^{n_j} y_{k,j}}{\sum_{k=1}^{n_j} a_{k,j}} \right)^2$
$L_{q,i}$ q = 0.05, 0.25, 0.75, 0.95	Percentiles of the population length distribution	Catch per length class $y_{l,i}$	$L_{q,i} = l_{q,i} \left| \frac{\sum_{l=1}^{lq} y_{l,i}}{y_i} = q \right.$ $Var[L_{q,i}] = \frac{q(1-q)}{y_i \left(y_{l_{q,i}} / y_i \right)^2}$

COMMUNITY METRICS	REQUIRED INPUT	ESTIMATOR
Diversity Δ_1	N_i	$\Delta_1 = \frac{N}{N-1}\left[1 - \sum_{i=1}^{n} \left(\frac{N_i}{N}\right)^2\right]$ $Var[\Delta_1] \approx \sum_i^n Var[N_i] \left(\frac{2N_i}{N^2} - \sum_i^n \frac{2N_i^2}{N^3} \right)$ Confidence interval by parametric bootstrap
Community average weight pmoy $\bar{b}$	N, B (total abundance and biomass in community)	$\bar{b} = B/N$ $Var[\bar{b}] = Var[B]/N^2 + B^2 Var[N]/N^4$
Community average length lbcomm $\bar{l}$	Catch per length class y_l	$L_{bar_i} = \frac{\sum_{l=1}^{L} y_l \, l}{y}$ avec $y = \sum_{l=1}^{L} y_l$ $Var[L_{bar}] = \left(\frac{\sum_{l=1}^{L} y_l \, l^2}{y} - L_{bar}^2 \right) \Big/ y$
Proportion of large individuals PropG15, PropG20, PropG25, PropG30 p_{large}	$y_l(t)$ catch per length class l $y(t)$ total catch (measured species) Large size threshold *lbig*	$p_{\text{large}}(t) = \sum_{l>lbig} y_l(t) \Big/ y(t)$ $V[p_{\text{large}}] = \frac{p_{\text{large}}(1 - p_{\text{large}})}{y(t)}$
Average population length percentiles $l_{0.05}$, $l_{0.25}$, $l_{0.75}$, $l_{0.95}$	Population length percentiles $L_{q,i}$ S number of consistently measured species	$l_q = \sum_{i=1}^{S} L_{q,i} \Big/ S$

5.2.2 Results

Most community metrics fluctuated without any obvious trend (Figure 5.2.1). The proportion of large fish fluctuated more widely for small compared to larger threshold, probably because of a larger influence of recruitment events. Average length and proportion of large fish were precisely estimated, but average weight was not because fish are not individually weighed in these surveys, and weighing can be difficult onboard. Significant trends were recorded for the proportion of fish larger than 30 cm in the English Channel (increasing) and average weight in the Somme estuary (decreasing). These results seem somehow surprising as a majority of population average length and average weight were decreasing (and/or a majority of the significant trends in these population metrics were decreasing) in the English Channel, Southern North Sea, Bay of Biscay, Celtic Sea, Corsica, Seine, and Vilaine (Figure 5.2.2). In the nurseries and the Gulf of Lions the proportions of fish larger than 15 cm were below 20% and the proportions of fish larger than 30 cm were close to 0. For the shelf communities (except gulf of Lions) 25 cm seems an appropriate threshold for comparisons across communities as it approximately amounts to the 0.95 percentile of the community (Figure 5.2.1). Larger threshold will result in lower proportions with variance poorly estimated and a risk the proportion reaches zero along time.

The picture was different with average population length percentiles (Figure 5.2.3). Significant decreasing trends were found in at least two of $l_{0.95}$, $l_{0.75}$ and $l_{0.25}$ in the Bay of Biscay, Celtic Sea, English Channel and Southern North Sea. In addition, $l_{0.25}$ was significantly decreasing in the Seine Bay, and $l_{0.05}$ and $l_{0.25}$ were significantly increasing in the Somme Bay. That average length in the community was below the average 25th percentile or even below the average 5th percentile in all the shelf surveys suggests that these communities are strongly dominated by small-sized populations. Indeed, the diversity index Δ_1 suggests these communities have a high dominance (Table 5.2.3). The probability that two randomly chosen individuals belong to different species can be as low as 0.57 in the Bay of Biscay. In those highly dominated communities, metrics calculated for all individuals of the community regardless of species are likely to behave like metrics for the dominant population(s). This is the reason why the average percentiles across populations (unweighted by population abundance!) will provide a more balanced picture of the events in the sampled community.

Table 5.2.3. Summary of significant trends in size-based community metrics, average diversity index Δ_1 and number of populations included in average percentiles, in nine French surveys. In stationary situations, for 90 tests performed with $\alpha = 0.05$, 4 to 5 results would be expected to be significant just by chance. All the metrics are expected to decrease under the impact of fishing, except l0.05 which should be neutral to fishing.

	LBCOMM	PMOY	PG15	PG20	PG25	PG30	L0.05	L0.25	L0.75	L0.95	Δ_1	NO SPECIES
English Channel						↗		↘	↘		0.71	20
S-North Sea									↘	↘	0.71	16
Bay of Biscay								↘	↘	↘	0.57	34
Celtic Sea									↘	↘	0.78	48
Gulf of Lions									↗	↘	0.77	26
Corsica			↘								0.84	24
Somme		↗					↗	↗			0.74	17
Seine								↘			0.76	14
Vilaine											0.73	12

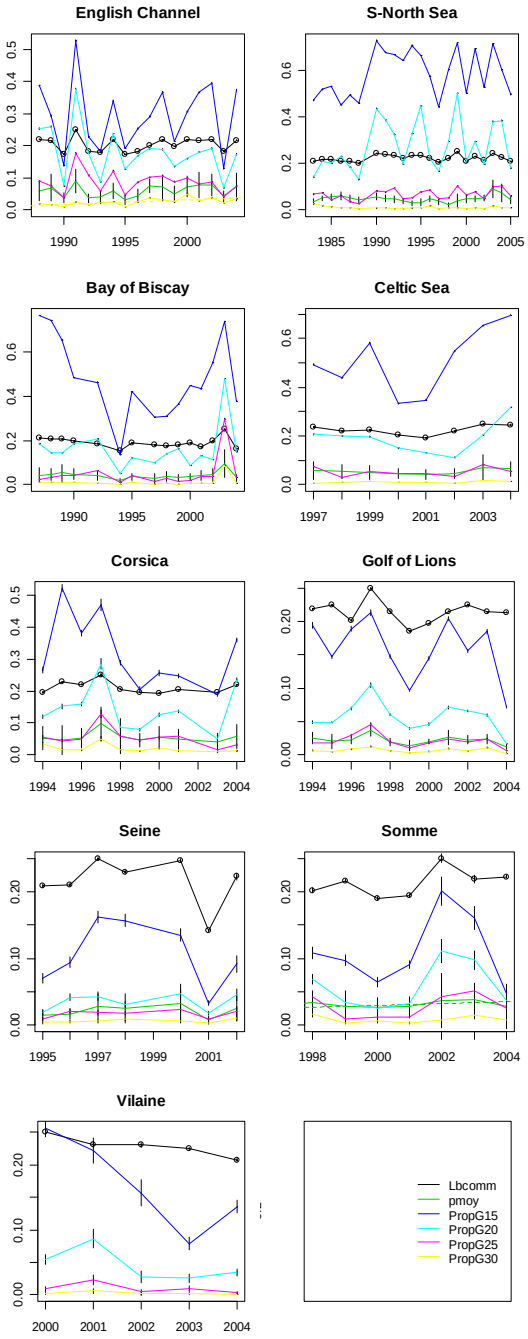


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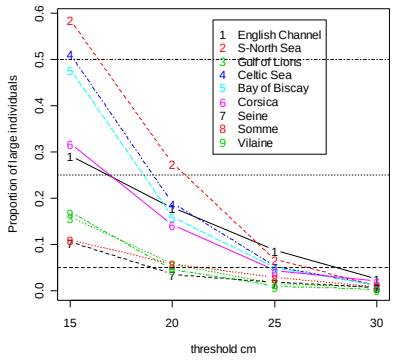


Figure 5.2.1. Time-series of community size-based metrics for nine French surveys. Average length was scaled to 0.25 to be plotted on the same scale as the proportions and average weight (in g). Continuous lines: observed fluctuations in metrics. Black bars: 95% confidence intervals. Dashed lines: fitted linear trend when P≤0.05. Bottom right panel: comparison of average proportion of large fish among communities with different thresholds.

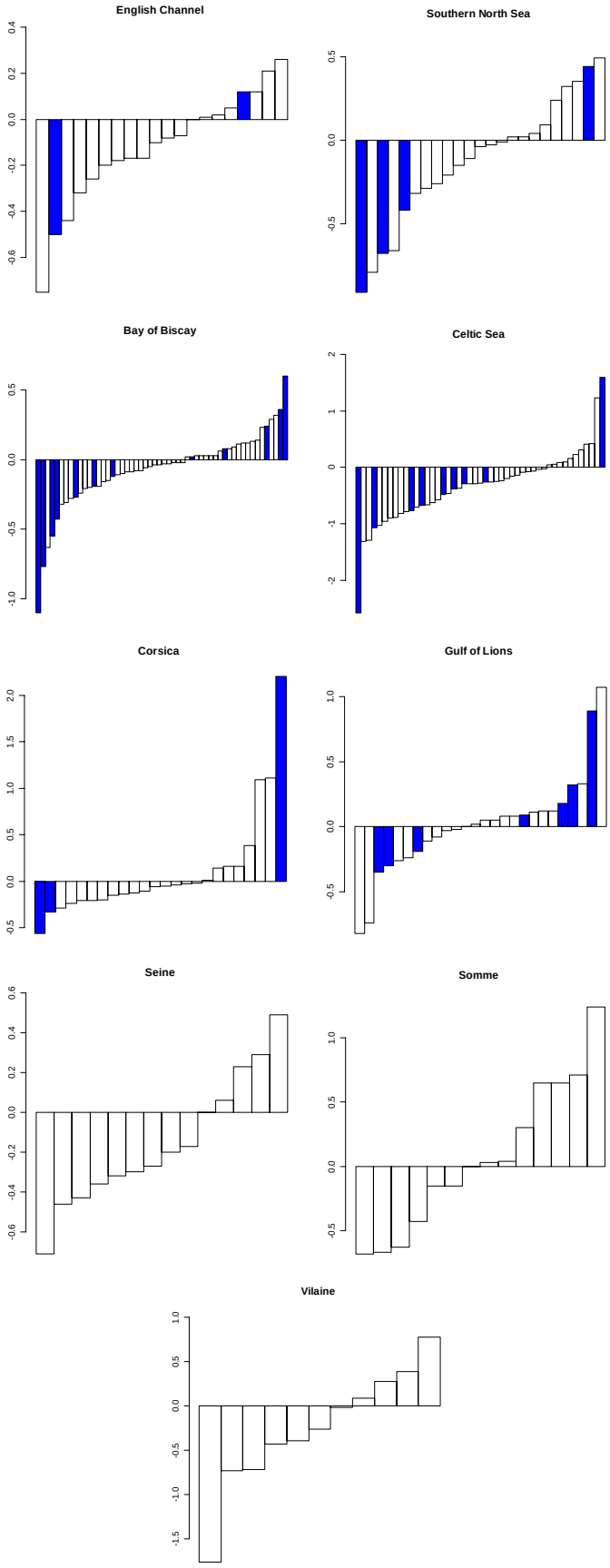


Figure 5.2.2. Slopes (cm.y^{-1}) of time trends in population average lengths in nine French surveys, ranked by increasing order. Coloured bars are for slopes significant at $\alpha = 0.05$.

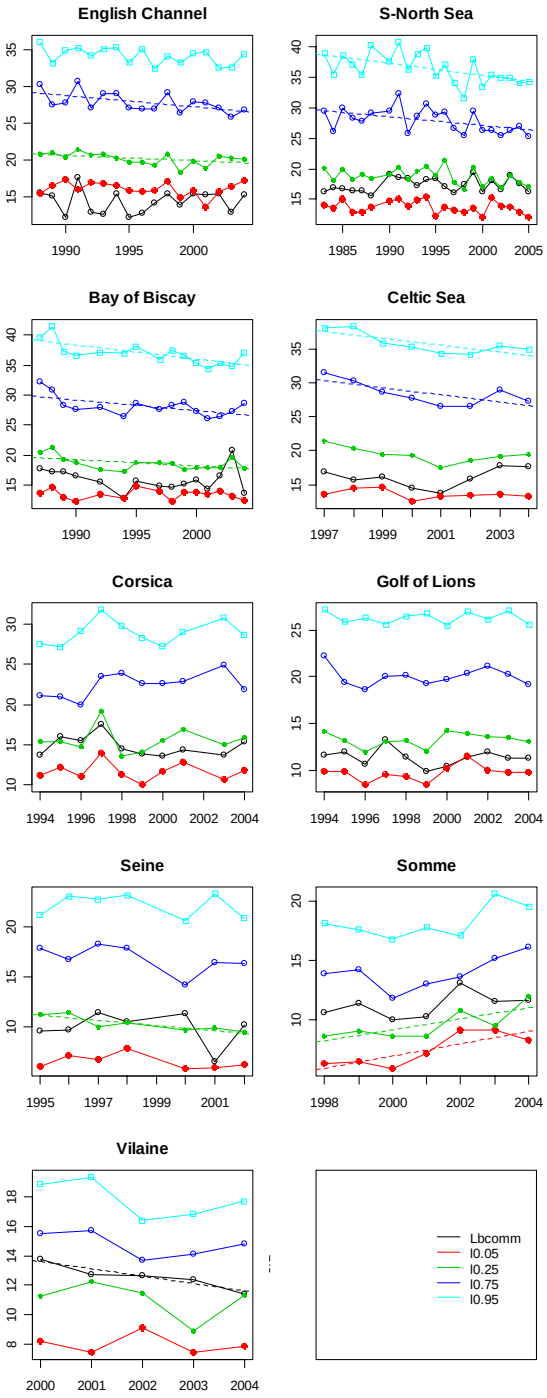


Figure 5.2.3. Time-series of length-percentile population metrics averaged across populations for nine French surveys (number of populations per community: see Table 4). Lengths in cm. Continuous lines: observed fluctuations in metrics. Dashed lines: fitted linear trend when P≤0.05.

5.2.3 Conclusion

- Community metrics calculated regardless of species are driven by the most abundant population(s) in strongly dominated communities
- Therefore average $l_{0.95}$ might be a more balanced large-fish metric than the proportion of large fish in those communities
- Depending on the community dominance structure, either average weight or the proportion of large fish (even communities) or average population metrics (uneven communities) can be expected to be sensitive and responsive to changes in the size-structure of the community. Community average length does not seem to be a sensitive metric. For the shelf communities off France, 25 cm seems an appropriate threshold for large fish.
- A suite of community metrics could be used to monitor the changes in the size-structure of a community in addition to population metrics, including some of the ones examined here as well as in the 2006 report of WGFE. As an indicator to report about ecosystem health, WGFE suggests that the proportion of large fish might not be very sensitive; a possible alternative is average $l_{0.95}$ at least in strongly dominated communities.

5.3 Temporal and spatial trends of diversity size spectra – Case-study in the North-western North Sea

When contrasting species diversity against fishing effort (spatially or by means of time-series), contradictory results may result from the fact that the diversity-disturbance relationship is mediated by trophic cascades. Demersal fish assemblages, spanning a wide size-range, include more than one trophic level (Cushing, 1975), thus diversity of the whole demersal community could be the outcome of different simultaneous responses of the different trophic levels present. This is an argument for exploring diversity patterns separately for different fish size-classes, reflecting different trophic levels.

Huston (1994) holds that changes in diversity in response to disturbance can be explained by means of trophic dynamics such as predation and competition. One of the basic assumptions is that, for a given trophic level, coexistence among competitor species (i.e. species exploiting the same food resources) is a non-equilibrium state that is maintained as long as sources of disturbance, such as predation, control the expansion of dominant species at the expense of the other less competitive ones (Hutchinson, 1959; Huston, 1994). According to this hypothesis, diversity within each trophic level is (also) a consequence of the degree of predation by higher trophic levels. Other mechanism (e.g. fisheries on forage species) may also affect diversity but were not studied here.

Based on this, it is expected that an indirect consequence of the removal of top predators caused by fisheries should be the expanding of the dominant prey populations, resulting in an overall decline in diversity of prey size classes within the assemblageOn the other hand, fishing disturbance acts directly on the predator component of the assemblage so one might expect different mechanisms to regulate diversity changes for these larger individuals (e.g. moderate fishing may reduce competition thus increasing diversity, intense fishing may be causing the decline of some populations and thus of diversity).

Demersal fish diversity size spectra were compared among areas undergoing different fishing pressures in the North-western North Sea (SAGFS survey data, from 1926 to 1996). Fishing pressure was estimated based on commercial trawling effort in 1990–1995 and divided into 3 nominal categories: LOW, MEDIUM and HIGH (for details on the data sets see Greenstreet and Rogers, 2006).

5.3.1 Questions asked

Size-classes most targeted by fisheries (based on minimum landing sizes) were considered the "predator" classes, the average prey size they feed upon were considered the "prey" classes. To test

a) whether different size classes respond differently to fishing disturbance in terms of diversity and

b) whether changes in diversity of small fish can be explained through a predatory release mechanism

Spatial analyses were conducted to compare spectra shapes (fitted curve comparison tests) and "prey" size class diversity and biomass (ANOVA one-way comparisons, with fishing effort as a 'factor') between heavily fished and less fished areas (pooled samples from 1982–1996).

To disentangle habitat-type effects from fishing effects in the spatial comparisons, the evolution of diversity size-spectra in time (1926–1996) for LOW, MEDIUM and HIGH effort areas was also explored (fishing is assumed to have increased, habitat characteristics such as depth and sediment type are assumed to have remained constant). Spectra shapes are compared in time and trends in diversity are shown separately for "prey" and "predator" size ranges.

5.3.2 Methods

Size spectra

The size scale used for the x-axis of the spectra is $\log_2$ weight. Weight values for each species were obtained converting recorded lengths through weight at length relationships (see Jennings *et al.*, 2001 and references therein). Abundances and biomasses per size-class were normalized by dividing the biomass or abundance by the respective weight-class width (i.e. the weight interval in grams) and then logged ($Log_{10}((N/10h)/\Delta$ g) (Platt and Denman, 1977, 1978). Separate spectra were obtained for each of the three fishing effort areas.

Size class selection for hypothesis testing

Minimum landing sizes for commercial species were used to estimate the main targeted fish size range. Predator-prey size-ratio estimates available in published literature (Frœse and Pauly, 1998; Jennings *et al.*, 2001) were used to estimate the prey size range for the fishing targeted fish.

Expected fishing targets are weight classes 8–9 and above (fish >= 516 g) and preferred prey size range for these predator classes is expected within classes 4–5 (fish between 16–64 g).

Diversity

'Richness' is the diversity component measured by species count; whereas 'evenness' is the diversity component measured by the distribution of individuals among species; 'diversity' is understood here a generic term applied to either of the indices. Diversity indices estimated were:

Richness: N_0 = number of species (species richness or Hill's N_0); d = $(N_0-1)/LnN$ (Margalef's richness); Evenness: J' = H'/LnN_0 (Pielou's evenness); N_1 = exp H' (Hill's N_1); N_2 = 1/S (Hill's N_2), where "H'" is Shannon-Wiener's index and "S" is Simpson's index.

Due to their higher stability when reduced sample sizes are available, only results for J' and N1 are shown in the time-series analyses.

5.3.3 Results

5.3.3.1 Spatial comparisons

Because MEDIUM and HIGH effort areas show similar patterns, they will both be referred to as "highly" fished areas in the following text.

Size spectra

Fitted curves of the spectra do not show significant differences between less and more fished areas (Table 5.3.1). Although the spectra fitted curves did not differ, differences between areas under increasing fishing disturbance were detected on a finer scale analysis (*i.e.* comparing the "prey" and "predator" weight classes separately). Relative abundance of small fish was higher in highly fished areas (Table 5.3.2). Within the "prey" size-range, evenness indices indicated a decline in diversity for highly fished areas (Table 5.3.3), while richness indices indicated the opposite trend. These results were significant for N^0, d, J', in "prey" weight classes 4–5; for N^1 only in "prey" weight class 5; not significant for N^2. N^0 is the only index to also differ significantly in "predator" weight classes 9–10.

Richness indices had a strong correlation with the number of individuals in the sample (i.e. higher species richness wherever sample size was larger, sample size not shown here). Dominance indices did not show such a correlation. Since richness measures are known to be sample-size sensitive (Hill, 1973; Magurràn, 2004) and the abundance of individuals among "prey" size classes is higher in heavily fished areas, evenness indices, which are more robust to differences in sample-size, must used for this type of comparisons.

Table 5.3.1. Regression parameter estimates and confidence intervals for normalized size spectra, under each fishing effort level (LOW, MEDIUM, HIGH). Log_2 weight class range: 5–13.

		NORMALIZED ABUNDANCE			NORMALIZED BIOMASS		
		Estimate	St err	Pr (>|t|)	Estimate	St err	Pr (>|t|)
LOW	Intercept	6.09	0.08	<2 exp-16	6.29	0.082	<2 exp-16
	Slope	-0.74	0.009	<2 exp-16	-0.44	0.009	<2 exp-16
MEDIUM	Intercept	6.2	0.105	<2 exp-16	6.39	0.108	<2 exp-16
	Slope	-0.74	0.011	<2 exp-16	-0.44	0.012	<2 exp-16
HIGH	Intercept	6.41	0.19	<2 exp-16	6.63	0.19	<2 exp-16
	Slope	-0.78	0.02	<2 exp-16	-0.49	0.02	<2 exp-16

Table 5.3.2. One-way ANOVA results for normalized abundances. Weight classes as groups, fishing effort levels as treatments (LOW, MEDIUM, HIGH). *Post-hoc* pair-wise comparisons, that revealed significant differences at 5% confidence with Tukey and Fisher tests, are indicated with ' * '. Comparisons where only the Fisher test detected significant differences are indicated with ' (*) '.

LOG2 WT	FISHING EFFORT	N0			D			J'			N1			N2		
		Mean	*F*	*P*	*Mean*	*F*	*P*	*Mean*	*F*	*P*	*Mean*	*F*	*P*	*Mean*	*F*	*P*
4	L	6.82	7.665	0.001	0.77	3.43	0.038	0.48	6.57	0.002	2.60	2.64	0.078	2.12	2.096	0.130
	M	8.16		***	0.86		*	0.30		**	2.06			1.70		
	H	8.90			0.97			0.35			2.32			1.89		
5	*L*	9.05	17.563	0.000	0.98	7.5	0.001	0.44	6.44	0.003	2.85	3.43	0.038	2.26	2.678	0.076
	M	10.80		***	1.10		***	0.28		**	2.13		*	1.74		
	H	11.70			1.24			0.34			2.60			2.02		
10	*L*	5.92	3.206	0.046	1.35	1.37	0.261	0.66	0.34	0.712	3.29	0.72	0.489	2.58	0.345	0.710
	M	6.56		*	1.40			0.65			3.32			2.56		
	H	4.60			1.13			0.69			2.90			2.36		

Conclusions on spatial comparisons

Fitted models can mask some of the information contained in the size-related pattern. A specific test in classes 4–5 showed that these "prey" sizes are higher in biomass and abundance where fishing is more intense but lower in diversity, supporting the hypothesis of a predatory release effect (these classes are too small to be caught by most fisheries). This suggests that size-based diversity metrics can be informative, but diversity size-spectra slopes are not a good metric to express this information.

Table 5.3.3. One-way ANOVA of diversity indices within "prey" and "predator" weight classes, across areas under different fishing effort levels (L=LOW, M=MEDIUM, H=HIGH). See text for diversity index symbols. Significance levels are indicated as: '*' ≤ 0.001, '**' ≤ 0.01, '*' ≤ 0.05 . Only weight classes with significant differences are shown.**

			TUKEY (FISHER) *POST-HOC* 5% SIGNIFICANCE			
LOG2 WEIGHT CLASS	F	P	L,M	M,H	L,H	RANKED MEANS
4	3.57	0.033	*			M>H>L
5	5.49	0.006	*			M>H>L
9	3.34	0.041	(*)		*	L>M>H
10	2.01	0.142				M>L>H
11	3.08	0.052				M>L>H
12	2.42	0.096				M>H>L
13	0.43	0.650				M>L>H

5.3.3.2 Temporal comparisons

Diversity (N1) size spectra seem to have changed in time from a dome-like shape to a sinusoid shape (Figure 5.3.1), because of a relative decline in evenness in "prey" size classes (5 and below). Trends are similar over the whole study area; however, for "prey" categories, HIGH fished areas were the most diverse at the beginning of the time-series, while LOW areas appeared the most diverse at the end.

The "prey" weight classes showed a declining trend in J' (Fig 5.3.2), while "predator" size classes appeared relatively stable across the time-series or only slightly declining. This could be observed for all effort areas, but was less marked in the LOW fishing area.

The most visible change appears to have occurred during the gap between the first and second part of the time-series (the 1960s). This could be interpreted as an artefact due to a change in sampling method. However, the sampling protocol and design for SAGF surveys has remained unaltered throughout the time-series. There have been changes of survey vessels. This caused an increase in towing speed which should in turn increase sampling effort and thus the probability of sampling new species. Therefore, if a sampling artefact is present, it is likely to be an increase in time of number of species detected, rather than a decline. Furthermore, the indices shown here should be relatively robust against sample-size effects (for details on sampling methods see Greenstreet and Rogers, 2006 and references therein).

Conclusions on temporal comparisons

Temporal comparisons show a decline of diversity for small fish, further suggesting a predatory release mediated mechanism. They also show that "prey" and "predator" diversity are controlled by different mechanisms and do not change in time at the same rate and scale.

Overall conclusion

Overall these results support the hypothesis that, in heavily fished areas, the reduction of predatory disturbance causes the diversity of prey fish to decrease, while intense fishing disturbance on larger individuals may not affect their diversity.

5.3.4 Corollaries for diversity metrics

- *traditionally measured diversity is not informative of fish community-level changes*

Comparisons with diversity measured on the assemblage as a whole, using the same dataset, both spatially (Greenstreet and Rogers, 2006) and temporally (Greenstreet and Hall, 1996), show that, for each index applied, diversity trends mirror those found for weight class range 4–5.

In the case of communities highly dominated by small individuals, if diversity is measured over the whole assemblage, the index value will be driven by the diversity of the most abundant individuals, i.e. the smaller ones. Hence the diversity of the whole assemblage as commonly measured, may not describe properly the whole fish community.

- *hidden sample-size effects*

If the value of the diversity index applied to the whole community is driven by the diversity of the most abundant size-classes, and the index selected is highly sensitive to sample size, higher diversity will be measured in those areas where these individuals are more abundant. A frequently observed ecosystem effect of fisheries is a relative increase in biomass of small individuals (Jennings and Kaiser, 1998; Gislason and Sinclair, 2000). Thus, since highly exploited areas are more likely to yield larger samples of small individuals, they will appear more species-rich, although in reality they are dominated by few very abundant species as indicated by evenness indices (e.g. Pielou's J'). This may explain some contradictory trends found in studies shown in ICES 2001 and ICES 2002 (in some cases based on the same datasets as presented here).

- *temporal scale of change*

The time series shown here suggests that the main effects of fisheries on diversity have occurred before the 70s and have remained stable ever since. This suggests that, for systems where fishing effort intensity followed similar patterns (e.g. same period of intensification of the exploitation rate) fish diversity may have shifted to a new value before the beginning of available data time-series (e.g. >20 years ago).

5.3.5 Progress made towards developing diversity metrics

These results can contribute to the theoretical linkage of fishing effects to diversity changes

- spatial and temporal results are consistent, suggesting that differences detected are due to a fishing effect
- different trends were detected for different size classes, suggesting the use of size-based metrics is ecologically meaningful
- trends in small fish suggest a predatory release mechanism can explain some diversity patterns, although further testing is required
- some contradictory trends previously found may be explained by the use of sample-size sensitive metrics such as species richness.

Further testing of the predatory release hypothesis, together with studies that are specifically focussed on the mechanisms underlying changes in large fish diversity, may be relevant steps towards the development of diversity metrics.

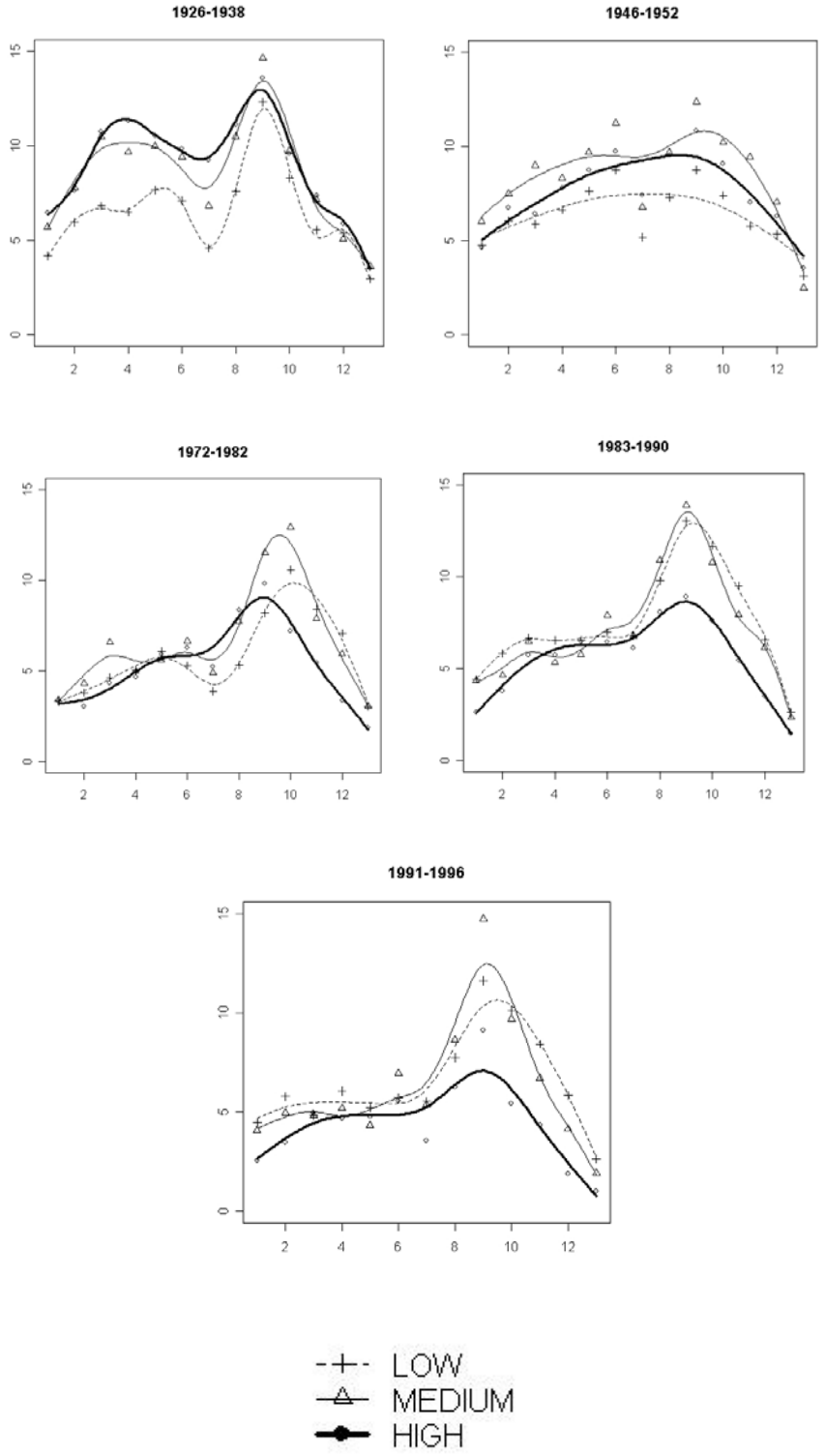


Figure 5.3.1. Diversity (Hill's N1) spectra by fishing effort area (LOW, MEDIUM, HIGH) for different time-periods (fitted GAM curves; only averages are shown for clarity of representation).

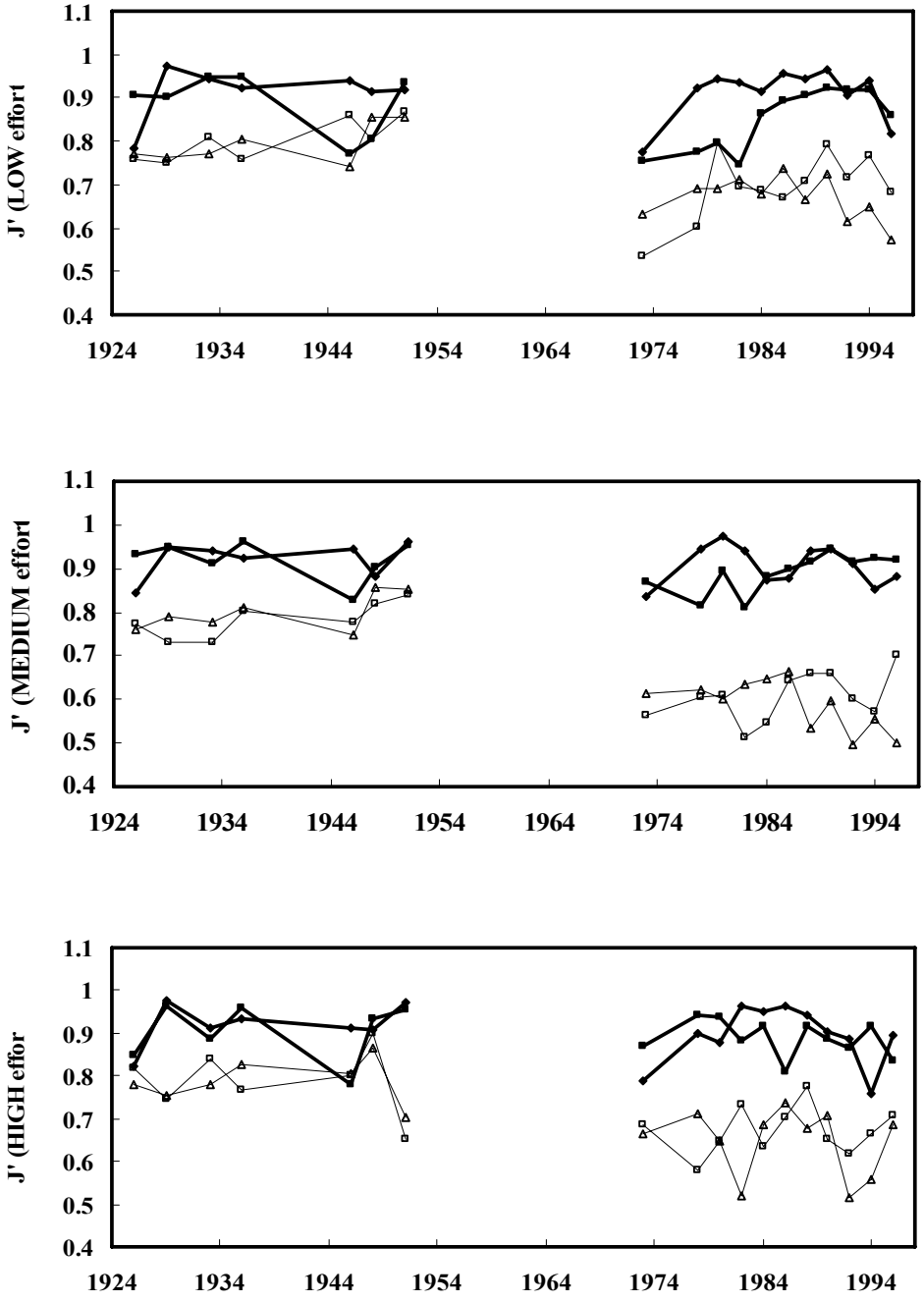


Figure 5.3.2. Diversity (Pielou's evenness) time-series for log_2 weight classes 4, 5 9 and 10. For: a. LOW, b. MEDIUM and c. HIGH fishing effort areas.

5.3.6 References

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6 Trends in and recovery of the proportion of large fish and mean fish weight in the North Sea

Terms of Reference:

ToR f) from current model population estimates and survey data show the historical trend in the proportion of large fish and mean fish weight (North Sea);

ToR g) for current models of North Sea fish communities:

i) determine future trajectories of the proportion of large fish (<30 cm) and mean fish weight under different scenarios of fishing mortality.

ii) from (g-i) determine the time to recover to reference levels in the early 1980s as determined in (f) for the selected indicators.

In 1997 ministers attending the Bergen Intermediate Ministerial Meeting tasked OSPAR with the development of an Ecological Approach to Management (EAM). The OSPAR approach involved the setting of Ecological Quality Objectives (EcoQOs) for ten individual issues of Ecological Quality. Fish communities is issue number five on this list. OSPAR recognised that such an approach would be heavily dependent on indicators. In 2000 therefore, OSPAR sought advice from ICES regarding the identification of appropriate indicators to support an EcoQO based EAM. ICES considered that fishing was the single anthropogenic activity likely to have the greatest impact on fish communities. Bearing in mind that OSPAR's principal objective was the mitigation of the detrimental effects of anthropogenic activities, ICES proposed seven criteria that could be used to identify appropriate indicators for quantifying the detrimental impact of fishing on fish communities. ICES applied these criteria to a variety of potential fish community indicators and determined that indicators of the size structure of fish communities performed best. Following the advice from ICES therefore (ICES 2001), the Element of Ecological Quality for fish communities proposed in the Ministerial Declaration of the 2002 Bergen North Sea Ministerial Conference was "Changes in the proportion of large fish and hence the average weight and average maximum length of the fish community". However, at this time it was not possible to set any operational EcoQOs. Subsequent OSPAR and ICES work has focused on addressing this outstanding issue.

ICES (2006) proposed the following goal for fish communities in the North Sea:

> "Halt as rapidly as possible, and begin to reverse by 2010, both the decline in the mean weight and the proportion of large fish."

To achieve such a goal, short and medium-term operational targets were suggested. The short-term targets were:

1) Halt the decline in the proportion of fish greater than 30cm in length in survey catches immediately;
2) Halt the decline in the mean weight of fish in survey estimates immediately;

and the medium-term targets were:

3) Restore the proportion of fish greater than 30cm in length to 1.4 times 1997 survey estimates by [year "x"];
4) Restore the mean weight of fish to 1.3 times 1997 survey estimates by [year "y"].

The increment values of 1.4 for the proportion of large fish and 1.3 for the mean weight of fish were derived from Scottish August Groundfish Survey (SAGFS) temporal trends published by Greenstreet and Rogers (2006) and assuming that a target point might be located along the trend lines at around the early 1980s, when ICES advice for the commercial stocks was

generally for the "*status quo*" (ICES 2006). However significant indicator trends were only observed for ICES rectangles in the northwestern North Sea that were heavily or moderately fished. As anticipated for indicators that were responsive primarily to variation in fishing pressure, no trends were observed in ICES rectangles that were only lightly fished. Calculating these indicators for the entire area covered by this survey, including the lightly fished rectangles, might therefore alter these increment values.

The targets were expressed as increments of 1997 indicator values because the SAGFS ceased in this year. This particular time series is therefore not available to monitor the effectiveness of management action. It was hoped that comparison of time series derived from different survey data sets would display strong correspondence. Thus even though indicators values calculated in 1997 for different datasets may differ in absolute terms, the relative trends would be sufficiently similar as to allow alternative survey data sets to be used to monitor progress towards the targets. For example, Piet and Jennings (2005) demonstrate similar negative temporal trends in mean weight and mean maximum length of fish for two very different survey data sets, the quarter 1 International Bottom Trawl Survey (IBTS) and quarter 3 Dutch Beam Trawl Survey (DBTS).

WGFE considered that the proportion of large fish and mean fish weight metrics should be applied separately to the demersal fish and the pelagic fish components of the whole fish community. This was because the shorter lived pelagic species are influenced more strongly by variation in recruitment, and therefore variation in the two metrics is likely to be less tightly linked to variation in fishing pressure (Ricker 1995; Ottersen and Loeng 2000; Badalamenti *et al*., 2002; Lekve *et al*., 2002; Wilderbuer *et al*., 2002). Trends in these two components also tend to be opposing rather that coincidental. For example, during the period of the "gadoid outburst" in the late 1960s early 1970s many demersal species increased in abundance considerably, while several pelagic species declined markedly. Combining both components to derive single metrics for the whole fish assemblage therefore risked obscuring the response of each component to changes in fishing pressure, which has been considered to be a prerequisite requirement of a "good indicator" (ICES, 2001). This is therefore reflected in the analyses described below. Several published studies describing trends in fish community indicators focus on only the demersal component (e.g. Greenstreet and Hall 1996; Greenstreet et al 1999; Greenstreet and Rogers 2006)

6.1 Trends in the proportion of large fish and mean fish weight in the North Sea surveys

6.1.1 Scottish August Groundfish Survey (SAGFS)

In previous analyses of the SAGFS spatial concerns were also of interest. Consequently trawl samples were aggregated in time (year groups) so as to ensure adequate sampling effort in each time-area cell. This was not necessary in this analysis where simple trends in the indicator values over the whole 75 ICES statistical rectangle area covered by the survey were all that were required. The data set and analytical procedures have been described in detail elsewhere (Greenstreet and Hall 1996; Greenstreet et al 1999; Greenstreet and Rogers 2006). The extent of the area covered by the survey is indicated in Figure 6.1.1.1. Trends in four indicator values (proportion of fish >30cm by both number and biomass, mean weight of all fish, and mean weight of fish >30cm) are shown in Figure 6.1.1.2. Given the wide variance in the value of each metric over time, and for direct comparison with the data presented by Greenstreet and Rogers (2006), the analysis was repeated based on the same year group aggregations as used in this earlier study Figure 6.1.1.3.

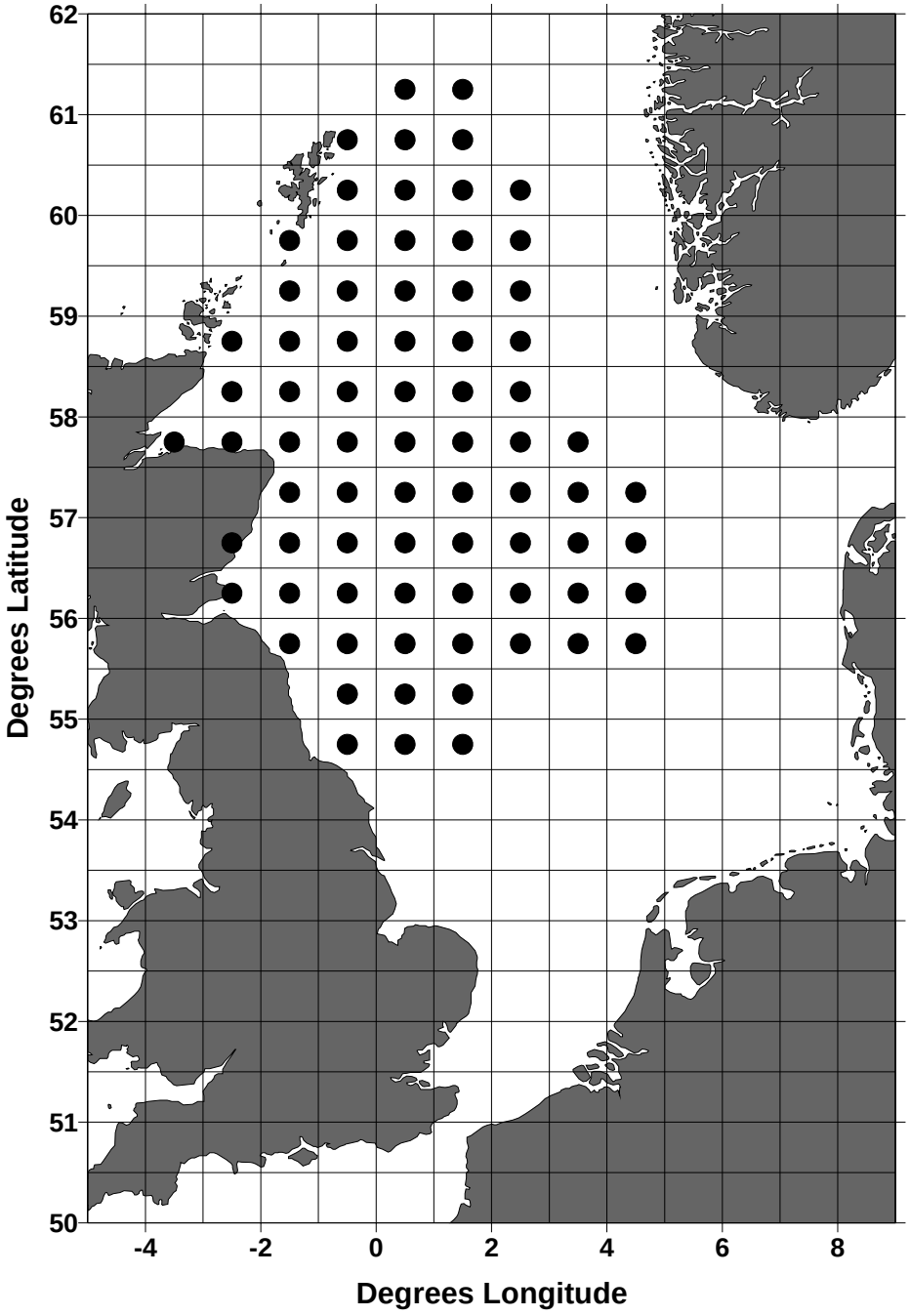


Figure 6.1.1.1. Map of the area covered by the SAGFS.

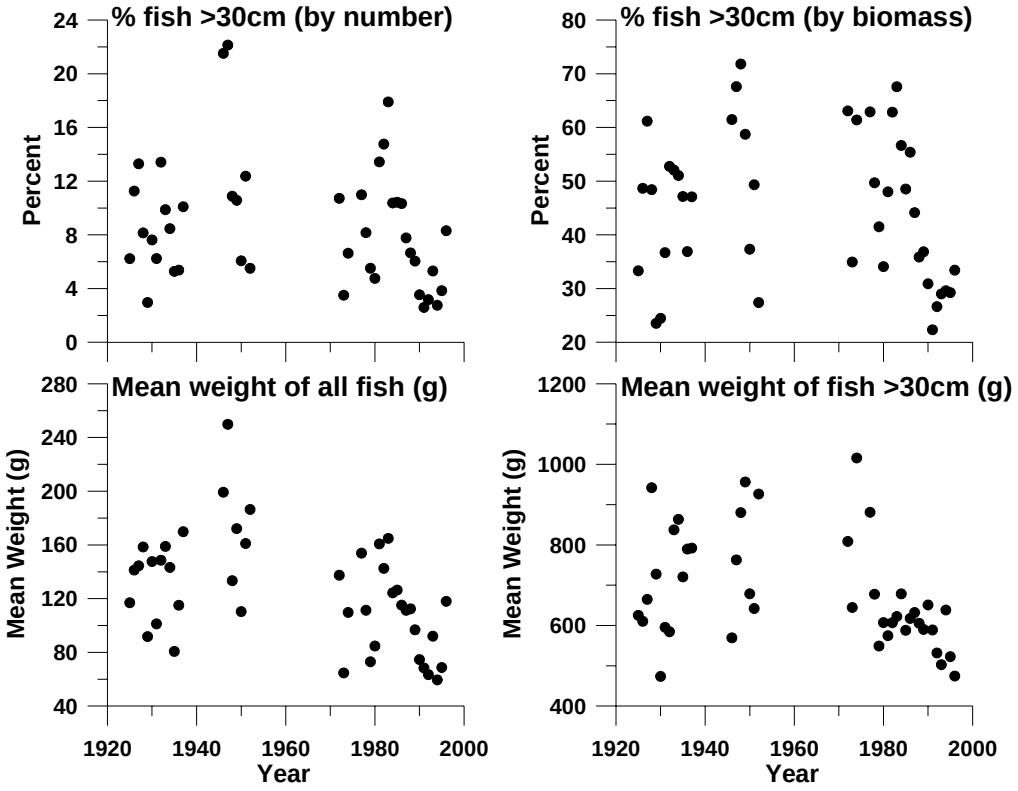


Figure 6.1.1.2. Trends in the proportion of fish >30cm by number, proportion of fish >30cm by biomass, mean weight of all fish, and mean weight of fish >30cm.

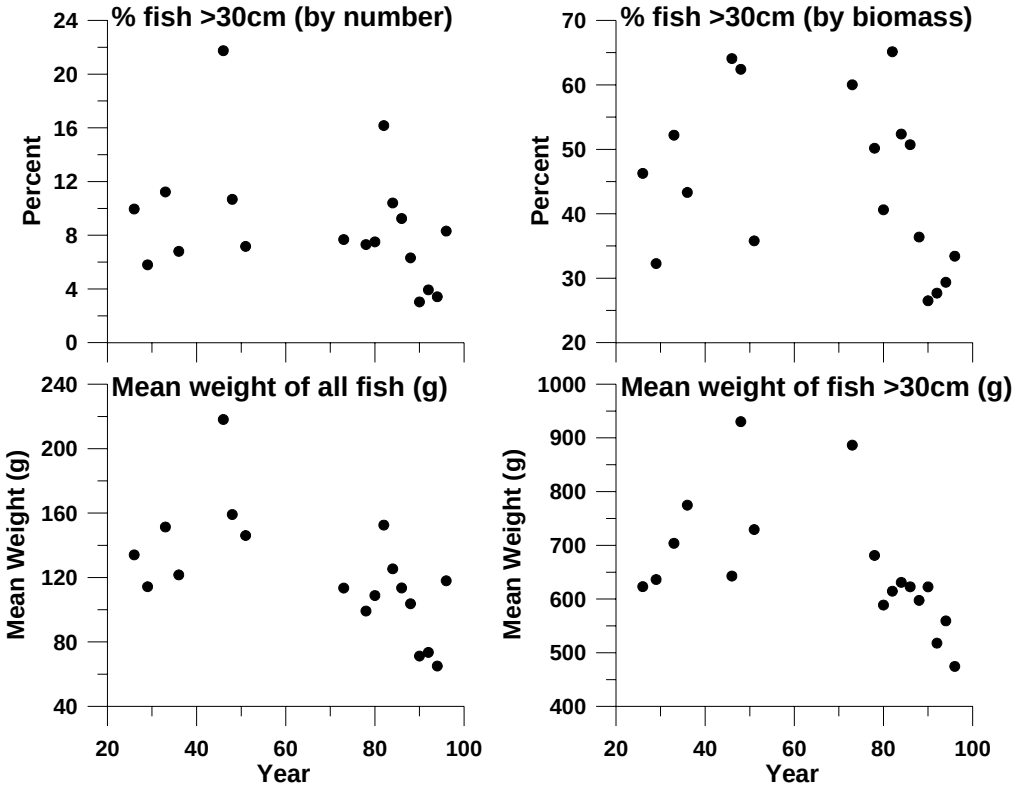


Figure 6.1.1.3. Aggregated year-group trends in the proportion of fish >30cm by number, proportion of fish >30cm by biomass, mean weight of all fish, and mean weight of fish >30cm.

6.1.2 Quarter 1 International Bottom Trawl Survey (Q1 IBTS)

The IBTS quarter 1 (Q1) data was analysed to look for changes in the proportion of demersal fish greater than 30cm and changes in average fish weight over time. The Q1 data for the time period 1983 to 2006 was used in this analysis as prior to 1983 different survey gears were used by different countries. In 1983 all countries participating in the IBTS began using the GOV with a trawl duration of 30 minutes (except Scotland). In 1999 Scotland changed from 1 hour trawls to 30 minute trawls to come in line with the rest of the participating countries. Only demersal fish were used in this study so all species regarded as pelagic were excluded from the dataset. All hauls coded as invalid and hauls where some species were counted and not measured were also excluded. Length-weight relationships were used in order to calculate the biomass of each species in each length category. The proportion of demersal fish (by number) decreased significantly (R^2=0.56) from 1983 to 2001 (Figure 6.1.2.1a). From 2002 the proportion of fish >30cm increased to a peak in 2004 and has since decreased sharply. Figure 6.1.2.1a also shows the proportion of haddock (by number) >30cm. The proportion of haddock >30cm shows a similar pattern to the plot for all demersal species combined, particularly towards the end of the time series. The proportion of demersal fish (by biomass) decreased significantly (R^2=0.62) over the time period 1983 to 2001 and is shown in Figure 6.1.2.1b. From 2002 the proportion of fish (by biomass) >30cm increased to a peak in 2004. The proportion of haddock (by biomass) >30cm again tracks the trend shown by all demersal fish >30cm. The mean weight of all demersal fish decreased from 1983 to 2000 (R^2=0.57) as shown in Figure 6.1.2.2. From 2001 the mean weight increased to a maximum of 110g in 2004 and has since decreased.

The results of this analysis show that in the North Sea, changes in the proportion of fish greater than 30cm from 2001 to 2006 were strongly influenced by haddock. An EcoQO based on size in the North Sea appears to be very sensitive to large recruitment events such as the large haddock recruitment as shown here and is perhaps therefore not the most suitable indicator of fishing impact.

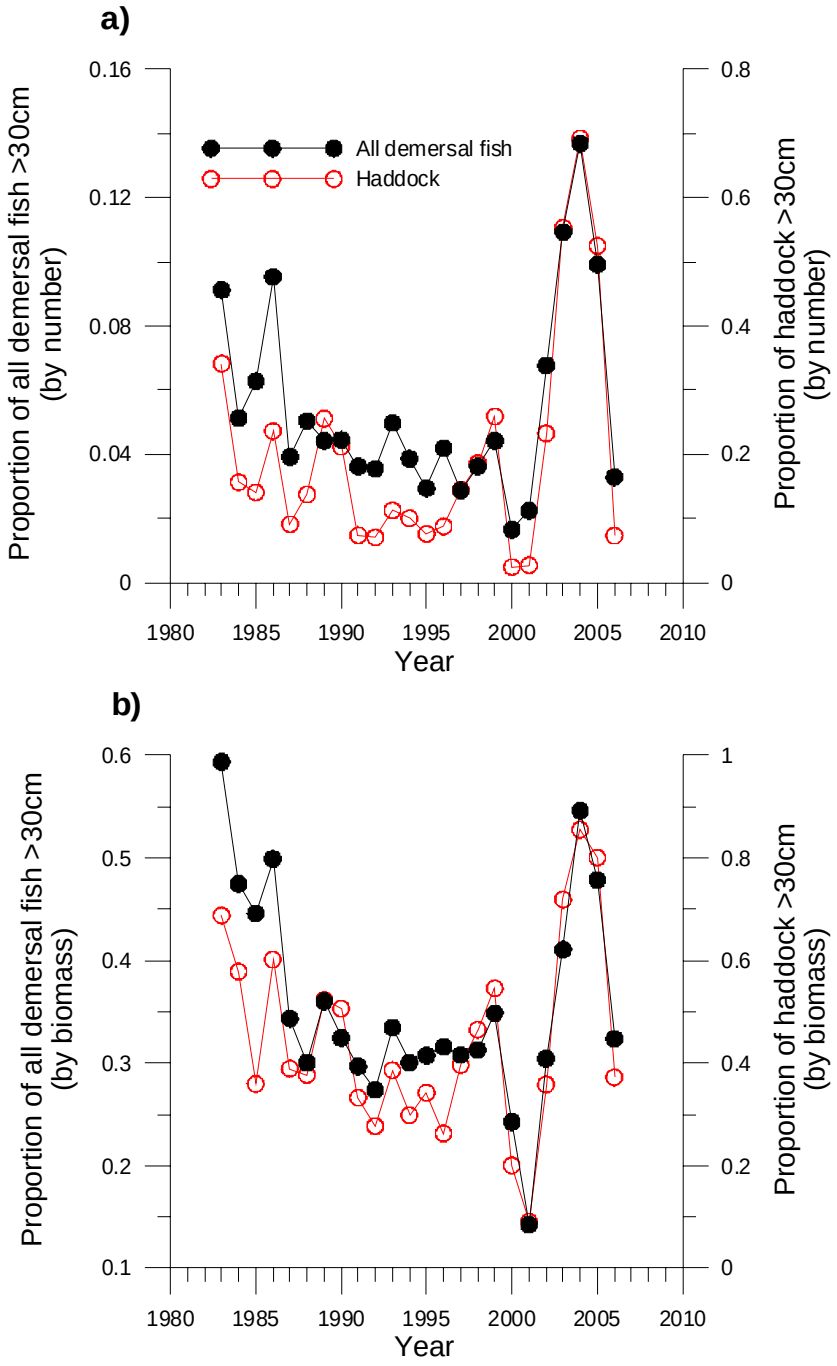


Figure 6.1.2.1. Trends in the proportion of all demersal fish and haddock greater than 30cm by a) number and b) biomass from 1983 to 2006.

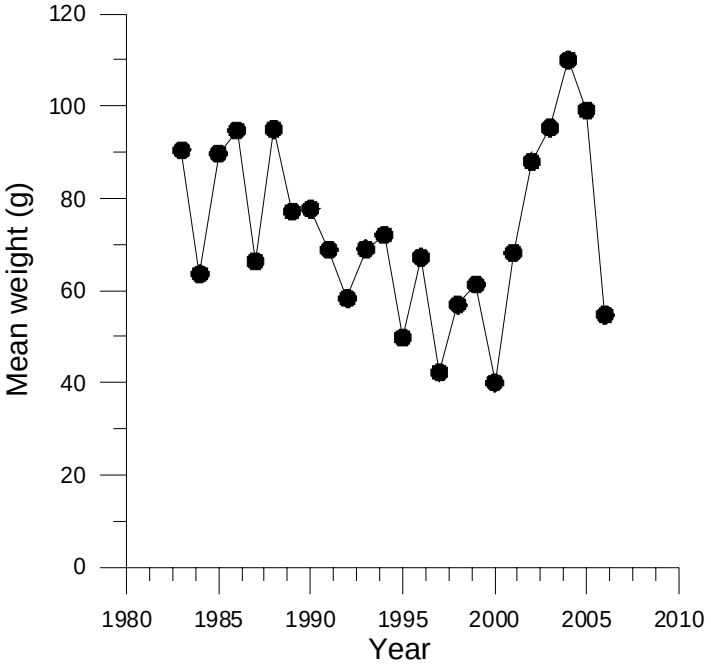


Figure 6.1.2.2. The mean weight of all demersal fish from 1983 to 2006.

6.2 Using MSVPA to estimate historical trends in ecosystem indicators in the North Sea

MSVPA has previously been used to evaluate the effect of fishing mortality on the slope and intercept of the size spectra of demersal fish in the North Sea (Rice and Gislason 1996). The investigations showed that these two community metrics were sensitive to fishing mortality. The effect of fishing mortality on the indicators (a) mean weight in the community and (b) biomass proportion or (c) abundance proportion of fish larger than 30 cm has not been evaluated using the MSVPA. However, a simulation study has estimated the power to detect trends of survey based indicators of mean weight in the community to be greater than that of survey based indicators of slope and height of the size spectra (Nicholson and Jennings 2004). The working group considered the development in a total of seven community estimators: Mean weight in the community, $\overline{W}$, biomass proportion of fish larger than 30 cm, $PB_{>30cm}$, abundance proportion of fish larger than 30 cm, $PN_{>30cm}$, slope of the size spectra, $slope$, height of the size spectra at a length of 11 cm (10 to 12 cm, the smallest length group included in the estimation), $height$ and number of stocks below B_{pa}. Further, the historical trends in the indicators mean weight and the proportion of large fish are requested by ICES under TOR f. These requests have been posed in spite of comments from WGECO that 'it appears that there is a relationship with fisheries for metrics of both mean weight and mean maximum length of fish in the community. However, this relationship is not straightforward, not well understood and certainly not tightly linked in space and time. As such these two metrics will be poor performance metrics, and should preferably be used only for surveillance of the fish community (ICES 2004). Since then, it has been demonstrated that neither mean weight nor slope and intercept of the size spectra are sensitive to plausible trends over a sampling period of less than 5 years (Nicholson and Jennings 2004).

6.2.1 Methods

The estimations used the MSVPA key run produced by the study group on multispecies assessments in the North Sea (SGMSNS) (ICES 2006b). This run includes grey gurnards as predators as well as an extended set of seabird consumptions. The estimates of mean weight at age of a given species were used directly rather in the estimations rather than first estimating weight distribution of that age group. Mean length was estimated by applying the weight

length relationship of each species as given in Fishbase (Norway pout) or Coull *et al.* 1989 (all other species). This procedure has been used successfully in the past to estimate size distribution of the community from MSVPA output (Rice and Gislason 1996). Fish smaller than 10 cm or larger than 110 cm were excluded from the calculations. To estimate the size spectra slope and height, numbers at length were pooled in groups of 10–12, 12–15, 15–18, 18–22, 22–27, 27–33, 33–40, 40–49, 49–60, 60–74, 74–90 and 90–110 (width of each length group is 0.2 units of natural log to the length). The community estimators used were calculated both for demersal [cod (*Gadus morhua*), haddock (*Melanogrammus aeglefinus*), whiting (*Merlangius merlangus*), saithe (*Pollachius virens*), Norway pout (*Trisopterus esmarki*), plaice (*Pleuronectes platessa*) and sole (*Solea solea*)] and pelagic [herring (*Clupea harengus*), sprat (*Sprattus sprattus*) and sandeel (*Ammodytes spp.*)] and demersal species together. The species together constitute about 84% of the number of fish caught in the English groundfish survey (ICES 2006a). Only 1st quarter estimates were used to assure comparability with B_{pa}.

Two estimates of fishing effort were calculated, and average F and an average standardised F. Average F was calculated as the average F(2–4) of the demersal stocks (cod, haddock, whiting, saithe, Norway pout, plaice and sole). In 2006, the working group investigated an alternative estimate in 2006 where F of a given species was weighted by SSB. However, this estimate was not very different from the simple average and the group concluded that the weighting did not seriously improve estimates was therefore not worth the extra work (ICES 2006a). An average standardised F was calculated by first expressing the F(2–4) of each species in units of standard deviations from the mean and then averaging these values over species. The average standardised value was multiplied by the standard deviation averaged across species and the mean F over all species was added. This assures values which are on a scale comparable to the average F-values. The standardisation procedure gives higher weight to changes in F of species with a relatively constant F, whereas the average F gives equal weight to all nominal changes in F. The community indicator in a given year was then compared to the average F and standardised F over the past three years (year-1, year-2 and year-3).

6.2.2 Results

The temporal development in F and indicators are seen in Figures 6.2.2.1 and 6.2.2.2, respectively. The standardised and non-standardised fishing mortalities were highly correlated (correlation=0.97) and there was little effect of standardisation. Mean weight was highly influenced by sandeel stock size when all stocks were considered, but even when only demersal species are included, the development is highly erratic with no clear trend. The proportion of fish larger than 30 cm exhibits a similar lack of pattern and in addition has occasional spikes caused by large year-classes of haddock reaching a size greater than 30 cm (the 1963, 1967 and 1999 year classes). The slope of the size spectra increased over 1960s, remained at a high level during the early 1970s and decreased from around the mid 1970s. The number of stocks below B_{pa} increased dramatically in the early 1980s, but even before this period, 2 and occasionally 3 stocks were below B_{pa} in some years. Significant linear trends were seen only in the slope and height of the size spectra and the number of demersal stocks below B_{pa} (Table 6.2.2.1). Of the demersal indicators, only slope and height were significantly correlated with the number of stocks below B_{pa}. (Table 6.2.2.2, Figure 6.2.2.3). When the estimation of the indicators was based on all species, the proportion of large fish was also correlated with the number of stocks below B_{pa}, though the correlation was less than that with the slope of the size spectra. Fishing mortality was positively correlated to the number of stocks below B_{pa}, as expected (Table 6.2.2.3, Figure 6.2.2.4). It should be noted that the two are not statistically independent. However, F was also correlated to the height of the size spectra and hence the abundance of fish too small to be caught in commercial trawls. This result may be caused by a link between F and SSB and between SSB and recruitment. The

mean weight and proportion of large fish in the demersal community was uncorrelated to F whereas the mean weight in the entire community was negatively correlated to non-standardised F.

Table 6.2.2.1. Trend in community indicators from 1963–2003 MSVPA output. Significant values are in bold and significant at the 5%(*), 1% (**) or 0.1% (***) level. Demersal: demersal community only, All: both demersal and pelagic species together.

INDICATOR	DEMERSAL	ALL
Mean weight	-0.04	0.02
Biomass proportion l>30	0.06	-0.28
Abundance proportion l>30	0.09	-0.08
Slope	**-0.36***	**-0.45****
Height	**-0.47****	-0.23
Number below Bpa	**0.79*****	**0.77*****

Table 6.2.2.2. Correlation between the number of stocks above B_{pa} and community indicators from 1963-2003 MSVPA output. Significant values are in bold and significant at the 5%(*), 1% (**) or 0.1% (***) level. Demersal: demersal community only, All: both demersal and pelagic species together.

INDICATOR	DEMERSAL	ALL AND BPA(DEMERSAL)
Mean weight	0.11	-0.16
Biomass proportion l>30	-0.13	**-0.57*****
Abundance proportion l>30	0.04	**-0.36***
Slope	**-0.52*****	**-0.69*****
Height	**-0.50*****	-0.04

Table 6.2.2.3. Correlation between fishing mortality (averaged over the previous 3 years) and community indicators from 1963–2003. Significant values are in bold and significant at the 5%(*), 1% (**) or 0.1% (***) level. Demersal: demersal community only, All: both demersal and pelagic species together. F non-standardised: simple average across the demersal community. F standardised: F expressed in units of standard deviations from the mean for each species and the averaged across species.

	F NON STANDARDISED		F STANDARDISED	
Indicator	Demersal	All	Demersal	All
Mean weight	0.10	**-0.33***	0.11	-0.30
Biomass proportion l>30	-0.09	-0.17	-0.14	-0.21
Abundance proportion l>30	0.05	-0.24	0.02	-0.26
Slope	0.14	-0.14	0.12	-0.15
Height	**-0.66*****	**-0.53*****	**-0.65*****	**-0.53*****
Number below B_{pa}	**0.43****		**0.40***	

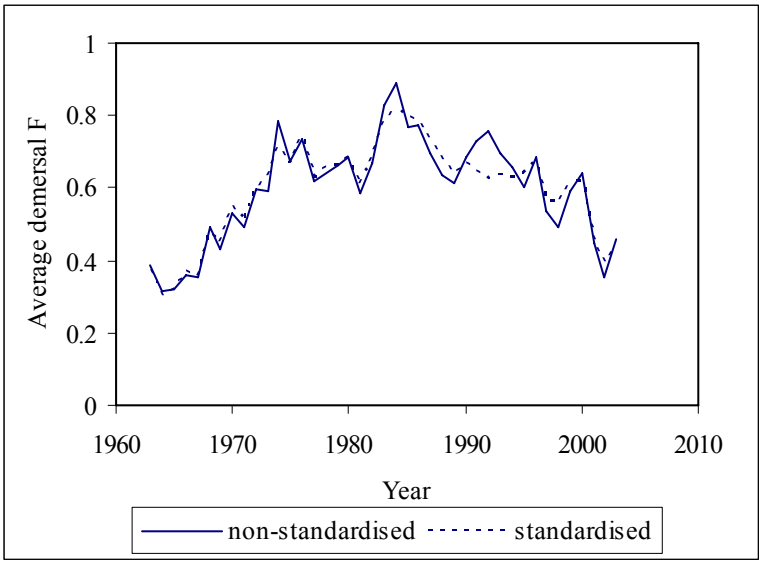


Figure 6.2.2.1. Average fishing mortality from MSVPA. Standardised within species prior to averaging (standardised) and simple average (non-standardised).

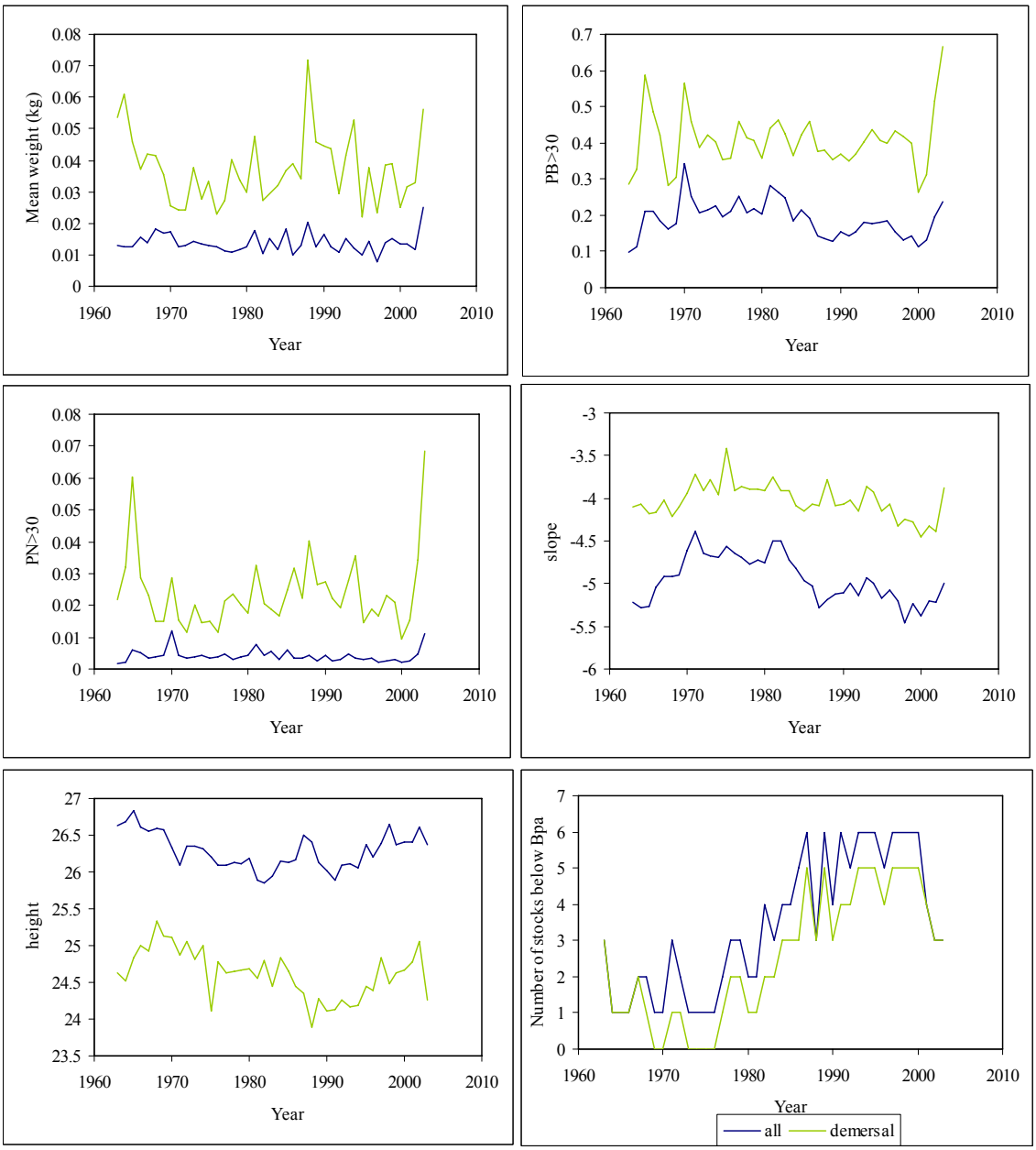


Figure 6.2.2.2. Historical development in ecosystem indicators for the North Sea. Values estimated from MSVPA stock estimates of all stocks (all) or demersal stocks (demersal).

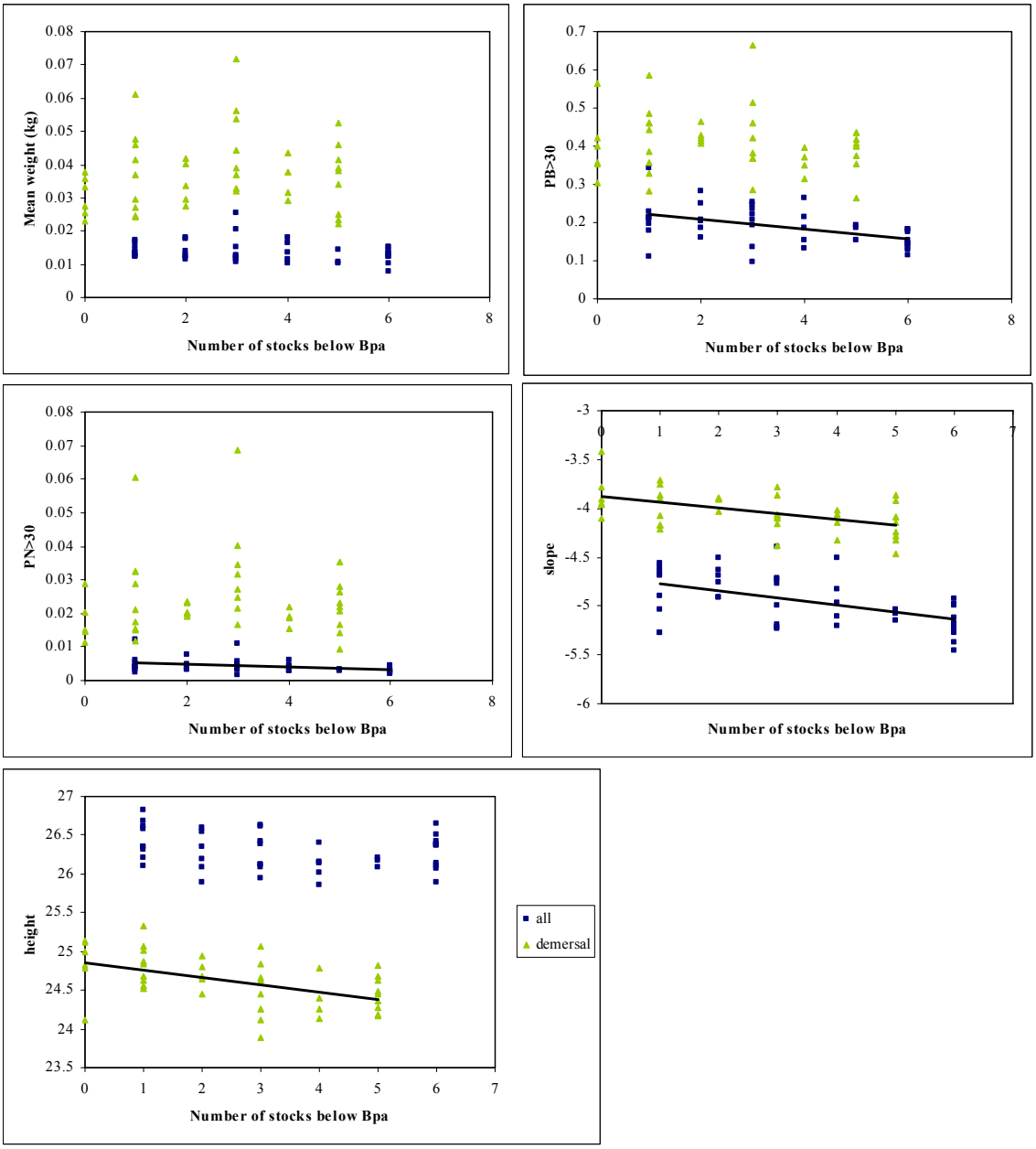


Figure 6.2.2.3. Ecosystem indicators estimated from MSVPA as a function of the number of stocks below B_{pa}. Regression lines are inserted in cases where a significant correlation was found.

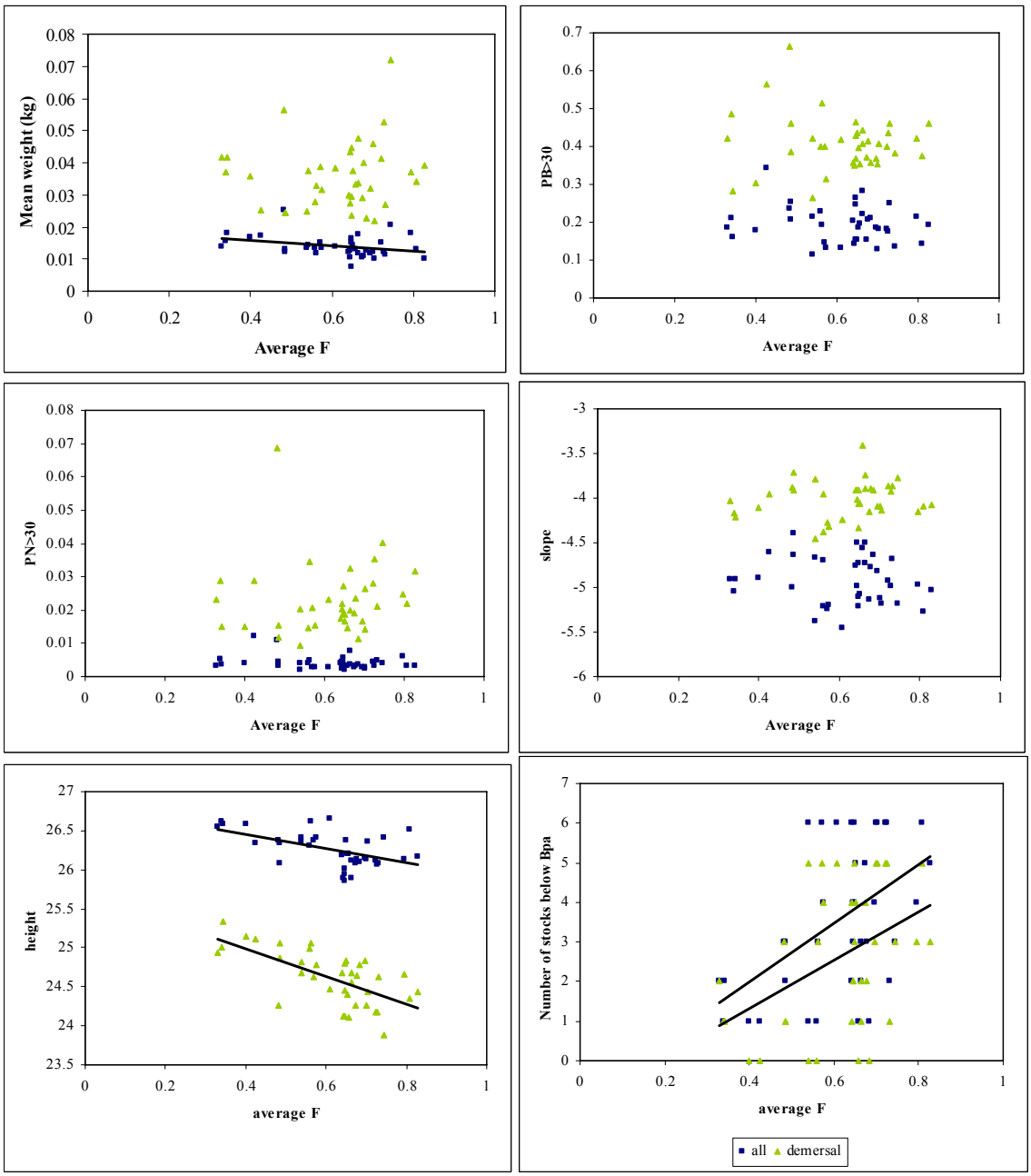


Figure 6.2.2.4. Ecosystem indicators estimated from MSVPA as a function of average fishing mortality. Regression lines are inserted in cases where a significant correlation was found.

The erratic nature of the three indicators [(a) mean weight of fish in the community (b) proportion of large fish by numbers (c) proportion of large fish by biomass] derived from the MSVPA output meant that it is difficult to define a particular value in which they reflect a "healthy" community state, i.e. a target; therefore, using these indicators as Ecosystem Quality Objectives poses a problem. As the proportion of large fish indicators do not show a significant trend with community F, they appear unresponsive to fishing mortality and thus may not be amenable to management control. Though the indicators may be variable and have a low sensitivity to changes in F, they still reflect a composite community structure and thus can be used to monitor direction of community change towards desirable states.

6.3 Projections of the community large fish and mean weight indicators under different fishing scenarios in a single species context

Because of the difficulties using MSVPA in forecast mode, single species projections for the seven demersal MSVPA species were conducted and the proportion of large fish indicators (PLF) and mean individual weight for the combined results were calculated. These are single species projections interpreted in a multispecies/community context therefore there is no guarantee that they reflect the community reality quantitatively or qualitatively. These projections should simply be used as a means to show what a desirable kind of projection results of a multispecies model might look like in order to be useful to management. If these kinds of projections are to be used as a basis for management advice, they should be redone by the appropriate stock assessment expert groups to conform with ICES standards on assessment and projection.

6.3.1 Methods

Single species projections using the seven MSVPA demersal species (cod, haddock, Norway pout, plaice, saithe, sole, whiting) were performed. The results of the MSVPA historical reconstruction for these species provided the basis for the projections though here the species were not linked to each other dynamically hence these are the results of seven independent single species projections.

Numbers at age were taken from the first quarter of 2003 in the MSVPA output to start the projections. Weight at age for calculation of biomass was taken from first quarter 2003. The maturity ogive was the same as used in the single species assessments – these are also used in MSVPA. M at age was the mean quarterly value from 1994 to 2003 and then multiplied by four to make an annual value for the annual projections. The M value was Z-F and therefore contains both the predation M and residual M components of natural mortality.

The fishery selectivity pattern for projection was taken as the average F at age for the recent period (2001–2003). Fpa values apply to ages 2–4 for cod and haddock, 1–2 for Norway pout, 3–6 for plaice and 2–6 for sole and whiting and were distributed according to the mean F over these ages from 2001–2003 and then distributed over all ages according to the mean F at age pattern.

Recruitment (R) was modelled as SSB*R/SSB where the R/SSB value was resampled from the MSVPA reconstruction. The maximum recruitment level for a species was fixed at the maximum observed in the series; hence the stock recruit function used here is a hockey stick where the slope is randomly sampled from the observed slopes for the bootstrap projections. Projections do not include 0-groups and therefore R is to age 1.

Projections were made for a 25 year period and are based on the median value from 1000 bootstrap runs. The 5% and 95% quantiles are not reported owing to the dangers of interpretation of these values as a measure of confidence given the huge conceptual assumptions made by interpreting single species projections in a multispecies context. Green dashed lines in the figures are approximate values for the indicators at a time when the community was in the 'safe' (green) zone in the late 1960s (Fig 6.4.2.6). They represent the mean annual values from 1967 to 1970 inclusive (see Figure 6.2.2.2)

6.3.2 Results

Fishing at Fpa (Table 6.4.2.1) leads to an increasing proportion of large fish (PLF) by numbers and biomass over the 25 year projections (Figure 6.3.2.1). Similarly, mean individual weight and fish biomass increases. Declines were shown in the first three years for PLF but this is owing to existing age structure of the populations input in 2003. In all cases, the indicator levels from the late 1960s exceeded, that is if they were to be set as targets, then the targets

were met. For PLF-numbers this take about 8 years and for PLF-biomass it takes about 12 years.

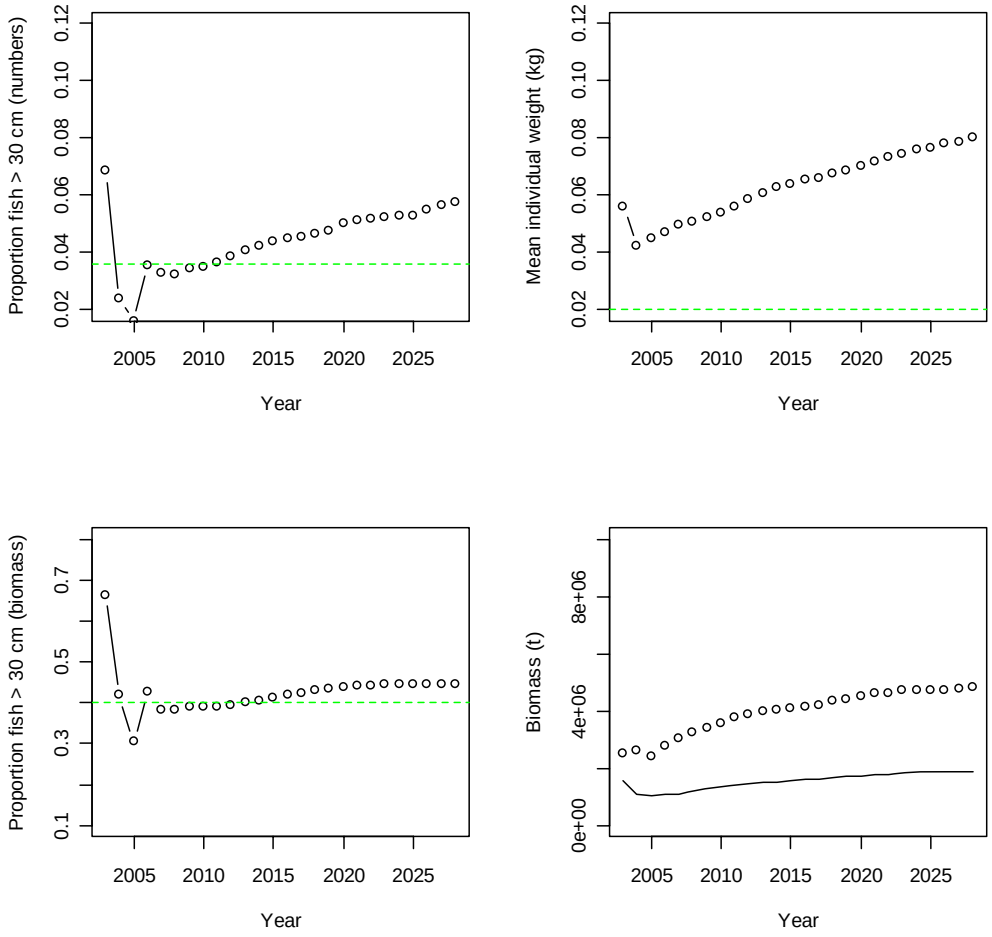


Figure 6.3.2.1. Projection of single species cohort models for the seven demersal MSVPA species while fishing at Fpa with approximate target values for indicators from the late 1960s indicated by the green dashed lines. Medians of 1000 Monte Carlo projections are presented where recruits per spawner were resampled from the historical data. The dotted line in the lower right panel represents total biomass for the seven demersal species, while the solid line represents the mature biomass; there is no target value for total biomass.

When projections are run with fishing at Fpa * 0.75, which would be a more conservative harvesting strategy closer to Fmsy, results are qualitatively similar to fishing at Fpa (Figure 6.3.2.2). Quantitatively the proportions of large fish by numbers or biomass are larger and targets are met within 4 years of the start of the projection.

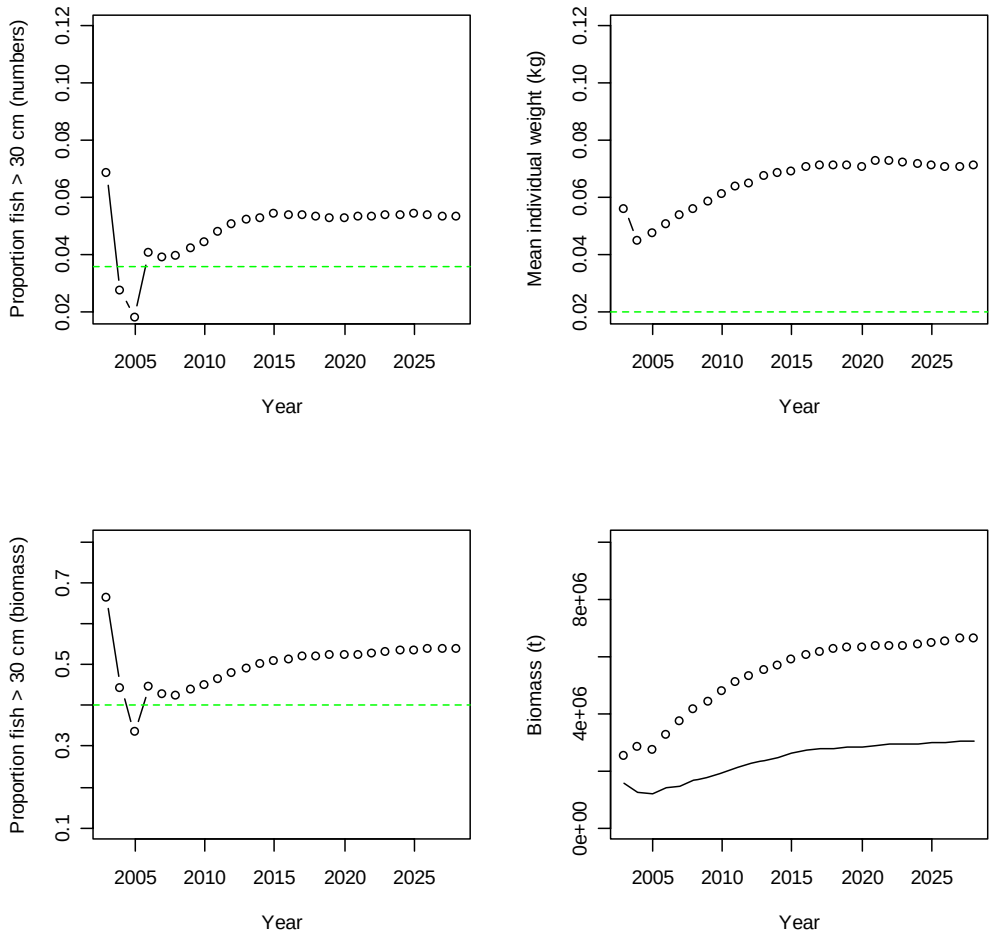


Figure 6.3.2.2. Projection of single species cohort models for the seven demersal MSVPA species while fishing at 75% Fpa with approximate values for indicators from the late 1960s indicated by the green dashed lines. Medians of 1000 Monte Carlo projections are presented where recruits per spawner were resampled from the historical data. The dotted line in the lower right panel represents total biomass for the seven demersal species, while the solid line represents the mature biomass, there is no target value for total biomass.

The projections run in the absence of fishing suggest that the virgin steady state values for the proportion of large fish by number and biomass would be about 12% and 80% respectively while the mean weight of fish would be about 140 g. Total biomass levels would be on the order of 45 million tonnes. These absolute biomass levels should not be taken too seriously given that they will be the result of the balance between recruitment and natural mortality, neither of which shows non-linear compensatory dynamics related to density dependence or predatory functional responses, respectively. The plots are shown here merely to indicate virgin steady state indicator values that this model produces.

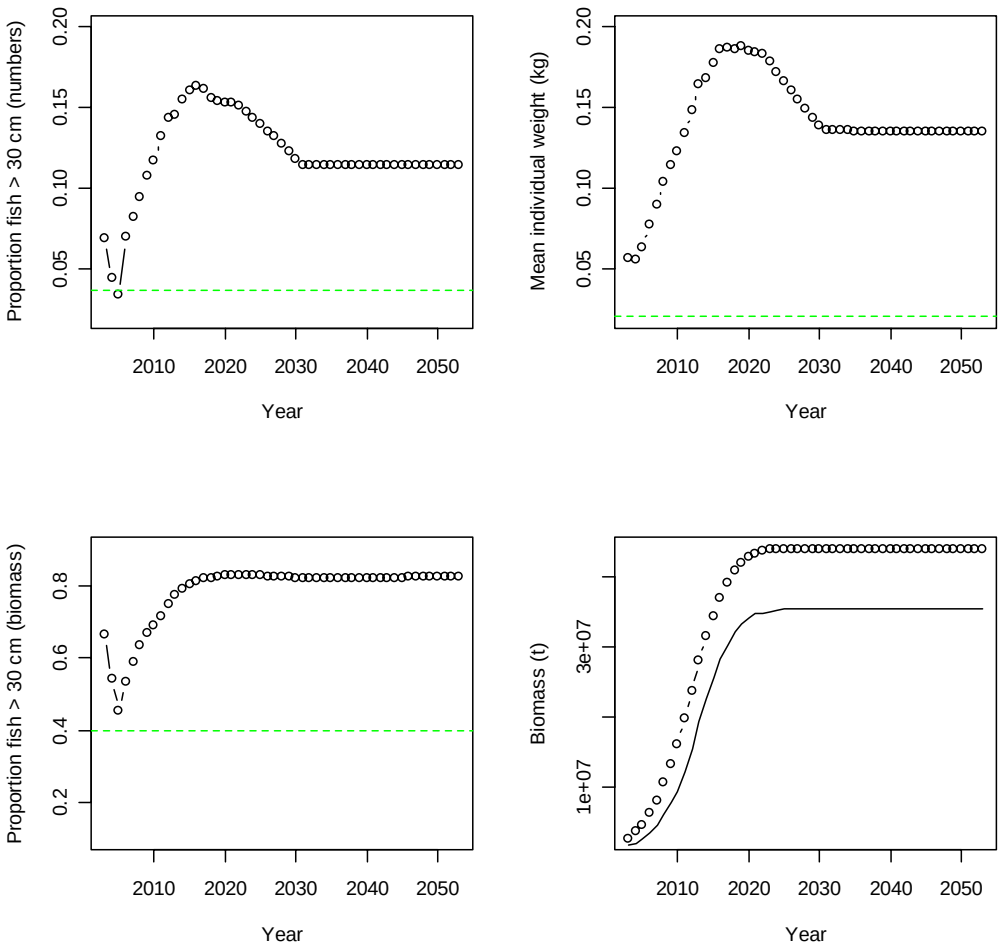


Figure 6.3.2.3. 50 year projection of single species cohort models for the seven demersal MSVPA species in the absence of fishing with approximate values for indicators from the late 1960s indicated by the green dashed lines. Medians of 500 Monte Carlo projections are presented where recruits per spawner were resampled from the historical data. The dotted line in the lower right panel represents total biomass for the seven demersal species, while the solid line represents the mature biomass, there is no target value for total biomass.

It should be noted that a constant smooth steady state is reach in this model (Figure 6.3.2.3) even though R/S is randomly sampled because if stock biomass gets large enough R is always at it maximum which was a hard limit and thus the projection become deterministic once the R asymptote is reached.

6.4 Deciding the target

To decide a target for any ecosystem indicator, the first step is to determine a period where the ecosystem was considered to be in a desirable state. The target of the indicator can then be set as the average value within this period and the range of values observed in this period are indicative of acceptable variation in the indicator. The working group examined two potential methods to estimate the target period, one was based on the proportion of stocks which were estimated to be below B_{pa} and the other on an estimate of the average state of the ecosystem judged by average SSB and F compared to average values of B_{pa} and F_{pa}.

6.4.1 The proportion of stocks estimated to be below B_{pa}

The proportion of stocks estimated to be below B_{pa} using single species (ICES 2006c) and multispecies assessments are seen in Figure 6.4.1.1. Only the years from 1963 are displayed as before this time, SSB is only estimated for two stocks. The number of stocks included in the singlespecies estimates ranges from 4 in 1963 to 7 from 1983 onwards and hence the data prior to 1963 are not strictly comparable to MSVPA estimates which are based on 7 stocks for the entire period. In the early period, the proportion of stocks below B_{pa} varied between 0 and 0.43 with no trend. However, from 1984 onwards the proportion remains consistently above 0.4. Hence there is a clear shift in the general state of the assessed stocks from 1983 to 1984. The average state of the indicators (from MSVPA) examined in the period 1963–1983 and 1984–2003 is seen in Figure 6.4.1.2. There does not appear to be consistent differences between the two periods in any indicators beside the number of stocks below B_{pa}.

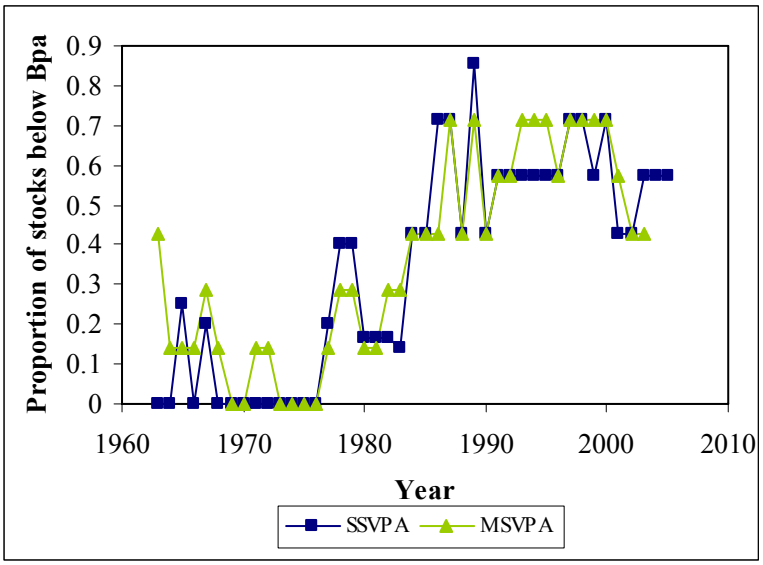


Figure 6.4.1.1. The proportion of demersal stocks estimated to be below Bpa using single species VPA (SSVPA) and multispecies VPA (MSVPA).

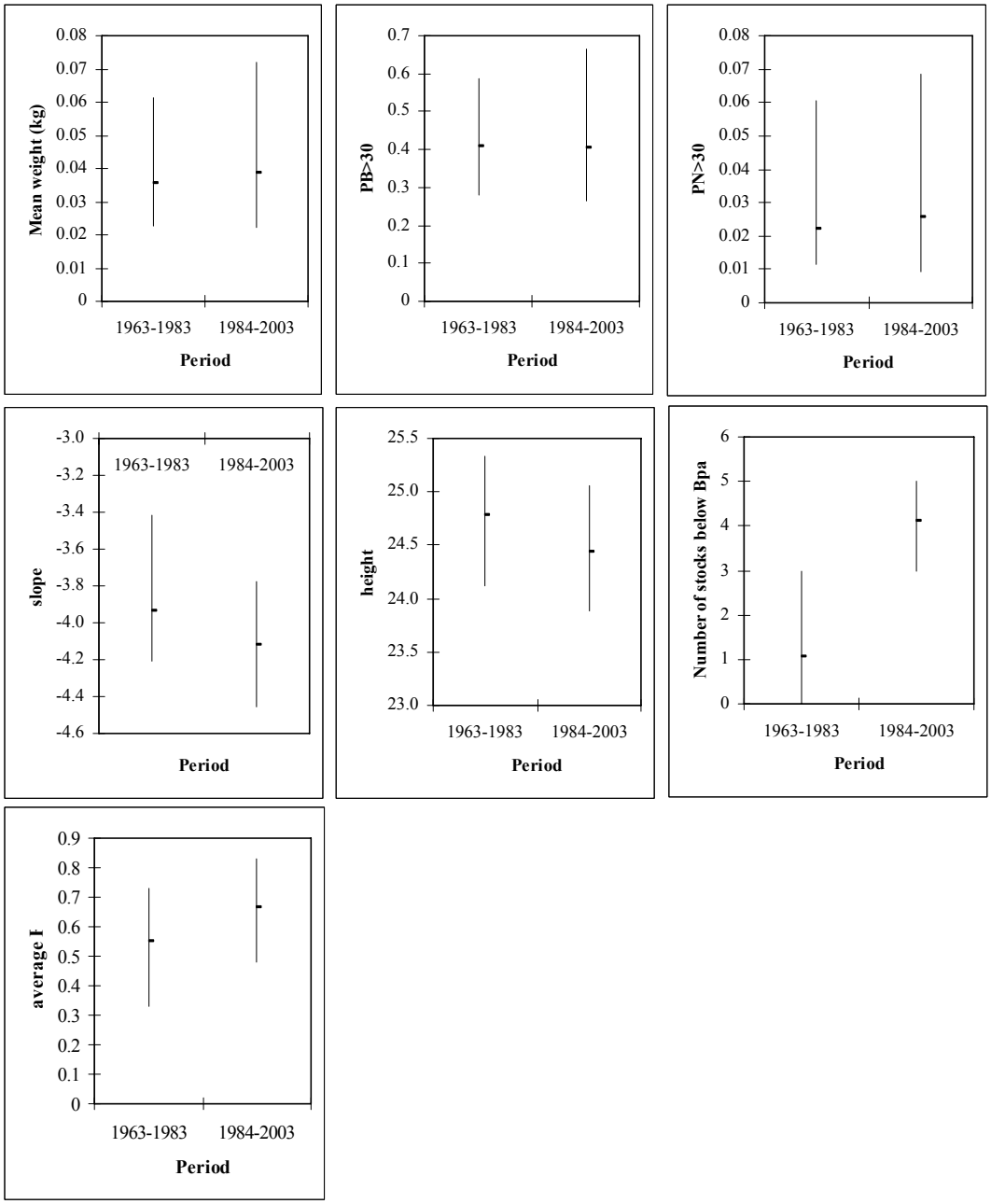


Figure 6.4.1.2. Average, minimum and maximum values of ecosystem indicators estimated from MSVPA output for the period 1963 to 1983 and in the period 1984 to 2003. Horizontal markers indicate average value of the indicator, bars indicate maximum and minimum observed.

6.4.2 Average state of the ecosystem judged by average SSB and F

ICES advice and stock assessment outputs were examined to identify a point in time over the last 44 years when fishing impact on the North Sea fish community might have been considered to have been within reasonable bounds. Simply considering ICES advice alone was not considered to be adequate because the objectives underlying this advice may have changed subtly over the years. Early in the time period it was felt that the underlying emphasis may have tended towards heavier exploitation of the stocks so as to protect the fishing industry, whilst in more recent years, the focus may have become more precautionary. Evidence for this is provided by the fact that in recent years limit and precautionary levels of spawning stock biomass (B_{lim} and B_{pa}) and fishing mortality (F_{lim} and F_{pa}) have been set for most commercially exploited stocks in the North Sea (Table 1). B_{lim} is the spawning stock biomass at which recruitment potential is likely to be compromised, while B_{pa} is the spawning stock

biomass, which given the uncertainty in the stock assessment, the possibility that the stock is actually lower than B_{lim} is very low. F_{lim} is the level of fishing mortality likely to drive the stock to B_{lim}, while F_{pa} is the level of fishing mortality that can be sustained by the stock without it being driven below B_{pa}. Data provided by the stock assessments was therefore analysed in an effort to find a more objective approach to identifying a period in recent time when fishing impact on the fish community may have been considered acceptable.

Table 6.4.2.1. Limit and precautionary reference points for spawning stock biomass and fishing mortality for the main assessed fish stocks in the North Sea (ICES 2006). Grey shaded italics cells show values assumed for the purposes of our analyses, since at present these values are undefined (see text for details).

ASSEMBLAGE	SPECIES	B_{LIM}	B_{PA}	F_{LIM}	F_{PA}
Demersal	Cod	70,000t	150,000t	0.86	0.65
	Haddock	100,000t	140,000t	1.00	0.70
	Saithe	106,000t	200,000t	0.60	0.40
	Whiting	225,000t	315,000t	0.90	0.65
	Plaice	160,000t	230,000t	0.74	0.60
	Sole	25,000t	35,000t	*0.55*	0.40
	Norway pout	90,000t	150,000t	*0.85*	*0.65*
	Average	**110,857**	**174,286**	**0.786**	**0.579**
Pelagic	Herring	800,000t	1,300,000t	Undefined	0.25
	Sprat	undefined	undefined	Undefined	undefined
	Sandeel	430,000t	600,000t	Undefined	undefined

Time series of fishing mortality (F) and spawning stock biomass (SSB) derived from single species stock assessments of seven demersal stocks (cod, haddock, whiting, saithe, plaice, sole, and Norway pout) were analysed (ICES, 2007). Data were only complete for all seven species from 1983 onwards. Weighted average "demersal community" F (see Section 6.2.1) and SSB were computed for each year on these data (Figure 6.4.2.1). The short duration of the time series was clearly an issue. The plot for SSB shows an immediate decline from the highest value in the time series. Similarly F during the first quarter of the time series was higher than in any subsequent period. Both plots emphasise the desirability of a longer time series that may reveal the trajectories for both variables in the lead up to these high levels.

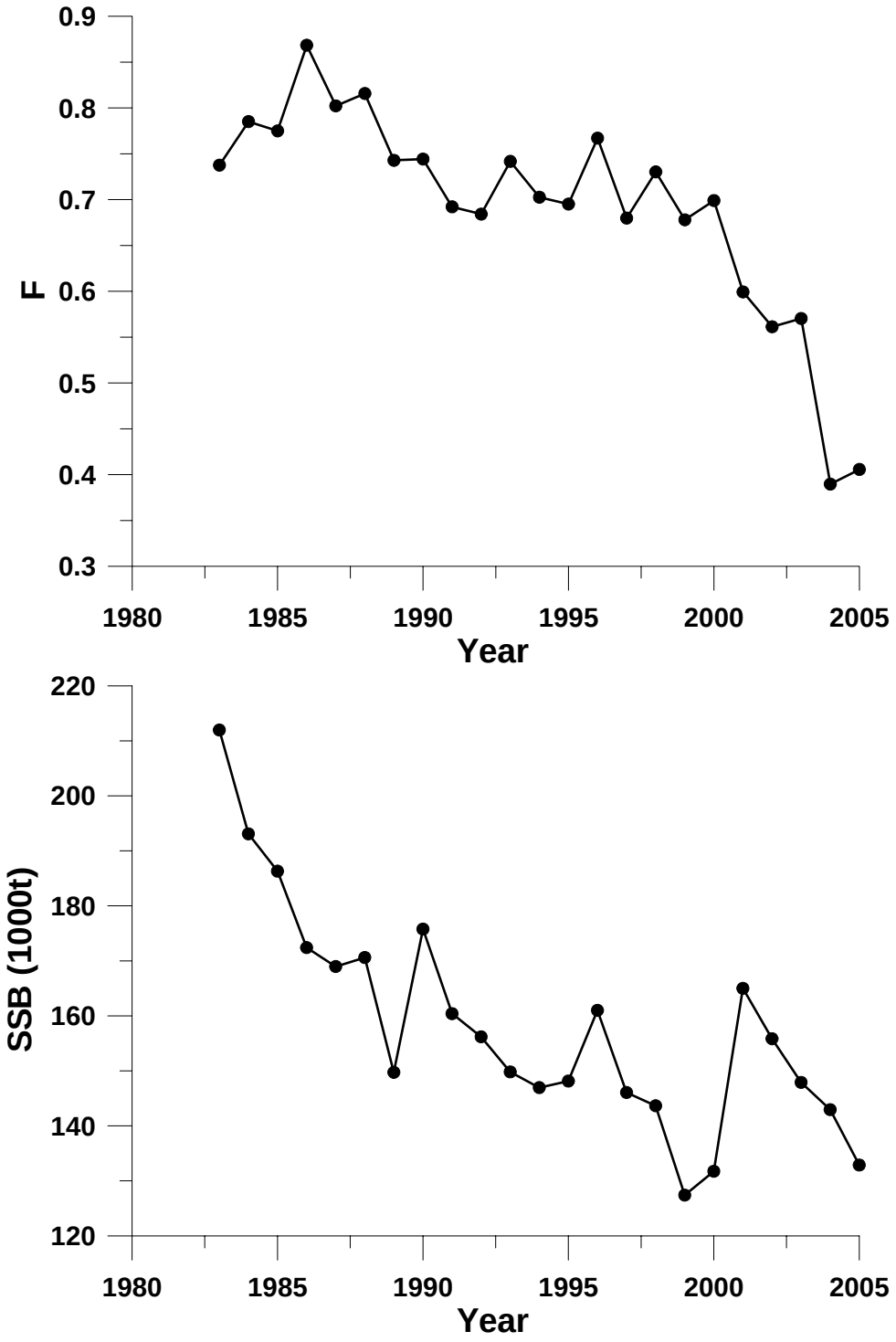


Figure 6.4.2.1. Trends in weighted average fishing mortality (F) and spawning stock biomass (SSS) calculated for seven stocks based on data provided by the single species stock assessments (ICES 2007).

The last run of the MSVPA model (ICES 2006) provides both SSB and F at age data for each of the seven demersal species, as well as three pelagic species (herring, sprats and sandeels), for the period 1963 to 2003. The same approach was adopted for these data, using average F across ages 2 to 5 for each species, which provided weighted community average F values that were in closest agreement with the values derived from the single species stock assessments (Figure 6.4.2.2). Demersal community average SSB derived from the MSVPA data were also closely correlated with the equivalent values derived from the single species stock assessments

(Figure 6.4.2.2). The longer time series provided by the MSVPA data captures the full range in variation in both weighted average F and SSB in the demersal community, and likewise for the pelagic community (Figure 6.4.2.3).

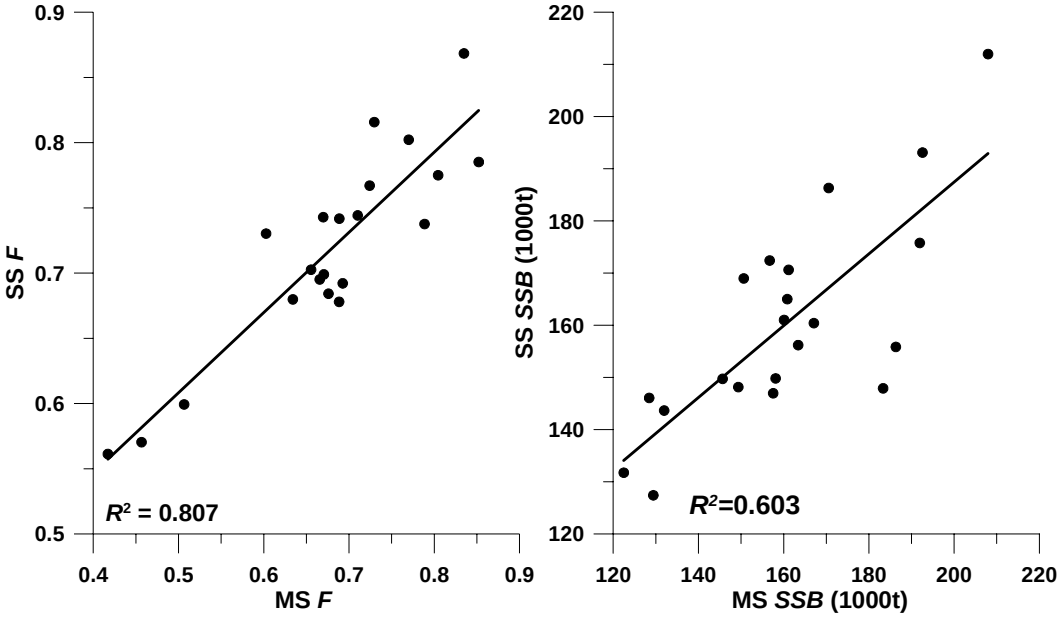


Figure 6.4.2.2. Relationships between estimates of weighted average "demersal community" F and SSB derived from the single species stock assessments and the multi species VPA model.

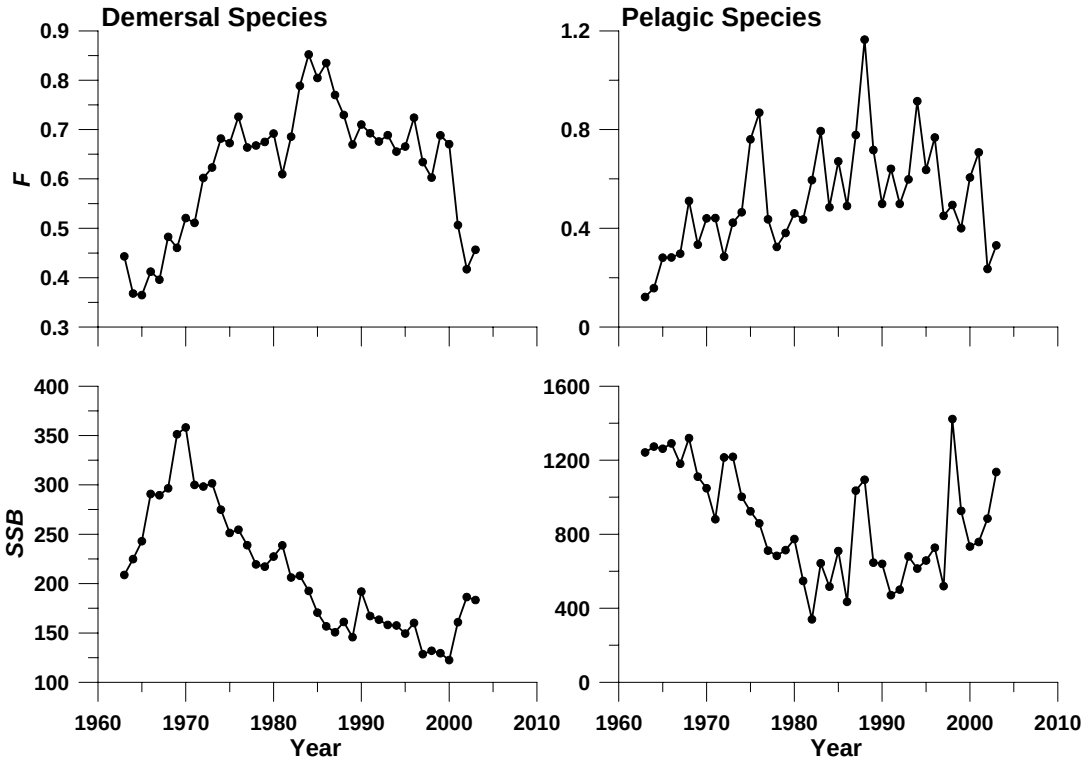


Figure 6.4.2.3. Trends in weighted average fishing mortality (F) and spawning stock biomass (SSS) calculated for seven stocks based on data provided by the multi-species VPA model (ICES 2006).

These data allow the construction of "precautionary plots" similar to those presented by ICES (2006) for single species, F is plotted on the x axis and SSB on the Y axis and the four precautionary reference points, F_{lim}, F_{pa}, B_{lim} and B_{pa}, are indicated. To do this for the MSVPA derived data, we first have to determine MSVPA equivalents for these four parameters. To do this, F and SSB derived from the SS stock assessments was regressed on F and SSB derived from the MSVPA (Figures 6.4.2.4 and 6.4.2.5 respectively), and the regression equations derived used to estimate equivalent MSVPA parameter values for each species (Table 6.4.2.2), given the SS stock assessment values shown in (Table 6.4.2.1). Sole F_{lim} and Norway pout F_{lim} and F_{pa} are not defined in ICES (2006). Sole F_{lim} was estimated by multiplying F_{pa} by 1.374, the average difference for the five species with both values defined, giving a value of 0.55 (Table 6.4.2.1). Norway pout SSB and F varied in a similar manner to whiting and haddock over much of the single species time series. So Norway pout was arbitrarily assigned F_{lim} and F_{pa} values of 0.85 and 0.65 respectively (Table 6.4.2.1). For both sets of data, single species assessment and MSVPA, demersal community average reference points can then be calculated (Tables 6.4.2.1 and 6.4.2.2 respectively).

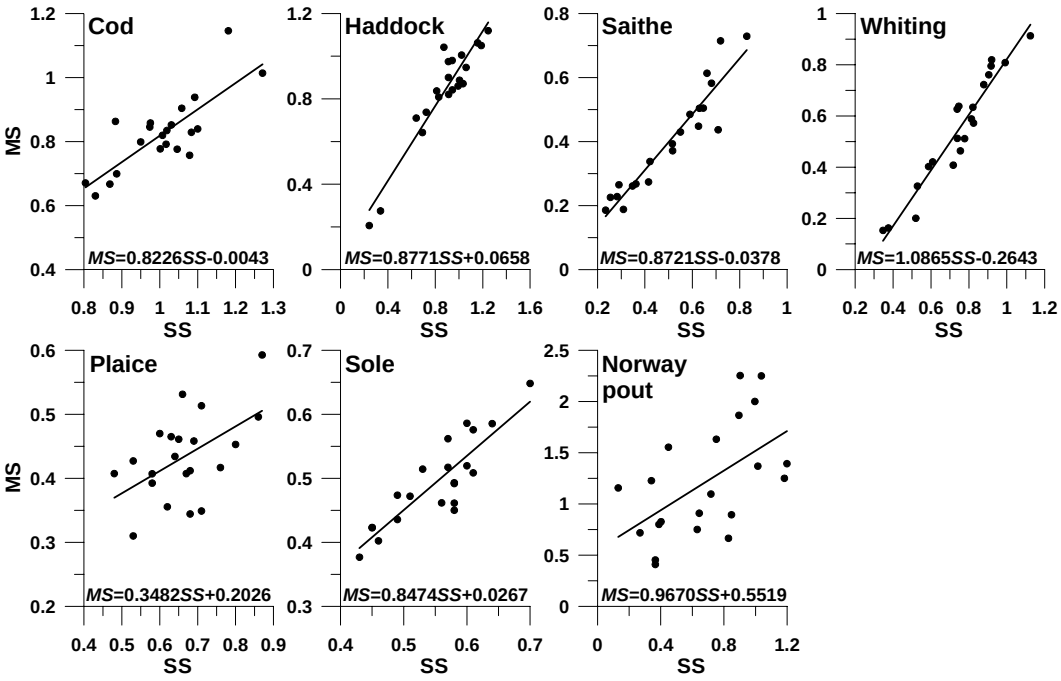


Figure 6.4.2.4. Relationships between F derived from MSVPA and SS stock assessments for each of the seven demersal stocks. Note that a 1:1 relationship between the two are not to be expected as the methods and tuning data may differ between MSVPA and SS stock assessments.

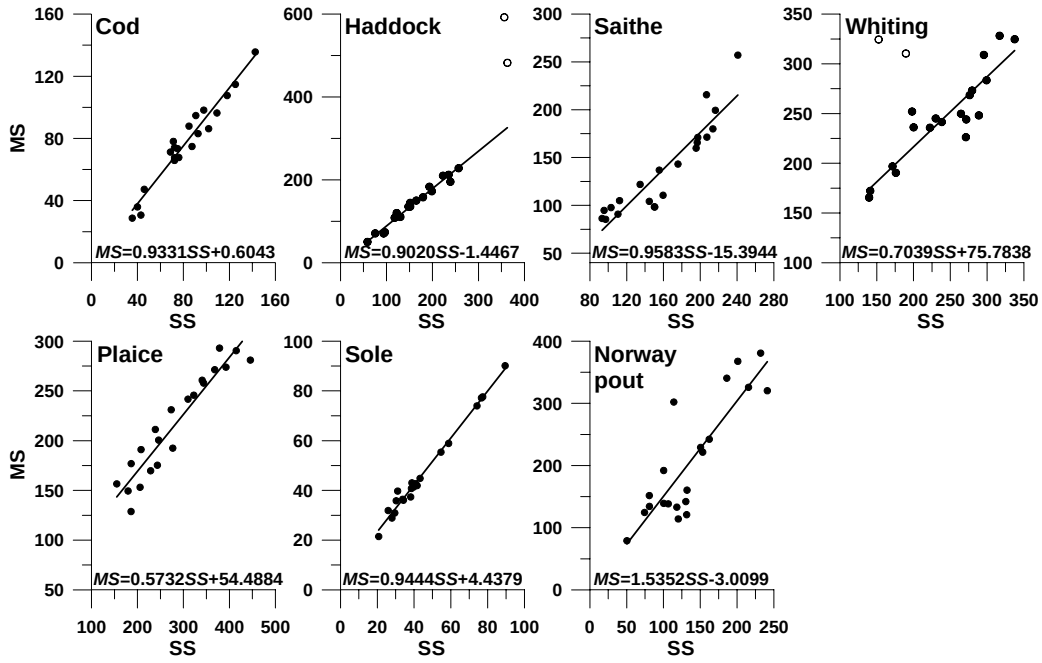


Figure 6.4.2.5. Relationships between SSB (1000t) derived from MSVPA and SS stock assessments for each of the seven demersal stocks. Note that a 1:1 relationship between the two are not to be expected as the methods and tuning data may differ between MSVPA and SS stock assessments. The open points were considered outliers and not included for the calculation of the regression.

Table 6.4.2.2. Equivalent MSVPA precautionary reference points for seven demersal stocks derived using the regression equations shown in Figures 4 and 5 and the SS stock assessment reference points given in Table 6.3.2.1.

ASSEMBLAGE	SPECIES	B_{LIM}	B_{PA}	F_{LIM}	F_{PA}
Demersal	Cod	65921	140569	0.703	0.530
	Haddock	88753	124833	0.943	0.680
	Saithe	86185	176266	0.485	0.311
	Whiting	234161	297512	0.714	0.442
	Plaice	146200	186324	0.460	0.411
	Sole	28048	37492	0.493	0.366
	Norway pout	135158	227270	1.374	1.180
	Average	**112061**	**170038**	**0.739**	**0.560**

Figure 6.4.2.6 shows the precautionary plot based on the MSVPA time series. This clearly indicates that the community was pushed out of the "desirable state" in the early 1970s as fishing mortality increased above F_{pa}. Through the remainder of the 1970s F remained above F_{pa} and gradually SSB declined. In 1983 F increased above F_{lim} and in 1986 SSB consequently dropped below B_{pa}. From 1986 onwards F declined so that in 1988 it was once again below F_{lim} and in 2001 dropped below F_{pa}. During this period however, there was no recovery in SSB and SSB remained below B_{pa}. In 2002 and 2003 SSB once again appeared to climb back above B_{pa} so that the community index was again situated in the "desirable" zone.

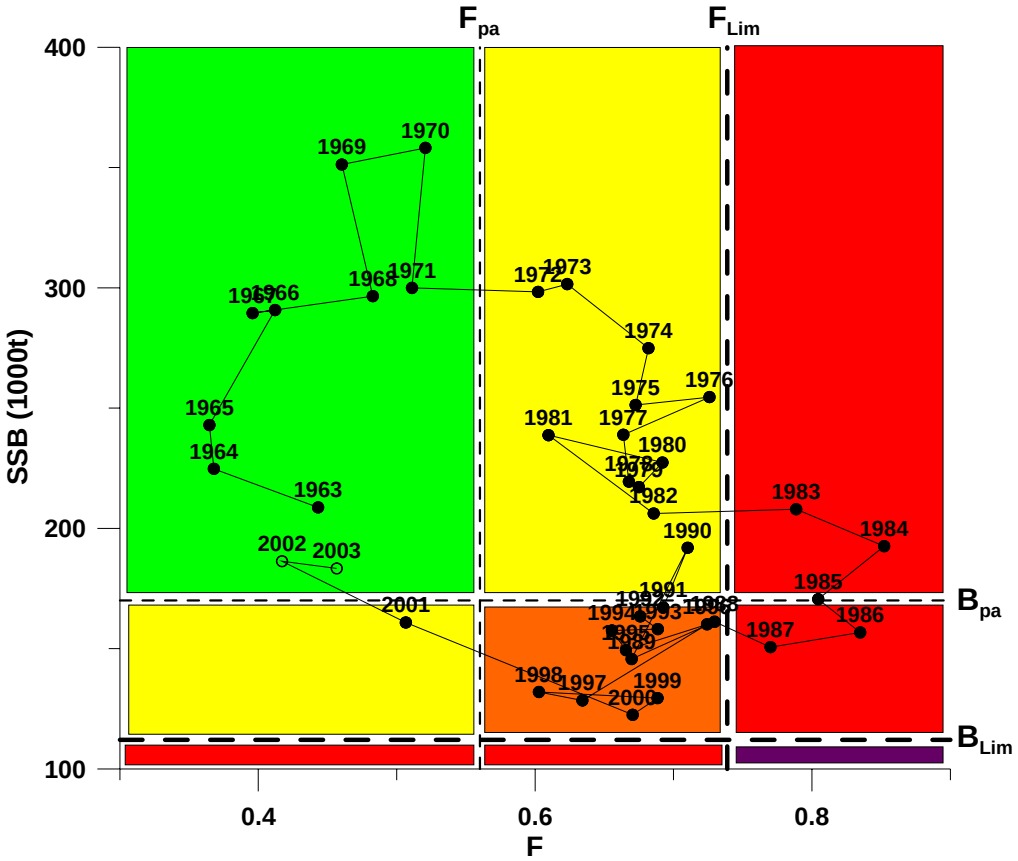


Figure 6.4.2.6. Precautionary plot for the seven commercially exploited demersal fish "community" based on MSVPA output to be used as a method to estimate a reference period for setting targets for fish community indicators. Unfilled circles (2002 and 2003) represent uncertain data points (see text).

The precautionary plot derived from the single species stock assessments data tells an almost identical story (Figure 6.4.2.7). However, the earlier part of the sequence is missing and the plots starts with the community in the zone indicating the need for remedial action. The major difference between the two data sets is that, unlike the MSVPA plot, the SS plot does not indicate any recovery in SSB in the early 2000s. Despite the marked reduction in F, the community average SSB remains well below B_{pa}. Figure 6.4.2.5, illustrating the relationship between MSVPA derived and SS assessment derived SSB, shows that for both haddock and whiting two outlier data points were excluded from the regression analyses. For both species, these were the 2002 and 2003 data points and it is clear that the MSVPA has over-estimated SSB relative to the SS assessments (comparing Figures 6.4.2.5 and 6.4.2.6). The fact that SSB remains low in 2003 and 2004 in the SS assessment time series also strongly indicates that error in the MSVPA data set is the reason for this discrepancy. So despite the steady reduction in community F since 1986, SSB has failed to recover above the B_{pa} reference point.

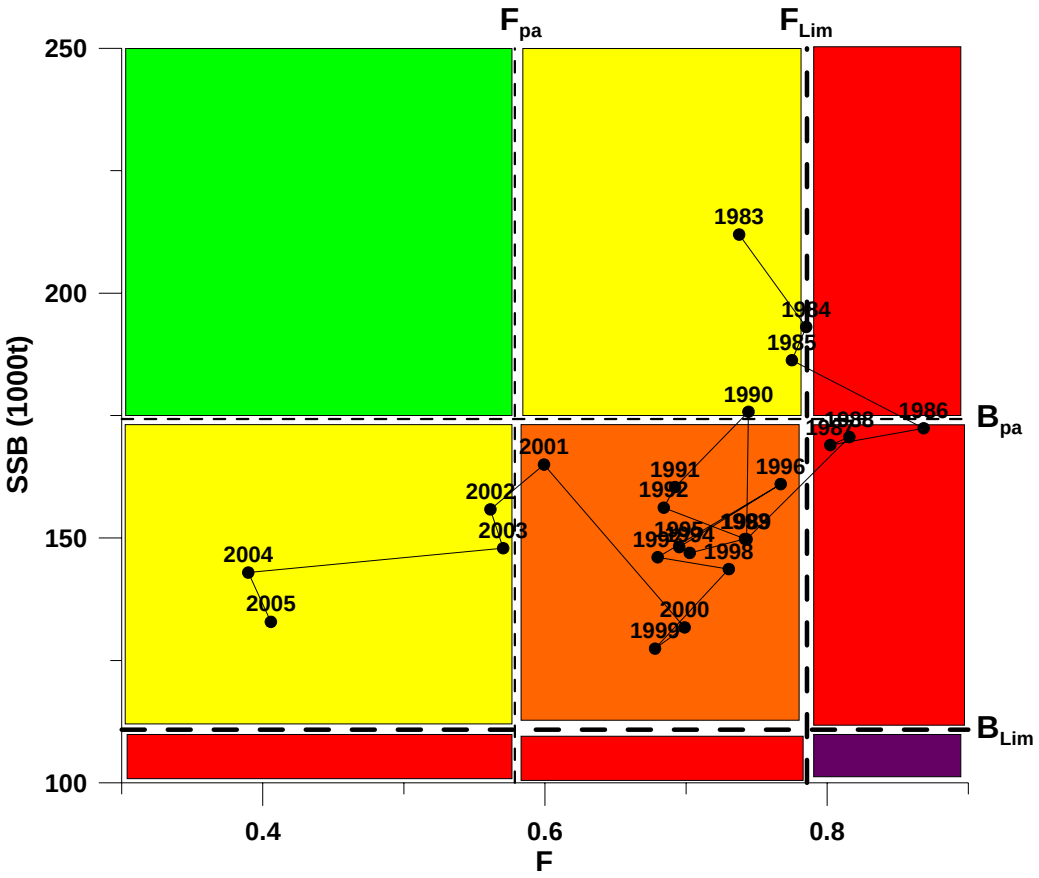


Figure 6.4.2.7. Precautionary plot for the seven commercially exploited demersal fish "community" based on single species stock assessments to be used as a method to estimate a reference period for setting targets for fish community indicators.

6.4.3 Conclusions regarding the target period for ecosystem indicators

The three methods (proportion of stocks above B_{pa} and the single and multispecies precautionary plots) all indicate that the period after 1984 is not an acceptable target period. It is unclear which period prior to 1984 is acceptable, as the demersal precautionary community plot will have been heavily influenced by the 'gadoid outburst' and thus may be an inappropriate target period. If a target period is consistent with the ICES single species F versus B plots, it should be a period where SSB was above B_{pa} and F was below F_{pa} with the caveat that these multispecies F versus B plots have not been sufficiently peer-reviewed or accepted by ICES.

6.5 Using ecosystem models to estimate future trajectories of ecosystem indicators in the North Sea under different fishing scenarios

Determining future trajectories in mean weight and the proportion of large fish refers to TOR g which also requests estimates of the time to recover reference levels in the early 1980s. These requests were discussed in detail by the working group and three subjects which needed further consideration were determined:

1) The ability of the proposed ecosystem indicators to reflect fishing pressure
2) Projections of future developments in the North Sea using MSVPA
3) The effect of climate changes on the North Sea ecosystem

These three points are discussed separately in the following.

6.5.1 The ability of the proposed ecosystem indicators to reflect fishing pressure

Unfortunately, several analyses demonstrated the poor relationship between the indicators mean weight and proportion of fish larger than 30 cm and fishing pressure both in the North Sea and other areas (Sections 6.1 and 6.2). The working group does therefore not recommend proceeding further with these indicators. Instead, the group recommends using the established indicators referring to biomass and fishing mortality of assessed stocks (B_{lim}, B_{pa}, F_{lim} and F_{pa}) to estimate the proportion of stocks which are within safe biologic limits and define specific indicators for non-assessed stocks. In particular, indicators on sensitive and declining species such as the composite threat indicator suggested by Dulvy *et al.* (2006) should be investigated further to examine whether they provide consistent results when applied to non-assessed species only.

6.5.2 Projections: caveats for present work and recommendations for future work.

The current version of the MSVPA (ICES 2006b) is unstable as species such as cod tend to die out because of an excessive predation pressure. Running the model in forecast mode is therefore no simple task, and all predictions have to be thoroughly evaluated before they are accepted as likely scenarios. The group had neither the expertise to run the model nor to evaluate the predictions; therefore, single species projections (Section 6.3) were performed to give some idea of the nature of results one might expect in a proper multispecies projection.

The single species projections (Section 6.3) must be regarded with caution. First, as these are stock assessment results, they should actually be done by a stock assessment working group. The obvious conceptual problem that these are single species results where species have no dynamic bearing on one another make the whole analysis questionable in the multispecies and community context upon which these community indicators are based. Technically, many non peer reviewed assumptions were made in order to run them and there are no guarantees that they conform to the standard ICES sets for these kinds analyses. We therefore recommend that if this approach is to be taken seriously in a management context such that hard recovery targets and times are required, it is essential that the expertise in multispecies modelling and assessment is available. This work could be addressed specifically to the working group on multispecies methods (WGSAM). Other ideas could include a joint workshop with assessment scientists, MSVPA scientists and ecologists attend, or that clear ToRs are passed between the appropriate working groups and joint attendance between selected members ensured. Finally, we recommend that community projection for any kind of community indicator is also conducted using size-spectra models and other community models such as OSMOSE and ECOSIM. Projections with community models can be conducted by WGFE members but unfortunately expertise in these methods was not available for the 2007 meeting.

6.5.3 The effect of climate changes on the North Sea fish community

Several stocks appear to be influenced by temperature during the larval stage (Svendsen et al 1995). As even the most optimistic views of future climatic changes foresees considerable decreases in recruitment success of e.g. cod (Kell *et al.*, 2005), recruitment success is unlikely to remain at historic levels for all species. Furthermore, climatic changes have led to changes in the spatial distribution of a number of species (Perry *et al.*, 2005) and thus changes in overlap with potential prey and predators. The MSVPA model is currently under development in the EU-project 'BECAUSE' to assure that known relationships between fish distribution and hence predation mortality and recruitment success and climate can be included when performing projections. Ignoring these effects on recruitment renders forecasts unreliable.

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Annex 2: Agenda

Work days begin 9:00 at IFREMER

Coffee daily 10:30 and 15:30 (supplied by IFREMER)

Lunch in IFREMER cafeteria (12:45 – 13:30)

Day ends ~20h00

Monday, 5 March 2007:

Introduction to the group and its work

Terms of Reference for 2007

Personal introductions

Presentations

Tuesday, 6 March 2007:

Presentations until lunch time

Work in groups

Wednesday, 7 March 2007:

Work in groups

Thursday, 8 March 2007:

Work in groups

Plenary presentation of group work

Friday, 9 March 2007:

Work in groups

Plenary presentation of group work

Annex 3: WGFE Terms of Reference for the next meeting

The **Working Group on Fish Ecology** [WGFE] (Chair: D. Duplisea, Canada) will meet in Copenhagen from 3–7 March 2008 to:

a) EcoQOs for threatened and declining fish species;

b) reconsider the relationships between EFH, abundance-occupancy and changing fish distribution in relation to climate and fishing pressure:

  i) define potential fish habitats.

  ii) examine processes underlying expansion and contraction of fish spatial distributions using case studies.

  iii) test the ability of methods to uncover relationships in abundance-occupancy.

  iv) examine the relation between spatial organisation and community and population metrics.

c) evaluate metrics to characterise, monitor and detect changes in the structure, function and productivity of fish communities;

d) evaluate methods to detect changes in fish distributions using a large set of maps.

WGFE will report by 30 April 2008 to the attention of the Living Resources Committee.

Supporting Information

PRIORITY:	
SCIENTIFIC JUSTIFICATION AND RELATION TO ACTION PLAN:	The development of EcoQOs for fish communities and threatened and declining fish species are required by OSPAR. This work supports Action Points 2.2 and 3.2. Essential fish habitat studies have implications to management issues and will also aid in the interpretation of abundance-occupancy relationships. EFH work particularly supports Action Points 1.2.1, 1.2.2 and 1.4.2. Fish distribution mapping studies in relation to environmental variables relates to action point 1.6
RESOURCE REQUIREMENTS:	
PARTICIPANTS:	The Group is normally attended by 15–20 members and guests.
SECRETARIAT FACILITIES:	None.
FINANCIAL:	No financial implications.
LINKAGES TO ADVISORY COMMITTEES:	
LINKAGES TO OTHER COMMITTEES OR GROUPS:	The development of EcoQO indicators is work that should be conducted in conjunction with WGEF (Elasmobranch Fishes). Work on simulation in fish communities for the testing of EcoQOs is closely related to the development of multispecies modelling in WGSAM. The work of this group is an important information source for WGECO (Ecosystem Effects of Fishing)
LINKAGES TO OTHER ORGANIZATIONS:	

Annex 4: Recommendations

RECOMMENDATION	ACTION
1 Projections of community indicators using multispecies models must be conducted by appropriate experts in the field	Coordinate WGFE and WGSAM meetings **or** pass specific terms of reference between these two groups **or** organize a separate workshop by person or by correspondence on this specific task.