Report of the Working Group on Fisheries Ecology

Andrey Dolgov, Jim Elkis, Micholas Dulvy, and Geir Huse

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Knipovich Polar Research Institute of Marine Fisheries and Oceanography
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Simon Fraser University
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Institute of Marine Research in Norway
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International Council for the Exploration of the Sea
Conseil International pour l'Exploration de la Mer

International Council for the Exploration of the Sea
Conseil International pour l'Exploration de la Mer
H. C. Andersens Boulevard 44–46
DK-1553 Copenhagen V
Denmark
Telephone (+45) 33 38 67 00
Telefax (+45) 33 93 42 15
www.ices.dk
info@ices.dk

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- Compatibility between IUCN decline criteria and ICES reference points was re-examined and it was concluded that for the most part they are compatible, i.e. it is unlikely that threatened or endangered status would be assigned to a species whose biomass was larger than the safe biological limits.
- A first attempt to model recovery of the North Sea EcoQO was made with an ecosystem model suggesting that recovery on the time frame of a decade was realistic.
- Methods for comparing maps of species distributions were explicated and a new method for multidimensional map comparison was proposed. Methods like this could be used as a standard for common TORs assigned to several working groups (e.g. OSPAR climate request) in order to produce compatible output for a unified ICES response.

Executive Summary

In 2008 WGFE finalised its contribution to the OSPAR request to examine impacts of climate induced changes in the marine physical environment on the distribution and abundance of fish. The analyses the 2008 report are more succinct than in the 2007 report and many of the suggestions of WGECO were heeded. Analysis were broken down by species and OSPAR region and where direct data analysis could not be done by the group, literature was summarised and in particular the report of the EU funded project RECLAIM was integral for filling in gaps. Many demersal and pelagic species changed abundance and distribution in all areas and while some of these changes can be attributed to change in temperature between time periods, others cannot. It is also apparent that warming is some cases has meant that species once considered strays are become much more common. In other cases the warming has improved recruitment for some species thus creating a shift in the apparent range of the species though not necessarily a change in individual movement. Overall, many species have shown positive changes in abundance in most OSPAR areas. Southern species have tended to increase over all areas while northern species have tended to recede from the south and heavily exploited species such as cod have decreased in abundance everywhere.

WGFE revisited work started in 2004 (see the 2004 WGFE report) examining the compatibility between ICES reference points for exploited stocks and IUCN decline criteria. On the whole it was concluded that it is unlikely that an IUCN list threshold would be breached for a declining stock before an ICES PA or limit reference point would be breached. The potential difficulties for managers is obvious if such a situation presented itself, i.e. they would be compelled to protect a stock under conservation legislation while at the same time the stock would be deemed sufficiently abundant to exploit commercially. This appears unlikely to occur especially if ICES manages stock to remain above Bpa as a stock’s biomass dropping below Bpa should trigger a sufficiently strong fishery management response to reverse the decline well before any IUCN decline criterion would take effect.

Recovery of the North Sea demersal fish community to the EcoQO (fish community target state indicative of health) prescribed by WGECO was tested this year under a range of community fishing scenarios using a mass balance size-based model. This model suggested that the fish community would react to changes in fishing pressure quickly and achieve a new steady state on the order of 5-10 years. Further simulation work is required to determine the necessary change in fishing mortality required to achieve the particular EcoQO target. To this end, the potential modelling methods that may aid in this kind of projection were reviewed. Some models appear more suitable than others for exercises of this nature and in the coming years it is apparent that ICES will acquire the appropriate suite of tools and expertise to provide useful advice on fish community exploitation for achieving EcoQO targets.

Essential fish habitat and abundance-occupancy were addressed through three case studies. The first study examined the relationship between occupancy and abundance for several species in the Bay of Biscay and Celtic Sea accounting for habitat characteristics. Though large changes in total abundance were found, there were no changes in occupancy that transcended habitat variables. A second study examined how trawling affects fish abundance, habitat and subsequently species composition and habitat utilisation. Exploited fish species initially responded to the respite in fishery exclusion and over a longer period their productivity also increased in the non-trawled area which seemed more related to the curtailment of destructive fishing practices. A final study of inter- and intra-specific abundance-occupancy relationships in the Georges Bank fish community showed that most species did contract their occupancy with a decline in abundance. When this relationship was examined over species but within years, it was apparent that the statistical quality of the relationship decreased over time from the 1960s to present and this has been interpreted a result of habitat loss and fragmentation over time which interferes with how fish use space on Georges Bank.

WGFE has traditionally included studies proposing new indicators for characterising fish communities and testing how they respond to fishing, environment and their relation to other measure of fish community state. One study is presented here which examines redundancy between common community indicators for the a North Sea surveyed community such as diversity indices, total biomass, mean body size and mean largest body size. While some indicators were redundant others were not. Overall, if it is accepted that the indicators are meaningful for showing characteristic of fish communities, than it would take a suite of indicators to appropriate characterise the community.

In 2007 WGFE constructed many maps of species distributions in relation to environmental variables in order to meet the demands of the OSPAR term of reference on species responses to climate change. Though the maps were helpful for presenting large amounts of information it was realised that there were few analyses that dealt with quantitative comparisons of maps or differencing them to find changes over time. WGFE in 2007 therefore proposed future work that would deal with comparisons between maps usually between time periods. The two studies are presented here: (1) a method to dimensionally reduce multivariate maps to a 3 variable representation and compare them over time; and (2) differencing maps (using three different methods) between periods in order to highlight changes in species distributions between periods. WGFE anticipates further work on these methods at future meetings.

1 Introduction

1.1 Terms of reference for 2008

a) consider the reports of the Ad Hoc Groups on;

Hydrographic Attributes

Trend Analyses & Quantifying Relationships

Formulating Hypotheses and Predictions about Mechanisms

Selecting Species for More Intensive Investigations

and use their recommendations concerning (1) recommended time series, (2) analytical methods and suitable software, (3) hypotheses and guidance for their use, and (4) a suggested list of species for intensive study, to complete 'the assessment of changes in the distribution and abundance of marine species in the OSPAR maritime area in relation to changes in hydrodynamics and sea temperature.

b) revisit and update the 2004 work of WGFE on the IUCN red listing criteria for marine fish in order to answer a request to:

> review the IUCN criteria and evaluate suitability and usefulness of using these on marine fish species. Compare the criteria with the ICES series of reference points.
>
> evaluate whether the risk assessment criteria used by IUCN are suitable for marine fish species and, if necessary, other and more appropriate methods and procedures.
>
> discuss and suggest procedures within ICES to gather useful biological information about non-commercial species that may improve the red list evaluation of these species.

c) assess the various modelling approaches, which WGECO (ICES, 2007) noted were in development, to determine which, if any, of these models might also be suitable to address this advice issue, and to this end apply the MSVPA in forecast mode (MSFOR) to complete the matrix below;

Fishing mortality averaged across the seven main demersal species (cod, haddock, whiting, saithe, sole, plaice, Norway pout)	EcoQO Indicator Proportion (by weight) of fish > 40cm in length	Time to reach Indicator target
0.85Fpa	0.27	?
	0.30	?
	0.33	?
1.00Fpa	0.27	?
	0.30	?
	0.33	?
1.15Fpa	0.27	?
	0.30	?
	0.33	?

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

define potential fish habitats

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

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

examine the relation between spatial organisation and community and population metrics

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

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

1.2 Participants

Ken Haste Andersen (AQUA, Denmark)
Tom Blasdale (JNCC, Scotland)
Nick Dulvy (SFU, Canada)
Daniel Duplisea (DFO, Canada)
Jim Ellis (CEFAS, England)
Mike Frisk (SBU, USA)
Jakob Gjøsæter (IMR, Norway)
Simon P.R. Greenstreet (FRS, Scotland)
Emma Guirey (FRS, Scotland)
Ralf van Hal (IMARES, Netherlands)
Remment ter Hofstede (IMARES, Netherlands)
Dave Kulka (DFO, Canada)
Pierre Petitgas (IFREMER, France) (via correspondence, Section 7)
Anne Sell (ISF, Germany)
Alberto Serrano (IEO, Spain)
Verena Trenkel (IFREMER, France)

Full contact details of participants are provided in Annex 1.

1.3 Background

The Working Group on Fish Ecology (WGFE) 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. WGFE has met annually since 2003 always before the Working Group on the Ecosystem Effects of Fishing Activities (WGECO) for which WGFE's report is anticipated for that group to complete its work. WGECO has traditionally commented on the report of WGFE and asked specific pieces of analysis to be performed at its next meeting thus providing a clear linkage between the more pure science products of WGFE and the need for advice to inform policy considered by WGECO.

WGFE has addressed issues on non-commercial fish species, including species of conservation importance, fish communities and assemblages, impacts of climate change on fish communities and other aspects of fish ecology (e.g. feeding habits and prey rations, habitat requirements), to support ICES advice in areas related to ecosystems, biodiversity, conservation and climate change. In addition to supporting scientific advice, WGFE is a group that can incorporate and germinate scientific ideas related to methodological development and ecological synthesis; thus, WGFE has become a point of initiation into ICES for new researchers and researchers who have not before been involved in practical applications of their science.

1.4 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 abundance and distribution of selected fish species in response to environmental variables in the OSPAR marine area

ToR - a) consider the reports of the Ad Hoc Groups on;

i) Hydrographic Attributes
ii) Trend Analyses & Quantifying Relationships
iii) Formulating Hypotheses and Predictions about Mechanisms
iv) Selecting Species for More Intensive Investigations

and use their recommendations concerning (1) recommended time series, (2) analytical methods and suitable software, (3) hypotheses and guidance for their use, and (4) a suggested list of species for intensive study, to complete 'the assessment of changes in the distribution and abundance of marine species in the OSPAR maritime area in relation to changes in hydrodynamics and sea temperature.

2.1 Consideration of the reports of Ad Hoc groups

WGFE considered the reports of the Ad Hoc groups mentioned in the ToR: i) Hydrographic attributes and iii) Formulating Hypotheses and Predictions about Mechanisms. Two groups, on ii) Trend Analyses & Quantifying relationships and iv) Selecting Species for More Intensive Investigation, did not meet before the WGFE meeting.

The Working group on Ocean Hydrography (WGOH, ICES 2008b) suggested a set of time series on hydrodynamic attributes and temperature. Sub-surface temperature and salinity time series are provided for the five OSPAR sub-regions, from single locations within each region, and have been selected for their suitability to show the integrated effects of large-scale patterns within the area. WGFE conducted extensive analyses last year in which different temperature time series were used, namely sea surface temperature (SST) and bottom temperatures over the entire North Sea, instead of from a single location. Since WGFE is elaborating this year on its previous spatial analyses using the North Sea as the key dataset, it was decided to continue those analyses with the spatial data that were already used. However, the series provided by WGOH is considered in later sections with which to interpret results.

The work done by the Study Group on Working Hypotheses Regarding Effects of Climate Change (SGWRECC, ICES 2008a) was based mainly on two extended scientific reviews, one carried out by the UK Marine Climate Change Impacts Programme (MCCIP) and one by the EU-funded project RECLAIM (see Sect. 2.8). Three hypotheses were proposed by SGWRECC which focus on fish (SGWRECC report 2.1, 2.4, and 2.5). All three hypotheses are worth investigating further.

As two of the ad hoc groups did not meet and given the lack of response to WGECO's (ICES, 2007b) recommendations, WGFE determined its own course in dealing with this TOR again in 2008. WGFE recommends for future work along these lines, that ICES provides a clear data acquisition and quality control strategy as well as some relatively explicit guidance for analysis, before the WG begins its work. Such information would aid this group and others to generate scientific products allowing ICES to compile a standardised product for OSPAR.

2.2 Changes in distribution and abundance of fish in the OSPAR maritime area

Last year extensive analyses were carried out by WGFE to assess changes in distribution and abundance of fish species. WGFE encountered difficulties in separating changes in abundance and distribution. A change in distribution can occur as a shift in one of the boundaries of the area occupied by the species, but it could also occur as spatial change in the "centre of density". In the first case the change in distribution might be a change in the geographical range of the area occupied by the species. In the second case distribution and abundance are related. In ToR d), an analysis is shown where for some species a relationship is found between an increase in abundance and the area occupied by the species (see Sect. 5 of this report).

The work of WGFE in 2007 has been reviewed by the Working Group on Ecosystem effects of fishing activities (WGECO, ICES 2007b), and detailed response is given in Sect. 2.6. The main commentary of WGECO was that the results of the analysis were not clearly separated into changes in distribution and changes in abundance and that analyses were limited mainly to the North Sea; therefore, the focus of WGFE this year was to separate and clarify the results in order to meet the framework suggested by WGECO. The criticism that the analysis was mainly in the North Sea could still not be fully rectified this year. WGFE was restricted in its possibilities to extend the analyses due to limitations in both time and availability of data for certain regions (OSPAR regions I lacks largely, IV the south and V lacking completely). WGFE dealt with other OSPAR regions primarily by drawing upon the RECLAIM and SGWRECC reports and relevant literature.

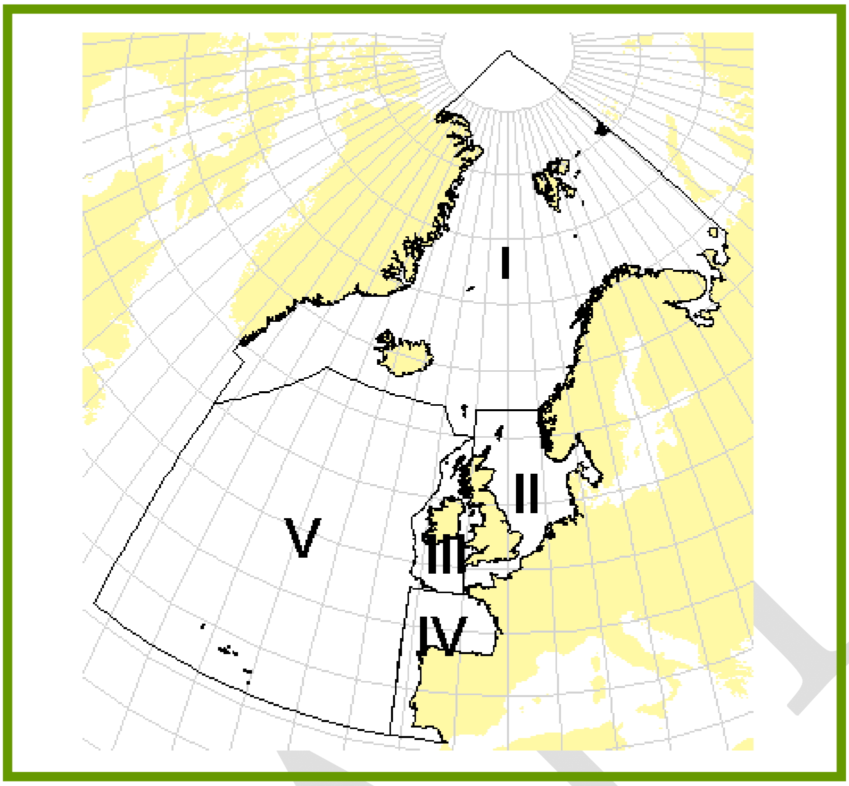


Figure 2.2.1: Overview of the five OSPAR regions: Region I - The Arctic, Region II - The Greater North Sea, Region III - The Celtic Seas, Region IV - The Bay of Biscay, Region V - The Wider Atlantic.

Overall changes in fish distribution by OSPAR regions

Rose (2005) compiled a literature review and analysed factors that might influence the distribution of fish species in the North Atlantic. Those analyses indicate that the limits of both spawning and feeding distributions are influenced by environmental factors that are partially controlled by climate variability, and likely, by climate change. Not all species may be impacted in the same manner and changes depending on ecological associations. Species who's spawning is limited to the cold waters of the arctic or boreal regions are expected to be most affected (e.g. the small pelagic capelin), and to respond relatively quickly to environmental change. Shelf species in general and cold-water species such as winter flounder, several species of sculpin, eelpouts, Arctic cod, the gadoids and several flatfish and other boreal species are mid-range in vulnerability and might directly benefit from warmer conditions though indirect effects (e.g. declines in abundance of their main prey) may have negative effects.

Corten and Van de Kamp (1996) showed that southern fish species increased in the North Sea during years of increased Atlantic inflow through the Dover Straits driven by increased southerly winds (and higher temperatures). These changes characterised temporal extensions of the species range rather than long-term colonisations. However further research indicated that it could have been long-term colonisation caused by increased temperatures, as is particularly noticeable in the case of sardine, anchovy, red mullet (the latter, two of the species in the present spatial analysis), lesser weever and bluemouth (Beare *et al.* 2004; 2005). The pattern of an apparent northward shift in the distribution of fish species from southerly waters shown in OSPAR region I (Bjoersnsson & Palsson 2004; Astthorsson and Palsson 2006;

Astthorsson *et al.*, 2007), region II (Perry *et al.* 2005; Beare *et al.* 2004; 2005) and regions III (Stebbing *et al.* 2002) and IV (Quero *et al.* 1998) have coincided with recent warming trends in the Northeast Atlantic.

Species habitat occupancy, latitudinal and depth distributions appear to be changing in response to interannual variation in several measures of temperature and/or hydrography. However, there is no single biogeographical measure that consistently responds to a single measure of temperature or hydrography across the range of species. There is considerable heterogeneity in individual species' response to the various measures of climate variability, although there is scope to determine the underlying ecological factors, such as niche (pelagic/demersal), trophic level and particularly body size. Comparative studies highlight a substantial proportion of species that do not appear to change distribution in response to climate variability (within the range of variability observed over the last fifty years). Finally, as fish communities are simultaneously exposed to several factors, particularly to changes in temperature and to fishing pressure, synergistic effects as well as counteracting processes need to be addressed and investigated further.

The analysis presented last year (ICES, 2007a) seems typical of multispecies climate-biological response analyses - many species showing heterogeneous responses, with the consequence that it can be difficult to uncover general patterns. Perry *et al.* (2005) found it difficult to define a single relationship between life histories and distributional response and based their conclusion on a categorical test (large versus small) rather than treating body size as a continuous variable. It may be that the variance in individual species trends confounds efforts to uncover a general pattern. Accordingly, the updated analysis presented here follows the categorical treatment of traits approach used by Perry *et al.* (2005). The purpose of this work has changed somewhat from looking at individual species sensitivity to climate variables with the recent focus changing toward developing an ecosystem-scale indicator of the biotic response of the aggregate demersal fish assemblage to climate variability and longer-term climate change (Dulvy *et al.* in press, Section 2.7).

The changes discussed above cannot be related to the changes in the hydrodynamic drivers based solely on statistical analysis. For example, fishing mortality rates have been higher in the southern North Sea than in the north (Heath *et al.* 2003, Heath *et al.* 2007), and so the apparent changes in distribution in this area could be a consequence of local patterns of fishing pressure (Hutchinson *et al.* 2001, Wright *et al.* 2006, Daan *et al.* 2005). The effects of fishing thus interact with the effect of climate. The disentanglement of the effects of hydrographic attributes and other drivers is difficult and must be considered as a work in progress. The reliability of the results attributed to hydrographic changes shown above and in the rest of the chapter are under debate by WGFE and in most cases no definite answers can be given.

OSPAR Region I (Arctic waters)

OSPAR Region I or the Arctic waters comprises of the Barents Sea, the Norwegian Sea, the Iceland Sea and shelf, the south-east Greenland shelf, the Greenland Sea, and a sector of the Arctic Ocean.

The water temperatures throughout OSPAR Region I have increased during the last decade (Drinkwater 2006; ICES 2008b). Related to this, the northern waters appear to show changes in abundances and distributions in local species and reports of higher numbers of 'southern species' appearing in areas within the OSPAR Region I., e.g. the abundance of Norwegian spring-spawning herring has risen since the temperature increases in the 1990s (Toresen and Østvedt, 2000), since then the population migrates out into the Norwegian and Greenland seas toward Iceland to feed and spawn (ACIA 2005). The comparison of the mean catches from the third quarter Norwegian Groundfish Survey in the Barents Sea during the periods II (1990-1999) and III (2000-2005) show increases for herring during the recent years but no change with respect to the earlier period I (1977-1989) (Table 2.2.1).

Also capelin, blue whiting, haddock, whiting and anglerfish have moved further north during the recent warming and show a large increase in abundance and distribution extensions (Berge *et al.* 2005, Astthorsson *et al.*, 2007). From the groundfish survey data in the Barents Sea this is obvious for haddock as well if the periods 1977-1989 and 2000-2005 are compared, although it is not the case for whiting for a small decrease is observed for the same comparison (Table 2.2.1).

Generally, the abundance of species that are on the northern boundary in these waters have increased while cold-water species have retreated (Bjoersnsson & Palsson 2004; Astthorsson and Palsson 2006; Astthorsson *et al.*, 2007). Looking at the mean catches during different periods from the 3rd quarter Norwegian Groundfish Survey data in the Barents Sea, show that most species of southern origin (OSPAR region I) increased in abundance. Thornback skate, anglerfish and whiting, were an exception to this trend (Table 2.2.1).

Table 2.2.1. Change in mean abundance (mean number per tow) in the Barents Sea (between the periods 1977-1989 vs. 2000-2005 (Q3) and 1990-1999 vs. 2000-2005 (Q3).

BARENTS SEA		1977-1989 VS 2000-2005	1990-1999 VS 2000-2005
Species	**Quarter**	**3**	**3**
Squalus acanthias	Spurdog	- -	+++
Amblyraja radiata	Thorny skate	+++	-
Raja clavata	Thornback skate	- -	- -
Gadus morhua	Cod	+	- -
Melanogrammus aeglefinus	Haddock	+++	o
Merlangius merlangus	Whiting	-	o
Lophius piscatorius	Anglerfish	- -	-
Pleuronectes platessa	Plaice	++	- -
Pollachius virens	Saithe	- -	+++
Clupea harengus	Herring	o	+++

Increase/decrease = +++/--- >100%; ++/-- 50-100%; +/- 10-50%; o = no change.

OSPAR Region II (Greater North Sea)

The largest changes in the temperature in analyses done by WGFE (ICES, 2007a) were seen in OSPAR Region II, corresponding to the observed largest changes in species distribution. Consequently the analyses within this area are extended (see section 2.5) to evaluate possible links with change in hydrodynamics and temperature.

Changes in distribution of thirty-five species were shown in analysis on the English groundfish survey in the North Sea (ICES, 2007a; see section 2.6, response to WGECO (ICES, 2007b) comments). The strongest relationship with climate-biogeography was found for three species exhibiting boreal-cold temperature distributions: herring, wolffish and Norway pout. Herring occupied a greater portion of the survey area, and was found more to the north and at greater average depth in warmer years. The southern range boundary of wolffish retracted northward in response to local warming. The depth range of Norway pout was shallower in years characterized by positive Gulf Stream Index values. When the relationship between body size and climate-distribution was assessed, it was observed that smaller species spread out and larger species retracted during warmer years changing their distribution within the North Sea.

IBTS data for the entire North Sea presented in WGFE 2007 indicated that some species significantly increased during the time periods considered, while others decreased, and further species did not show any significant North Sea-wide changes in abundance.

WGFE 2008 also considered regional differences in the changes within the North Sea. A number of studies have shown principal differences in the invertebrate and fish communities in the northern and southern North Sea, which are attributed to the hydrographic characteristics with a stable thermal stratification during summer in the north, and a permanently mixed water column in the south (e.g. Jennings *et al.* 1999, Zühlke *et al.* 2001, Callaway *et al.* 2002). Here we applied a separation of the two North Sea areas at the 50-m depth contour, roughly identical with the border between stratified and mixed water masses. For each area, the abundance data of individual fish species were compared between three periods (1977–1989, 1990–1999, 2000–2005) differing in temperature regime (section 2.3) in order to test the following hypothesis: demersal species inhabiting the shallow, mixed area experience seasonal changes of bottom temperatures with much greater amplitude than species in the deep areas that remain stratified during summer. Therefore, fish species inhabiting the southern North Sea may be expected to tolerate temperature changes much easier than species in the northern North Sea. A separate comparison of average abundances (ind. per 1-h haul) in both North Sea areas was therefore performed between the three periods. The distributions of haddock, saithe, anglerfish, and thorny skate are centered in the north (ICES, 2007a). If any of these species or rather local populations was sensitive to rising temperatures, one would expect to find a decrease in this part of the North Sea. However, abundances of saithe increased in the north (Table 2.2.2), and those of haddock increased over the entire North Sea. Thorny skate (*Amblyraja radiata*) significantly increased in the entire area between the first and second time period, but did not change strongly afterwards (see below, Tables 2.3.1 and 2.3.2). Anglerfish changed inconsistently between areas and time periods.

Among the pelagic species, sardine and anchovy strongly increased over both the first and the second time intervals [(’77-’89) to (‘90-’99) and (’90-’99) to (’00-’05)] and were increasingly present in the IBTS during the first and third quarter. Both species have highest summer abundances in the southern North Sea, while during winter additional regions of high numbers around the Scottish and British coast suggest an association to inflow from the Atlantic. This years extended analyses show that anchovy increased over the whole survey area (Table 2.4.2 and Figure 2.4.1) and that they consistently increased in abundance in the southern North Sea between all periods and quarters (Table 2.2.3). Horse mackerel only increased from the first to the second time interval and sprat showed no changes at all (ICES, 2007a). Increasing numbers within demersal populations were recorded for hake (northern area), striped red mullet, John Dory and anglerfish. Decreases in demersal populations were particularly obvious in Atlantic cod. Of these species hake, striped red mullet and John dory showed a consistent increase in abundance in the Northern North Sea (Table 2.2.2) and striped red mullet also in the southern North Sea (Table 2.2.3) making this likely to be related to changes in temperature. This year’s extended analysis indicates that Atlantic cod decreases over the whole area and that the densities of anglerfish changed from west to east with a particular high increase in the north-east (Table 2.4.2 and Figure 2.4.7). Among elasmobranch species, lesser spotted dogfish and thorny skate increased in abundance, while spurdog and thornback skate decreased (ICES, 2007a). The mean abundances analyses confirm these results for thornback skate (northern area, Table 2.2.2), spurdog and lesser spotted dogfish (southern area, Table 2.2.2). While this year's extended analyses indicated a change in the densities of thorny skate from west to east (Table 2.4.2 and Figure 2.4.6). Changes in some of the commercially important flatfish (plaice, sole) were not apparent in the GOV-based IBTS data set and are better represented in beam trawl surveys. This extended analysis indicate that sole changed in densities from the central (offshore) area to the coast, particularly to the south and west; thus, the quarter 1 distribution now more closely resembles quarter 3 (Table 2.4.2 and Figure 2.4.5). Plaice showed a consistent increase in abundance between the different periods in quarter 1 and 3 in the northern North Sea (Table 2.2.3).

Table 2.2.2. Consistent change in mean abundance in the Northern North Sea between the periods 1977-1989 vs. 2000-2005 (Q1) and 1990-1999 vs. 2000-2005 (Q1 and Q3). The table only lists species for which a consistent increase or decrease in abundance was documented; results for all 22 species are given in table 2.3.1 and 2.3.2 .

NORTHERN NORTH SEA		1977-1989 VS. 2000-2005	1990-1999 VS. 2000-2005	1990-1999 VS. 2000-2005
Species	**Quarter**	**1**	**1**	**3**
Raja clavata	Thornback skate	- -	- -	- -
Merlangius merlangus	Whiting	- -	-	- -
Merluccius merluccius	Hake	++	+	+
Pleuronectes platessa	Plaice	+++	+	++
Sardina pilchardus	Pilchard	+++	+++	+++
Pollachius virens	Saithe	+++	+++	+++
Zeus faber	John dory	+++	+++	+++
Mullus surmuletus	Striped red mullet	+++	+++	+++

Increase/decrease = +++/--- >100%; ++/-- 50-100%; +/- 10-50%; o = no change.

Table 2.2.3. Consistent change in mean abundance in the Southern North Sea between the periods 1977-1989 vs 2000-2005 (Q1) and 1990-1999 vs 2000-2005 (Q1 and Q3). The table only lists species for which a consistent increase or decrease in abundance was documented; results for all 22 species are given in table 2.3.1 and 2.3.2

SOUTHERN NORTH SEA		1977-1989 VS. 2000-2005	1990-1999 VS. 2000-2005	1990-1999 VS. 2000-2005
Species	**Quarter**	**1**	**1**	**3**
Scyliorhinus canicula	Lesser spotted dogfish	+++	+++	+
Squalus acanthias	Spurdog	- -	- -	- -
Sprattus sprattus	Sprat	++	+	+
Engraulis encrasicolus	Anchovy	+	+++	+++
Melanogrammus aeglefinus	Haddock	+++	+++	++
Solea vulgaris	Sole	- -	-	-
Mullus surmuletus	Striped red mullet	+++	++	o

Increase/decrease = +++/--- >100%; ++/-- 50-100%; +/- 10-50%; o = no change.

The North Sea area is intensely studied, and there are many articles on changes in the species composition. Here we will extend the analysis done by WGFE with the literature of the most recent years.

North Sea winter bottom temperature has increased by 1.6°C over 25 years, with a 1°C increase in 1988–89 alone. During this period, the entire demersal fish assemblage deepened by ~3.6 m decade^{-1} and the deepening was coherent for most assemblages (Dulvy *et al.*, in press, see section 2.7). The latitudinal response to warming was heterogeneous, and reflects (i) a northward shift in the mean latitude of abundant, widespread thermal specialists, and (ii) southward shift of small, abundant southerly species with limited occupancy in the North Sea. The centres of distribution of 15 out of 36 demersal species shifted by distances ranging from 48 to 403 km during the period 1977–2001 (Perry *et al.*, 2005). For blue whiting (*Micromesistius poutassou*) and redfishes (*Sebastes* spp.) the expectation is that they may retract completely from the North Sea by 2050. The mean latitude of species such as Atlantic cod and anglerfish shifted north in relation to warming whereas mean latitude did not change for some species but rather shifted to deeper, cooler depths with warming, e.g. plaice (Van Keeken, *et al.* 2007; Perry *et al.*, 2005), cuckoo ray (*Leucoraja naevus*) (Perry *et al.*, 2005). Hedger *et al.* (2004) also analysed the survey distributions of Atlantic cod and haddock in the North Sea. These authors showed that cod were found in deeper water during 1990–1999 compared to 1980-1989, but their distribution with respect to temperature was unchanged. The deepening response of North Sea bottom-dwelling fishes to climate change (Dulvy *et al.* in press) is the marine analogue of the upward movement of terrestrial species to higher altitudes. The assemblage-level depth responses, and both latitudinal responses, co-vary with temperature and environmental variability in a manner diagnostic of a climate change impact. The depth response of the demersal fish assemblage to temperature could be used as a biotic indicator of the effects of climate change in the North Sea and other semi-enclosed seas (Dulvy *et al.* in press).

Fishes with southerly biogeographical affinities are more commonly appearing in North Sea surveys in recent years and include species such as anchovy, striped red mullet, pilchard, John Dory and snake pipefish (Beare *et al.*, 2004; ICES, 2006). While Beare *et al.* 2004 terms these "waves of invaders", WGFE (ICES, 2007a) showed that they are actually good recruitment pulses of a normally low abundance resident species like the anchovy. Brander *et al. (*2003) suggested that sardines and anchovies have moved northward and have extended their ranges to include western Norway in recent years. Anchovy and sardine were almost totally absent from the North Sea until the mid 1990s, however small numbers had occasionally and sporadically been caught in the northern North Sea since 1925 (Beare *et al.* 2004). Other species have increased in abundance including: red and tub Gurnards, bib, poor cod, john dory, horse mackerel, mackerel, and lesser weaver (Beare *et al.*, 2004). Most species classified as having southern biogeographic affinities exhibited almost exponential increases in abundance since the mid-1990s, which might be though to support the case of Corten & van de Kamp 1996. They concluded that the increases were not part of a systematic long-term trend, but the result of increased transport of southern fish species into the North Sea and favourable winter temperatures. The increases however were common to what is a very diverse range of fish species, encompassing a variety of taxa and habitat preferences. The lesser weever, for example, is always confined to shallow, southern parts of the North Sea whereas the bluemouth is a deepwater shelf-edge species, yet both have shown similar increases since 1990.

The main ‘invasion’ of the northern North Sea by bluemouth took place in 1991 with the sudden appearance of many small (6-8 cm) individuals (Heessen *et al.* 1996), which became spread all around the North Sea up until 1997. Abundance then began to diminish and observations became almost exclusively limited to a small part of the Dogger Bank (Mamie *et al.* 2007), and no bluemouth were recorded in Quarter 1 of 2004. Examination of the length-frequency data for bluemouth suggests that there have been 2 main recruitments (Mamie *et al.* 2007). The first occurred in 1991 and the second in 1998. Heessen *et al.* (1996) suggested that the large pulse of oceanic water that entered the North Sea in 1990 (Heath *et al.* 1991) might have transported bluemouth eggs and larvae into the area, which then developed gradually. The study of Beare *et al.* (2004) adds further weight to that theory since the influx of small bluemouth individuals noted in 1998 were probably related to eggs and larvae transported by another large Atlantic input that occurred during November and December 1997 (Edwards *et al.* 1999).

There have been other, similar invasions (or reoccurrences) including the violet stingray *(Pteroplatytrygon violacea*), butterfish (*Stromateus fiatola*), thresher shark (*Alopias vulpinus*), and broad-billed swordfish (*Xiphias gladius*) and the current influx of snake-pipefish (*Entelurus aequoreus*) (Lindley *et al.* 2006; van Damme & Couperus 2006; Harris *et al.* 2007; Kloppmann & Ulleweit 2007). Kirby *et al.* (2006) speculate that the increased abundance of larval and juvenile snake-pipefish in CPR plankton samples as far west as the Mid-Atlantic Ridge may coincide with a rise in winter, spring and summer sea temperatures (January–September), when the eggs, which are brooded by the male, are developing and the larvae are growing in the plankton. Van Damme & Couperus (2006) put forward the alternative hypothesis that the sudden appearance of this species is a result of changes in zooplankton availability which in turn has been caused by changes in the hydroclimatic environment.

OSPAR Region III (Celtic Seas)

Analyses carried out by WGFE (ICES, 2007a) showed an increase in densities of hake, anglerfish, John Dory and lesser spotted dogfish, and a decrease in cod density. This year’s analysis (Table 2.2.4) shows an increase in abundance in a larger set of species which is usually different for the waters west of Scotland and the Celtic Sea. Sprat, anchovy, blue mouth and pilchard are the only species showing a consistent pattern (decrease or increase) in both seas and in the comparison between the different periods. The results of the other species are shown in the section 2.3. These changes in abundance are solely in the surveys used and don’t have to be related to changes in the hydrographic parameters. However the temperature time series provided by WGOH (ICES, 2008b) for this region show a clear increase in temperature in the later period and the first three species that show a consistent increase are Lusitanian warm generalist species and the fourth is an deepwater shelf-edge species.

Table 2.2.4. Change in mean abundance between the periods 1977-1989 and 2000-2005. The table only lists species for which a consistent increase or decrease in abundance was documented; results for all 22 species are given in table 2.3.1 and 2.3.2.

WEST OF SCOTLAND (WS) CELTIC SEA (CS)		WS 1977-1989 VS 2000-2005	WS 1990-1999 VS 2000-2005	CS 1990-1999 VS 2000-2005
Species	**Quarter**	**1**	**1**	**4**
Sprattus sprattus	Sprat	+	+++	+++
Engraulis encrasicolus	Anchovy	+++	+++	+++
Sardina pilchardus	Pilchard	+++	+++	+++
Helicolenus dactylopterus	Blue-mouth	+++	+++	+++

Increase/decrease = +++/--- >100%; ++/-- 50-100%; +/- 10-50%; o = no change.

For this region an increase in sightings of rare migrant species, blue-fin tuna (*Thunnus thynnus*), triggerfish (*Balistes capriscus*), thresher (*Alopias vulpinus*), blue sharks (*Prionace glauca*), sting-rays (Stebbing *et al.*, 2002), ocean sunfish ocean sunfish (*Mola mola*) (Houghton *et al.* 2006) and sailfin dory (*Zenopsis conchifer*) (Swaby & Potts 1999) was reported. Fluctuation in relative abundance of basking sharks within this area was positively correlated with fluctuations in SST and the NAO (Cotton *et al.* 2005). At a local scale the changes in abundance and distribution seem to be determined by the abundance of adult *Calanus helgolandicus*, while on a larger scale it was mainly SST (Sims & Quayle, 1998; Sims *et al.*, 2000). These observations indicate that although prey density is a key factor determining short-term distribution patterns, long-term behavioural choices by basking sharks may relate more closely to occupation of an optimal thermal habitat that acts to reduce metabolic costs and enhance net energy gain (Crawshaw & O'Connor, 1997; Sims *et al.*, 2003).

OSPAR Region IV (Bay of Biscay and Iberian Coast)

A complete cartography on the distribution of fish species in the VIIIc and IXa2 ICES divisions can be found in Sánchez *et al.* (1995, 2002). A complete list of fish and invertebrate by ICES division are also available in the web (www.ecomarg.net). Serrano *et al.* (2006, 2007) described the relationships between environmental variables (sediment type, organic content, temperature, salinity) and species distribution. Changes in the structure of fish communities of the Cantabrian Sea during the 90's in relation to driving oceanographic agents are described in (Sánchez & Serrano, 2003). Other papers deal with shifts in the abundance and distribution of indicator species before and after an oil spill (Sánchez *et al.*, 2006; Serrano *et al.* 2006b).

These studies have described 5 main assemblages in the area: coastal, inner shelf, middle shelf, outer shelf and shelf break. Regional differences are due to the narrower surface of the Cantabrian Sea shelf, in comparison with Galician or French shelves, that produces strong environmental gradients over a short distance. Depth is the most influential and stable factor determining the assemblages observed. The coastal, outer shelf and shelf-break communities are the strongest sources of variation, both intra-annual and interannual. The mixed influence of the two main hydrographic driving agents, Navidad poleward current and upwelling is responsible of a high percentage of interannual variability (Sánchez & Gil, 2000; Sánchez & Serrano, 2003). Sediment type has been described as a secondary structuring factor, below depth and depth-related variables (temperature, salinity), having more weight in benthic species than in demersal species (Serrano *et al*, 2007).

In the last years the number of new records of tropical affinity species has increased in the area (Arronte *et al*, 2004; Bañón, 2000, 2004; Bañón & Sande, 2008; Bañón *et al*, 2002, 2006, 2008; Punzón & Serrano, 1998). Two species related to this phenomenon were the grey triggerfish (*Balistes carolinensis*) and the flatfish (*Solea senegalensis*), previously unknown and now with a relevant biomass (Bañón *et al*, 2002). In most of the cited papers, climate change is described as the driving agent of this increase (ocean warming, changes in current patterns in the North Atlantic, bringing more southerly water into the northeast), but also increased exploration of deep-sea fish resources in recent years that has enhanced the discovery of new deepwater species northward of their known distribution area (Bañón *et al*, 2002).

Poulard & Blanchard (2005) attempted to identify the impact of large-scale hydroclimatic changes on the fish community in the Bay of Biscay using survey data from 1987 to 2002. Results suggested an increasing abundance trend with time for fish species having a wide distribution range in latitude (mainly subtropical ones), whereas the abundance of temperate and the least widely distributed species decreased steadily. They found an increase in pilchard and a decrease in anchovy, likewise they show a change in mean abundance between the 1990's and recent years (Table 2.2.5). Other species that display a large increase in their abundance in the Bay of Biscay also shown in the distribution maps from WGFE 2007 are sprat and bluemouth (ICES, 2007a).

Table 2.2.5 Change in mean abundance between the periods 1990-1999 and 2000-2005. The table only lists species for which an increase or decrease in abundance was documented; results for all 22 species are given in table 2.3.1 and 2.3.2.

BAY OF BISCAY		1990-1999 VS. 2000-2005
Species	**Quarter**	**4**
Sprattus sprattus	Sprat	+++
Engraulis encrasicolus	Anchovy	-
Helicolenus dactylopterus	Blue-mouth	+++
Sardina pilchardus	Pilchard	++
Zeus faber	John dory	+
Capros aper	Boarfish	-

Increase/decrease = +++/- - - >100%; ++/- - 50-100%; +/- 10-50%; o = no change.

OSPAR Region V (Wider Atlantic)

As in previous years, no data from this area were available for WGFE, therefore no analyses have been carried out. In general, the deep sea areas of this region the environment is considered to be less variable than surface systems. Moreover, due to the long life span of exploited species, variations in annual recruitment have a relatively minor effect on the standing biomass so short-term variability in the environment is unlikely to strongly influence stocks.

2.3 Changes in abundance

The change in abundance of the 22 species previously selected to study distributional changes in the North Sea among three periods (ICES, 2007a) was determined by comparing mean survey catches for different regions. These 22 species were selected by WGFE in 2007 because they are ecologically indicative pelagic and demersal species expected to show responses to the environmental changes including the threatened and declining species identified by OSPAR. Advice on species selection was not provided either by the ad hoc group or WGECO; therefore, WGFE continued work on its original species selection, examining data from several surveys: Norwegian Groundfish Survey (Q3) in the Barents Sea, the Western Division Bottom Trawl Survey (Q1) in the waters West of Scotland, the EVHOE Groundfish Survey in the Celtic Sea (Q4) and Bay of Biscay (Q4) and the International Bottom Trawl Survey (Q1 and Q3) in the North Sea. The North Sea (excluding Skagerrak/Kattegat) was divided into a Northern and Southern region due to hydrodynamic features, since the northern inflow of the Atlantic current causes a thermocline in deeper areas that prevents warming of the bottom waters during summer and cooling during winter. The main frontal zone of this inflow coincides approximately with the 50m depth contour, which is used as the boundary for the split into the northern and southern North Sea.

The mean catch per species was calculated by averaging the mean catches per subarea (either depth range or ICES-rectangle, depending on the stratification of the survey) for the entire period. The change in abundance was determined by the difference between the means of two periods, categorised by ‘no change’ (o), a small increase or decrease of 10–50% (+ or -), a medium change of 50–100% (++ or --), and a large change of more than 100% (+++ or ---). The changes between the periods 1977–1989 and 2000–2005 are given in Table 2.3.1, and between the periods 1990–1999 and 2000–2005 in Table 2.3.2. From these tables, the species that show a consistent increase or decrease for the different periods within the same region were selected and presented in the sections dealing with the separate OSPAR regions (section 2.2).

Table 2.3.1. Change in mean abundance between the periods 1977-1989 and 2000-2005.

1977-1989 VS. 2000-2005	AREA	NSN	NSS	WS	BS
Species	**Quarter**	**1**	**1**	**1**	**3**
Scyliorhinus canicula	Lesser Spotted Dogfish	+	+++	+++	
Squalus acanthias	Spurdog	- -	- -	- -	- -
Amblyraja radiata	Thorny skate	+++	+++	-	+++
Raja clavata	Thornback skate	- -	+	++	- -
Clupea harengus	Herring	o	++	+	
Sprattus sprattus	Sprat	+++	+	+++	
Sardina pilchardus	Pilchard	+++	+++	+++	
Engraulis encrasicolus	Anchovy	+++	+++	+++	+
Lophius piscatorius	Anglerfish	+++	+++	+++	+++
Gadus morhua	Cod	- -	- -	- -	-
Pollachius virens	Saithe	+++	- -	- -	
Melanogrammus aeglefinus	Haddock	++	+	+	
Trisopterus luscus	Bib	+++	- -	+++	- -
Merlangius merlangus	Whiting	-	++	+++	
Merluccius merluccius	Hake	+++	- -	+	++
Zeus faber	John dory	+++	+++	+++	
Capros aper	Boarfish	+++	o	+++	- -
Helicolenus dactylopterus	Blue-mouth	+++	+++	+++	
Trachurus trachurus	Horse mackerel	+++	+++	++	
Mullus surmuletus	Striped red mullet	+++	+++	+++	
Pleuronectes platessa	Plaice	++	-	+++	o
Solea vulgaris	Sole	- -	o	+++	

NSN = North Sea North, NSS = North Sea South, WS = West of Scotland, BS = Barents Sea. Increase/decrease = +++/- - - >100%; ++/- - 50-100%; +/- 10-50%; o = no change.

Table 2.3.2. Change in mean abundance between the periods 1990-1999 and 2000-2005.

1990-1999 VS 2000-2005	AREA	NSN	NSN	NSS	NSS	WS	BB	CS	BS
Species	**Quarter**	**1**	**3**	**1**	**3**	**1**	**4**	**4**	**3**
Scyliorhinus canicula	Lesser spotted dogfish	+++	o	+++	+	+	o	o	
Squalus acanthias	Spurdog	++	- -	- -	- -	- -	o	o	+++
Amblyraja radiata	Thorny skate	o	o	+	-	+++	o	o	-
Raja clavata	Thornback skate	- -	- -	- -	- -	++	o	o	- -
Sprattus sprattus	Sprat	-	-	+	+	+++	+++	+++	
Engraulis encrasicolus	Anchovy	++	-	+++	+++	+++	-	+++	
Helicolenus dactylopterus	Blue-mouth	- -	-	- -	-	+++	+++	+++	
Gadus morhua	Cod	- -	- -	- -	- -	-	o	o	- -
Melanogrammus aeglefinus	Haddock	+	- -	+++	++	O	o	o	o
Merlangius merlangus	Whiting	-	- -	+	++	O	o	o	o
Solea vulgaris	Sole	-	- -	-	-	+++	o	o	
Merluccius merluccius	Hake	+	+	- -	o	-	o	o	
Lophius piscatorius	Anglerfish	o	++	++	+	O	o	o	-
Trachurus trachurus	Horse mackerel	o	++	- -	+	+++	o	o	
Pleuronectes platessa	Plaice	+	++	o	- -	+++	o	o	- -
Sardina pilchardus	Pilchard	+++	+++	- -	-	+++	++	+++	
Pollachius virens	Saithe	+++	+++	- -	+	-	o	+++	+++
Zeus faber	John dory	+++	+++	-	+++	O	+	o	
Capros aper	Boarfish	- -	+++	o	o	++	-	+	
Mullus surmuletus	Striped red mullet	+++	+++	++	o	+++	o	o	
Clupea harengus	Herring	+	o	+	-	+	o	- -	+++
Trisopterus luscus	Bib	+++	o	-	- -	+++	o	o	

NSN = North Sea North, NSS = North Sea South, WS = West of Scotland, CS = Celtic Sea, BB = Bay of Biscay, BS = Barents Sea. Increase/decrease = +++/- - - >100%; ++/- - 50-100%; +/- 10-50%; o = no change.

2.4 Case study: Changes in species distributions in relation to temperature in the North Sea

Data

Temperature

Bottom temperature readings from the North Atlantic were extracted from the ICES Oceanographic database from 1977-2006 to evaluate relationships of thermal conditions to inter-annual and seasonal variability in species distributions. These point data were then transformed to a temperature surface using potential mapping in SPANS (refer to ICES 2007a for a description of this method of interpolation).

Mapping

Distributions of 22 pelagic and demersal species were previously mapped using IBTS (GOV demersal trawl) and Norwegian (Campelen shrimp trawl) survey data, described in the report WGFE (ICES, 2007a). Rationale for species selection is described above. From the 22 species, 8 were selected for further analysis of change in distribution in the North Sea between two periods, 1977-1989 vs. 2000-2005, (Table 2.4.2). The 8 species were chosen to illustrate a variety of patterns in spatial variation.

Survey data from quarter one were used for this analysis because it was the most consistently surveyed period. The survey coverage was standardized by removing any areas not surveyed in either of the two periods (an area covering 524,400 km^2). The maps from the two periods, for quarter 1 were overlaid and density at all locations was compared. Difference in density was classified to 12 levels (12 equal areas on the map) in a matrix model (spatial modelling in SPANS, Geomatica), 6 areas of varying degree of change in density where density in the first period was higher and 6 where it was lower. The resulting reclassified map illustrated where density of fish had changed between periods and to what degree (Fig. 2.4.1 - 2.4.8). Relative change in density within each of the categories was calculated and presented in Table 2.4.1.

Case Study Results

Two of the eight species examined, ***Engraulis encrasicolus*** (anchovy) and ***Mullus surmuletus*** (striped red mullet - Lusitanian) are more common and persistent in areas south of the North Sea. Anchovy is generally thought of as a species with sub-tropical affinity (Petitgas 2008) while red mullet is a Lusitanian species with southern biogeographic affinities within the north-east Atlantic (Knijn *et al.* 1993). Engelhard *et al.* (2008) noted that the high market value of this non-quota species, its increasing presence in northern parts of its distribution range (including the North Sea) in response to warming climate, in combination with the likelihood of a more targeted fishery for the species in the future, make red mullet a highly relevant case study in the context of climate change and fishery management.

Our analysis confirmed the greatly increased presence of these two species in the North Sea in recent years. Our analysis has added spatial resolution to the observed changes. Both species increased in density over a substantial part of the study survey area (Figure 2.4.1 and 2.4.2 and Table 2.4.2). In the case of anchovy, in the early period, 1977-1989, occasional occurrences were observed off Britain and in the Skagerrak. The species underwent an expansion and is presently widely distributed (over almost 80% of the survey area) and fairly densely concentrated over much of the North Sea, except at the most northerly and westerly extent. The areas of greatest increase in density are the south central area and the Skagerrak. An increase in density was observed over 75% of the survey area and a decrease over only 1%. Red mullet, absent from the surveys during 1977-1989 (although there are occasional survey records from prior to 1977), are now relatively common in the western side of the North Sea off Britain, comprising 48% of the survey area.

Two species, ***Melanogrammus aeglefinus*** (haddock – boreal) and ***Trisopterus luscus*** (bib - Lusitanian) showed a similar pattern of change in the southern half of the North Sea, increasing in density in the northwest while decreasing the south (Figure 2.4.3 and 2.4.4 and Table 2.4.2). A decrease in density was observed over 39% of the survey area and an increase over 25% for haddock and 30% and 18% respectively for bib. Over most of the area, changes in density were moderate to low for both species; however, where haddock concentrate in the northern half of the North Sea, density remained largely unchanged. The spatial change for haddock occurred only where density was low and thus had little impact on the overall abundance of that species.

Solea solea (sole - Lusitanian), is near the northern limit of its distribution in the southern North Sea (Rijnsdorp 2008). Within that area, it is the only species of the 8 examined that decreased in density offshore during the winter, while increasing inshore (Figure 2.4.5 and Table 2.4.2). An increase in density was observed over 23% of the survey area and similarly a decrease over 24% suggesting an equal shift to the inshore between the two periods. This apparent shift is consistent with a reaction to warming in the southern North Sea. The quarter 1 distribution in 2000-2005 (more inshore) was similar to the quarter 3 distribution observed in all periods, when the bottom temperature was warmer. Climate change is expected to affect sole populations through a number of different variables and mechanisms, the most important probably being a direct result of temperature (Rijnsdorp 2008).

Amblyraja radiata (thorny skate - boreal) and ***Lophius piscatorius*** (anglerfish - Lusitanian) underwent a similar increase in density in the eastern portion of the North Sea while undergoing a decrease in the western part (Figure 2.4.6 and 2.4.7 and Table 2.4.2). An increase in density was observed over 42% of the survey area and a decrease over 37% for thorny skate and a 45% increase and 18% decrease for anglerfish indicating an eastward shift between periods.

Gadus morhua (Atlantic cod - boreal), a species of prime commercial significance, is among the top predators inhabiting the northern temperate and cold waters of the Atlantic. This species has decreased significantly in the North Sea between 1977-89 and 2000-05 (Figure 2.4.8 and Table 2.4.2; section 7.3). The reduction in density was highest to the southeast along the Dutch coast where density decreased by about a factor of 100, while a limited increase was observed along the north-eastern fringe. An increase in density was observed over 11% of the survey area and a decrease over 87%. Within the North Sea, a northward shift in the mean latitudinal distribution of cod (what was determined to be an increase in density along the north and eastern periphery of the survey area, a fairly constant decrease over the rest of the area except a very large decrease to the southeast fringe) has occurred but there is much controversy as to the causes. Causes could include active migrations (now considered unlikely), higher fishing mortality in the south, local differences in recruitment, or a mixture of this and other causes (Engelhard *et al.* 2008).

Relation to temperature change: Spatial variation and change in bottom temperature in the North Sea in 1977–89 vs. 2000–05 in the first quarter (upper panels) is illustrated in Figure 2.4.1.9 and also shows how the 8 species distributed with respect to temperature between the 2 periods (lower bar graphs). The northern North Sea is the warmest sector in the winter. However, temperatures > 7^{o} C, constrained primarily to a small area north of the Shetland Isl. during 1977–89 is now spread over most of the northern half of the North Sea and as well, northeast of the English Channel. An increase in temperature between the two periods that averaged about 1^{o} C over the entire North Sea but degree of change varied spatially. Warming was less around the periphery of the North Sea (by 0-0.6^{o} C) and about 0.6 to 1.7^{o} C over most of the rest of the area. The greatest change occurred west of the Skagerrak and part of the Dutch coast.

If a species is a “temperature keeper” (where ambient temperature has an important influence on where the species occurs, *sensu* Perry and Smith (1994)) it is expected that that such species would maintain a similar temperature range during periods of temperature change, by changing their distribution. Figure 2.4.1.9 indicates that of the 8 species examined in this case study, bib, angler and cod were the species that maintained the most similar range of temperatures between 1977–89 and 2000–2005. In the case of bib, this species tended to shift to maintain cooler temperatures. Angler shifted to the north into warmer waters but temperature range remained about the same. In the case of cod, it underwent a substantial decrease in abundance and this decrease may in part relate to the warming that occurred over its entire North Sea distribution. Hedger *et al.* (2004) also analysed the survey distributions of cod and haddock in the North Sea. These authors showed that cod were found in deeper water during 1990–1999 compared to 1980–1989, but their distribution with respect to temperature was largely unchanged. Our analysis showed that greatest decrease in density occurred where temperatures were coldest (< 5° C).

Haddock, sole and thorny skate were more densely distributed in significantly warmer waters in 2000–2005. It appears that sole, formerly moving offshore in the winter (during 1977–89), possibly to avoid colder waters can now remain inshore year round given the warmer winter temperatures there (see ICES 2007a for winter distribution maps). In the case of anchovy and red mullet, as indicated earlier, these two are more (warmer water) southerly species that have spread into the North Sea in recent years (where they were previously largely absent), possibly because of increase in temperature.

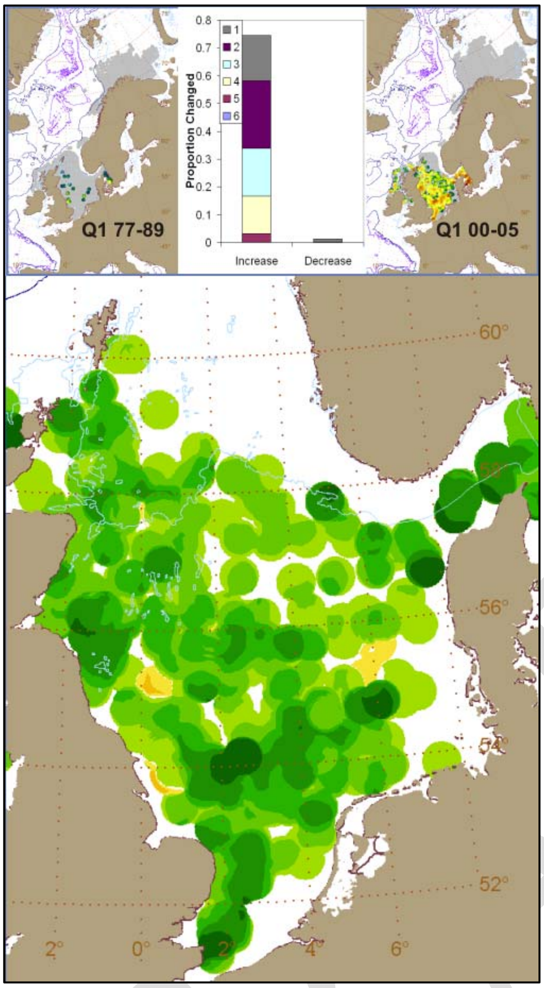
In no instances did the observed changes in distribution of any of the 8 species examined correspond completely to the observed changes in bottom temperature (distribution with respect to bottom temperature changed to some degree in all species examined). Clearly, other factors that were not examined here may be important. As well, the present analysis of changes in distribution looks at only the winter period. Patterns in distribution should also be examined during other times of the year, particularly in the summer to provide a more robust analysis of the changes that took place between 1977–89 and 2000–05. However, differences in survey areas at other times of the year make this more difficult.

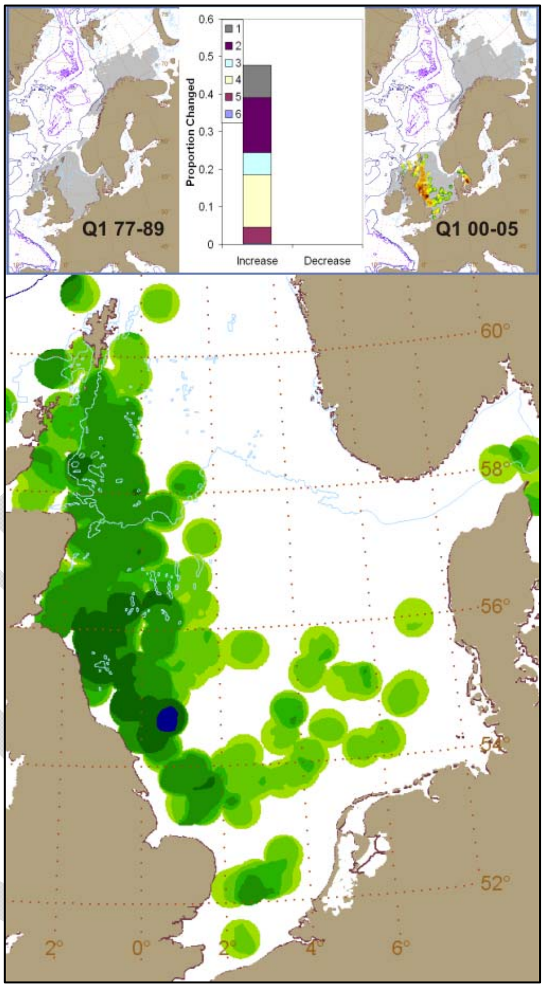
Table 2.4.1. Summary of the level of change, between periods in the categories illustrated on the maps (Figures 2.4.1-2.4.8) calculated as the average density in the change class in the first period (1977-89) divided by average density in the change class in the second period (2000-05) where an increase occurred and the inverse where a decrease occurred. X indicates change from or to zero values.

Change Class.	*Anchovy*	*Mullet*	*Haddock*	*Bib*	*Sole*	*Starry Ray*	*Angler*	*Cod*
Increase in Density within the Classification								
1		X						
2	X	X			X	X	X	
3	X	X			X	33.0	44.2	
4	344.0	X	X		25.9	22.4	9.6	0.0
5	71.2	X	4.4	15.1	6.5	11.2	3.0	1.5
6	13.0	X	3.9	X	X	3.7	3.0	
Decrease in Density within the Classification								
10			6.4	1.4	2.2	0.7	1.8	2.8
11			X	X	26.2	0.2	6.3	21.9
12				X	X	X	X	26.2
13					X	X		58.4
14						X		161.8
15								

Table 2.4.2. Summary of patterns in change in distribution of eight species in the North Sea between 1977-1989 and 2000-2005 based on IBTS survey data from quarter 1. "Increase" under "Change in Proportion of survey area" is the proportion of the total survey area where local abundance was observed to increase between periods. Conversely, "Decrease" is the proportion of the total survey area where local abundance was observed to decrease. "Distributional Changes" summarizes geographical shifts periods.

	SPECIES (CATEGORIZED BY PATTERN OF CHANGE)	CHANGE IN PROPORTION OF SURVEY AREA	DESCRIPTION OF DISTRIBUTIONAL CHANGES
Fig	**Increase over entire area**		
1	*Engraulis encrasicolus* (anchovy)	Increase - 75% Decrease - 1%	Occasional records off Britain and the Skaggarak in 1997-1989, now widely distributed and densely concentrated over much of the North Sea except the most northerly extent.
2	*Mullus surmuletus* (striped red mullet)	Increase - 48% Decrease - 0%	Not observed in the early period, now relatively common in the western side of the North Sea
	From South to North		
3	*Melanogrammus aeglefinus* (haddock)	Increase - 25% Decrease - 39%	Reduction in the southern most extent of the North Sea and an increase in the mid-extent of the area. There was almost no change in the distribution where the haddock were most highly concentrated, in the northern North Sea.
4	*Trisopterus luscus* (bib)	Increase - 18% Decrease - 30%	Decrease in the southern part of the North and an incraese in the northwest along the British coast.
	To Inshore		
5	*Solea solea* (sole)	Increase - 23% Decrease - 24%	Sole are distributed only the the southern part of the North Sea within that area, they shifted from the central (offshore) area to the coast, particularly to the south and west. The quarter 1 distribution now more closely resembles quarter 3.
	From West to East		
6	*Amblyraja radiata* (thorny skate)	Increase - 42% Decrease - 37%	This species shifted from the western half of the North Sea to the eastern part.
7	*Lophius piscatorius* (anglerfish)	Increase - 45% Decrease - 19%	This species shifted from the western half of the North Sea to the eastern part. The increase in density was particularly high in the north-east.
	Decrease over entire area		
8	*Gadus morhua* (Atlantic cod)	Increase - 11% Decrease - 87%	Cod decreased over most of the North Sea. The reduction in density was highwest to the south along the Netherlands. Some increase was observed along the north-eastern fringe.

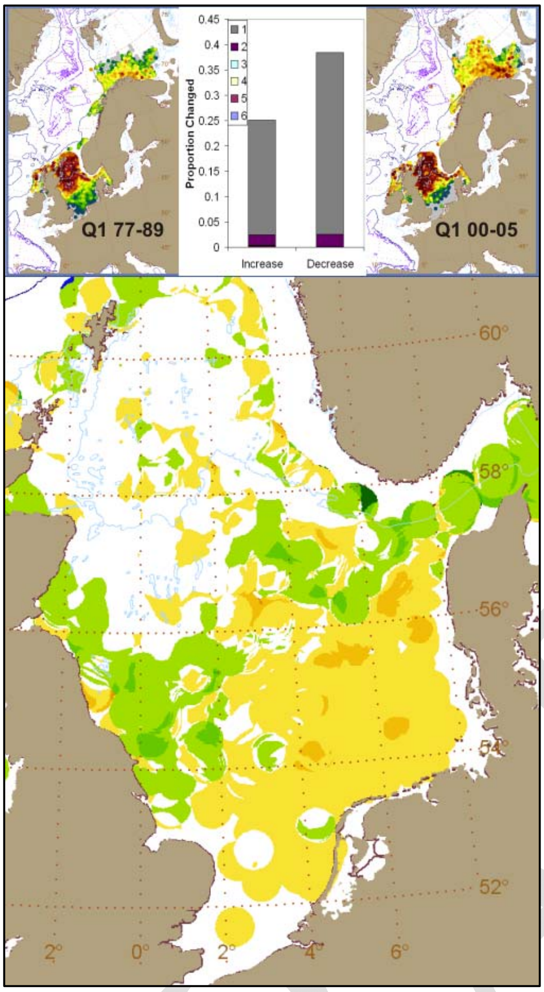


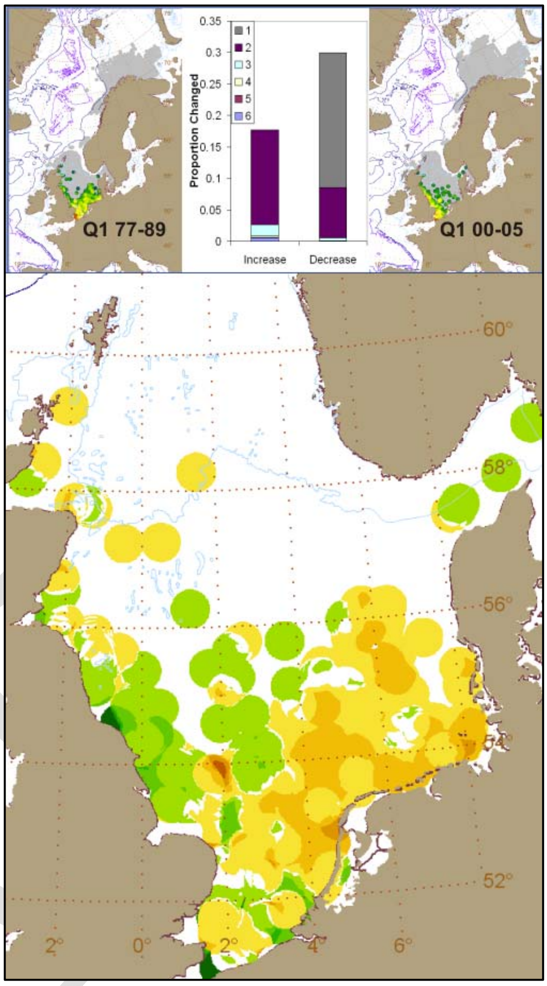


Left: Figure 2.4.1: Change in distribution of *Engraulis encrasicolus* (anchovy) between 1977-1989 and 2000-2005 in the North Sea, quarter 1.

Right: Figure 2.4.2: Change in distribution of *Mullus surmuletus* (striped red mullet) between 1977-1989 and 2000-2005 in the North Sea, quarter 1.

Upper left panel shows distribution in the initial period (1977-1989) and upper right panel for 2000-2005 (from ICES 2007a). The large lower panel shows change in distribution between the two periods. Blue to green (1-6) indicates an increase in density, dark green to blue indicating the largest change. Yellow to red indicates a decrease in density between the two periods, red indicating the largest changes. The upper centre graph shows proportion of the total survey area where an increase and decrease occurred, broken down by degree of increase or decrease (1-6).

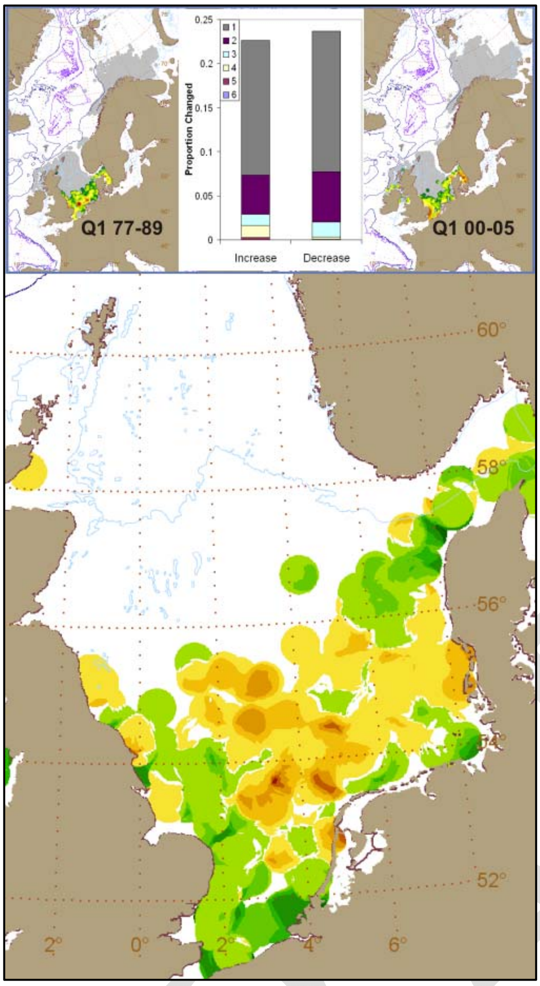


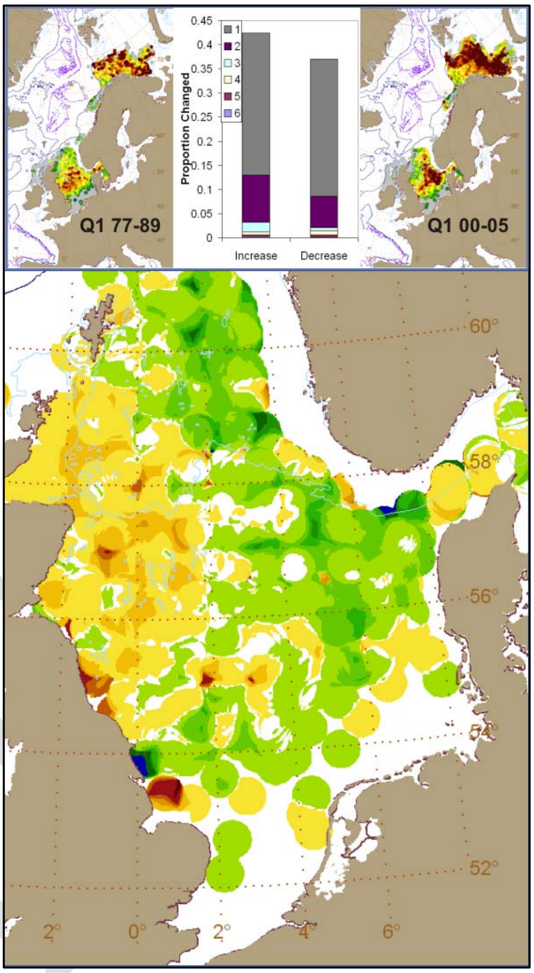


Left: Figure 2.4.3: Change in distribution of *Melanogrammus aglefinus* (haddock) between 1977-1989 and 2000-2005 in the North Sea, quarter 1.

Right: Figure 2.4.4: Change in distribution of *Trisopterus luscus* (bib) between 1977-1989 and 2000-2005 in the North Sea, quarter 1.

Upper left panel shows distribution in the initial period (1977-1989) and upper right panel for 2000-2005 (from ICES 2007a). The large lower panel shows change in distribution between the two periods. Blue to green (1-6) indicates an increase in density, dark green to blue indicating the largest change. Yellow to red indicates a decrease in density between the two periods, red indicating the largest changes. The upper centre graph shows proportion of the total survey area where an increase and decrease occurred, broken down by degree of increase or decrease (1-6).

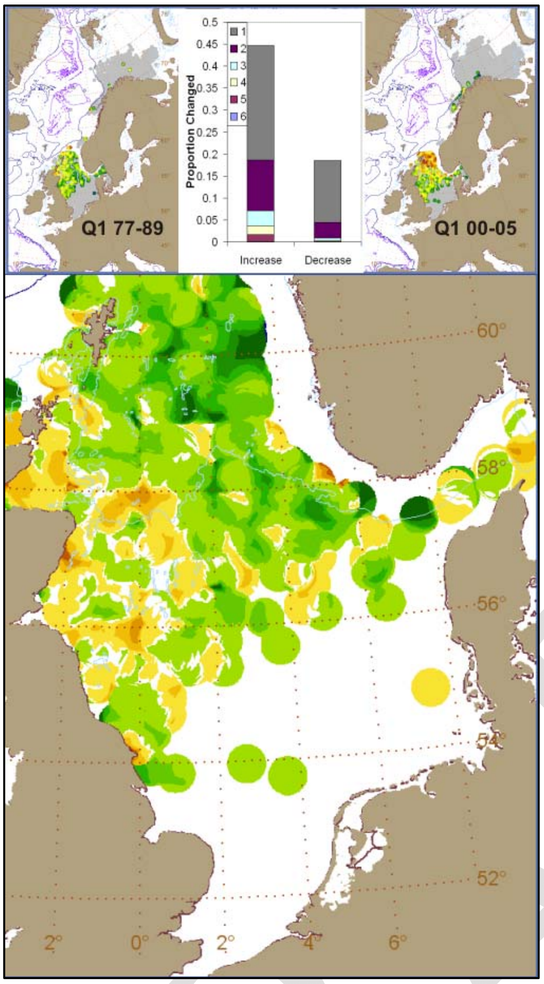


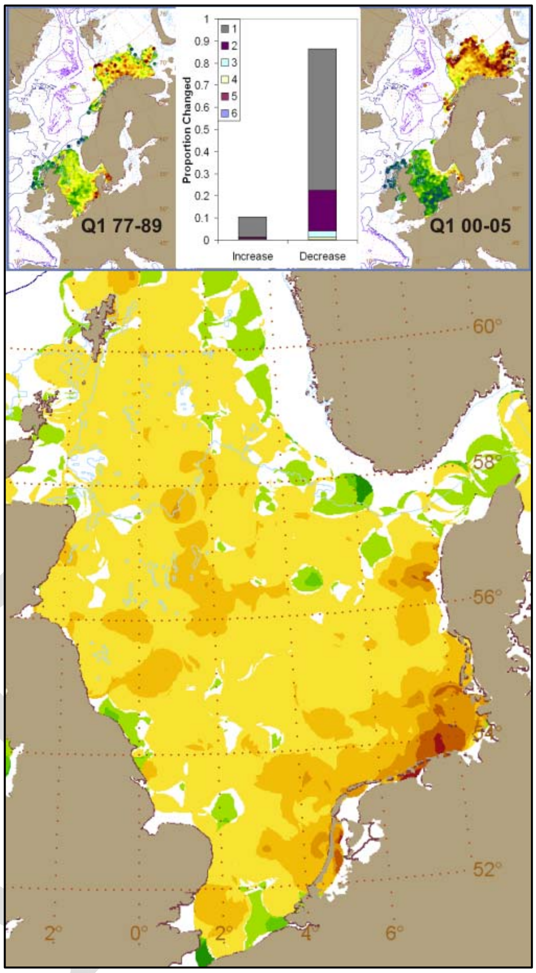


Left: Figure 2.4.5: Change in distribution of *Solea solea* (sole) between 1977-1989 and 2000-2005 in the North Sea, quarter 1.

Right: Figure 2.4.6: Change in distribution of *Amblyraja radiata* (thorny skate) between 1977-1989 and 2000-2005 in the North Sea, quarter 1.

Upper left panel shows distribution in the initial period (1977-1989) and upper right panel for 2000-2005 (from ICES 2007a). The large lower panel shows change in distribution between the two periods. Blue to green (1-6) indicates an increase in density, dark green to blue indicating the largest change. Yellow to red indicates a decrease in density between the two periods, red indicating the largest changes. The upper centre graph shows proportion of the total survey area where an increase and decrease occurred, broken down by degree of increase (1-6) or decrease (1-15).





Left: Figure 2.4.7: Change in distribution of *Lophius piscatorius* (anglerfish) between 1977-1989 and 2000-2005 in the North Sea, quarter 1.

Right: Figure 2.4.8: Change in distribution of *Gadus morhua* (Atlantic cod) between 1977-1989 and 2000-2005 in the North Sea, quarter 1.

Upper left panel shows distribution in the initial period (1977-1989) and upper right panel for 2000-2005 (from ICES 2007a). The large lower panel shows change in distribution between the two periods. Blue to green (1-6) indicates an increase in density, dark green to blue indicating the largest change. Yellow to red indicates a decrease in density between the two periods, red indicating the largest changes. The upper centre graph shows proportion of the total survey area where an increase and decrease occurred, broken down by degree of increase or decrease (1-6).

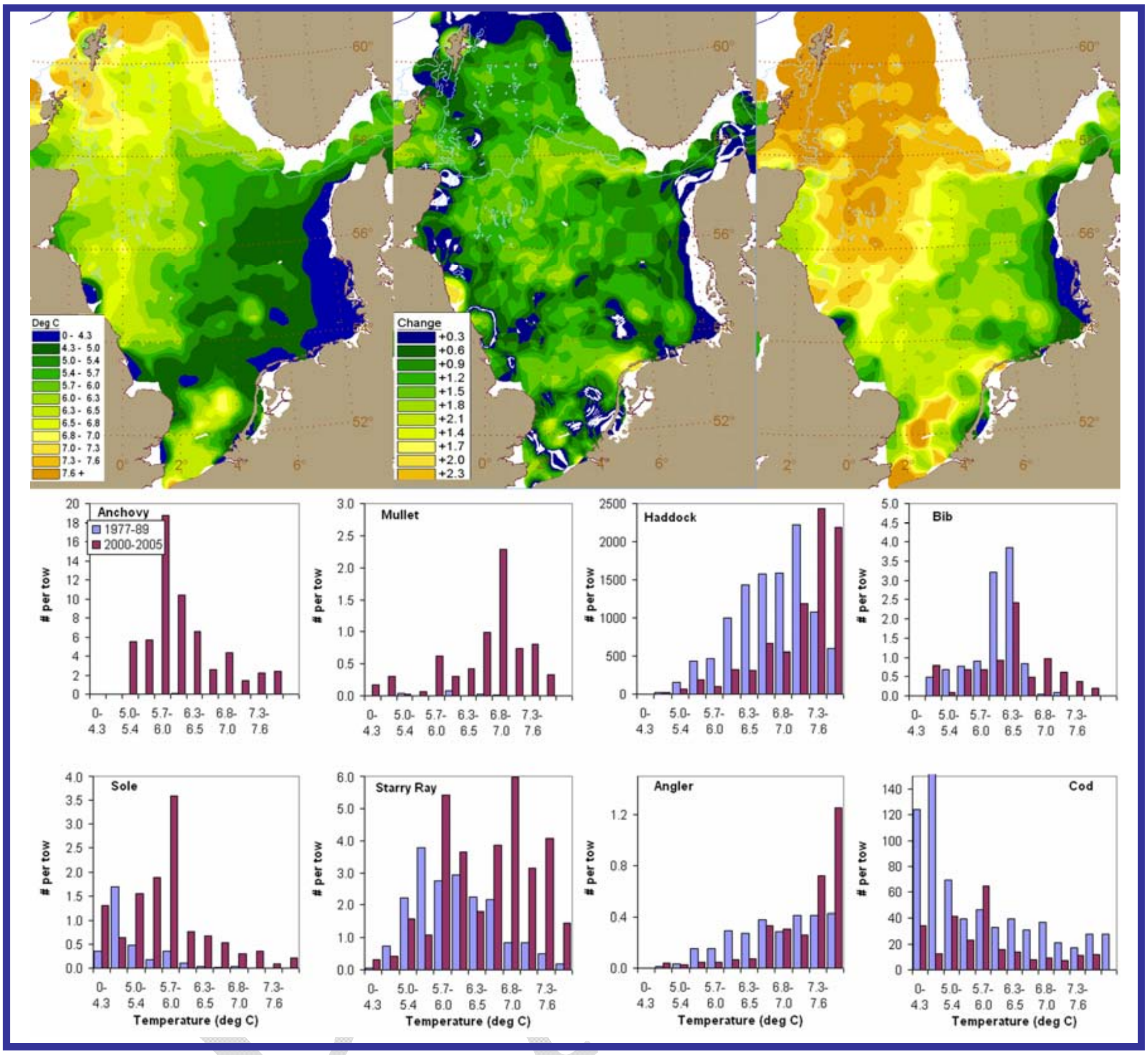


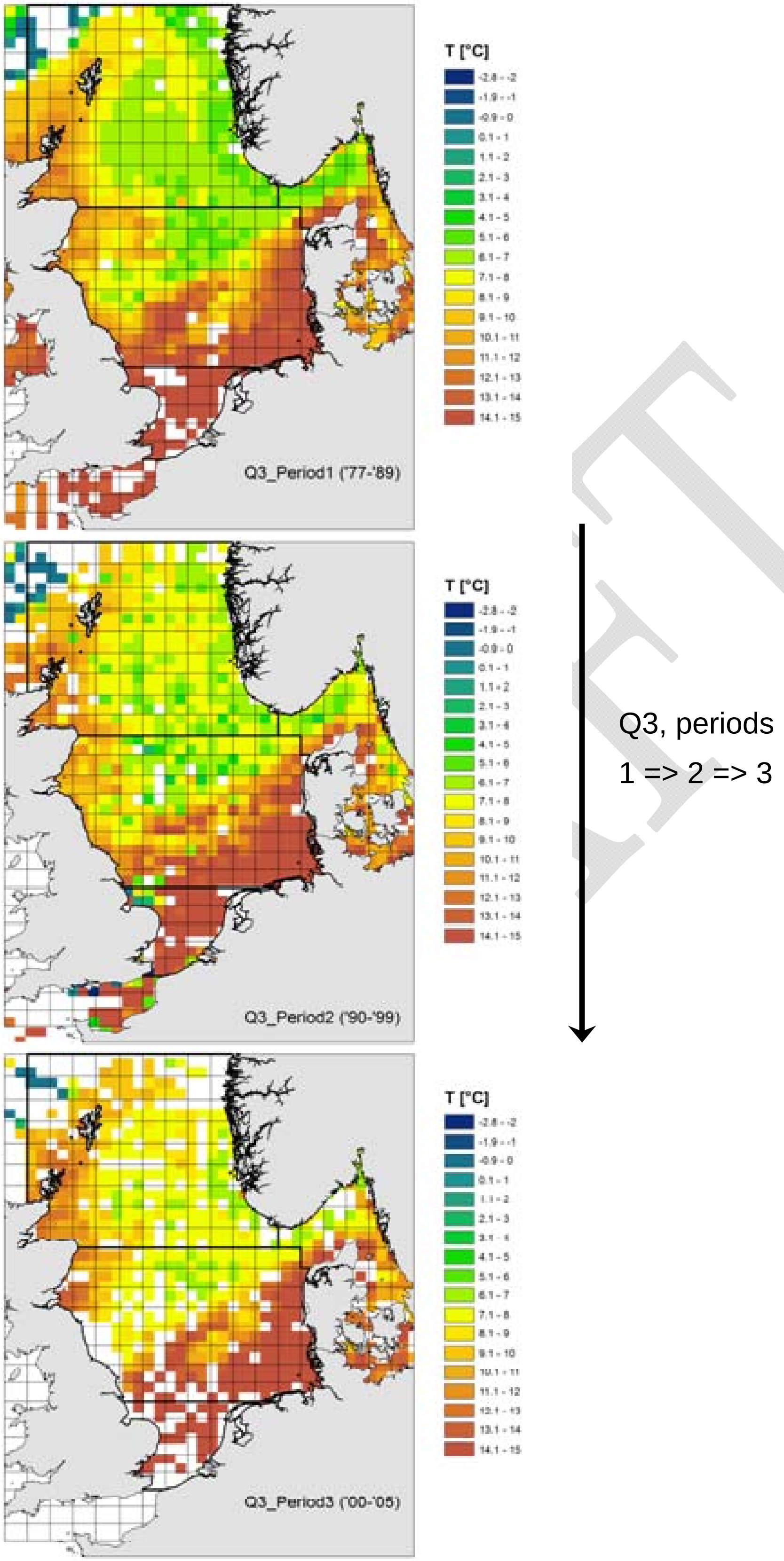
Figure 2.4.9: Upper panel (three maps, bottom temperature in the North Sea in the first quarter): Left map – 1997-2005, Right Map -2000-2005, centre map – difference between 1977-1989 and 2000-2005. Lower bar graphs – average fish density (GOV gear survey numbers per hour) within temperature categories comparing 1977-1989 to 2000-2005.

2.5 Case study: Changes in bottom temperature in the North Sea

Temperature data requested from the ICES database were analyzed from 1977–2005, consistent with the analyses in the WGFE report (ICES, 2007a). This analysis examined changes during various seasons. The geo-referenced point data were transformed into raster data with a cell size of about ¼ of an ICES rectangle. For each grid cell, the available point data from the respective period (1=1977–1989, 2=1990–1999, 3=2000–2005) were averaged and assigned to the entire cell. The procedure was repeated for each of the time periods, resulting in three raster maps of bottom temperature (Figure 2.5.1). In order to quantify spatially resolved changes in temperature, the values of the individual cells were compared and the difference mapped (Figure 2.5.2).

Within the database, the highest numbers of data points for bottom temperature were available for the longer first period (1977–89), resulting in the most complete coverage of the North Sea. During 1977–89, bottom temperatures during the first quarter of the year (Q1) in the southern North Sea were as low as zero degrees in some regions close to shore, and generally lower than 5 degrees in the majority of the southeastern section of the North Sea where water depths are less than 50 m. In the late period (2000–05), most of this region had warmed up to >= 6 °C during Q1. The resulting map of temperature change over the last ca. 20 years (Figure 2.5.2) indicates a temperature increase of up to 2-3°C in areas of the German Bight and along the Dutch coast. Temperature increases in the northern North Sea tended to be lower, but also exceeded 1 °C in many cases. Also within the last 10 years (comparison of periods 2 and 3), the southern North Sea experienced the largest increase, in many places by 1 °C (Figure 2.5.1 and 2.5.2). In the north particularly, changes remained smaller and in several places a decrease relative to the values during the second period become apparent.

During summer (Q3) greatest changes of bottom temperature within the last 20 years were again measured in the southeastern region, specifically in the German Bight and along the Danish coast where temperatures increased by 2-2.5 °C and in some cases 2.5-3 °C (Figure 2.5.2). A not quite as strong but consistent increase by about 1 to 1.5°C was seen in the central North Sea. In contrast, in a relatively wide area off the coast of Scotland summer bottom temperatures decreased by about 1 °C. Within the last 10 years, an even larger fraction of the western North Sea experienced a decrease of summer temperatures (Figure 2.5.3), which in some areas counteracted the increase observed between the periods 1 and 2.



Q3, periods
1 => 2 => 3

Figure 2.5.1: Gridded average values of bottom temperatures in the North Sea in quarter 3 during periods 1, 2 and 3.

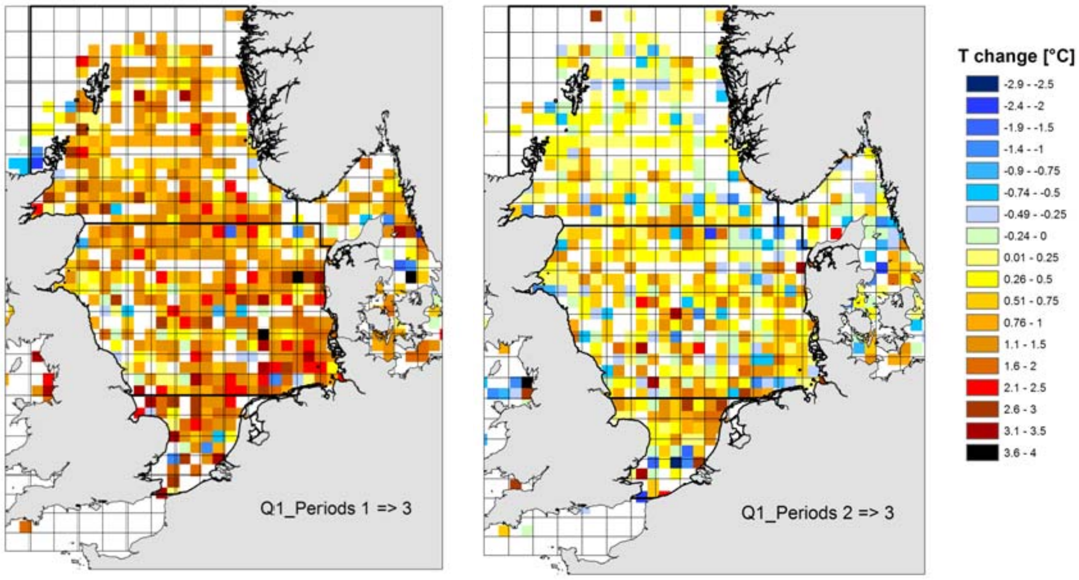


Figure 2.5.2: Gridded difference in bottom temperatures in the North Sea in quarter 1 between the periods 1 and 3 (left) and periods 2 and 3 (right). Red colours symbolize an increase in temperature, blue colours a decrease.

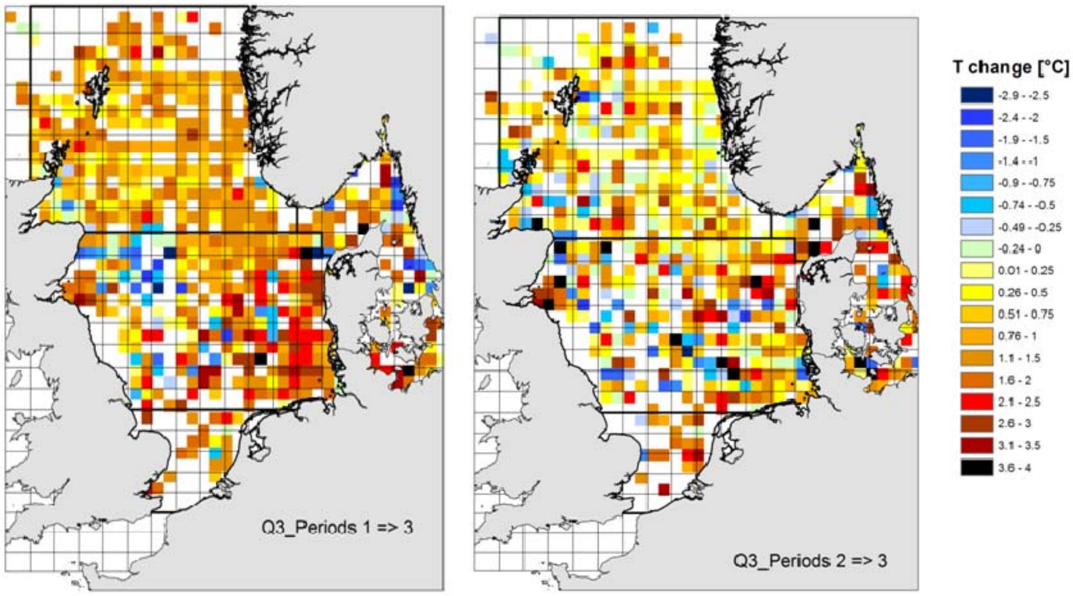


Figure 2.5.3: Gridded difference in bottom temperatures in the North Sea in quarter 3 between the periods 1 and 3 (left) and the periods 2 and 3 (right). Red colors symbolize an increase in temperature, blue colours a decrease.

2.6 Response to ICES WGECO comments (p. 110 ICES 2007b)

Bullet point 1 - Quantifying the thermal preferences of North Sea demersal fishes

If all thermal habitats available to bottom dwelling fishes were sampled in a representative manner then thermal preference of a species could be described by calculating the mean and variance of the temperature at the locations occupied by that species. There are two limitations of this approach; (i) it assumes the trawl survey encompasses and adequately describes the thermal structure of the entire habitat of North Sea fishes, and (ii) it overlooks information about the thermal properties of the sites not occupied by a species.

One approach is to use the frequency with which different temperatures were sampled across the survey grid to generate a probability distribution function of available temperatures. This can be used as a null expectation, against which the probability distribution function of the actual temperatures occupied by a fish species can be compared. Thermal *preference* may be inferred if the probability of occurrence at a given temperature is greater than expected from the thermal distribution of the survey sites. If the thermal distribution of the fish is similar to the thermal distribution of the survey then the thermal preference cannot be inferred, because the possibility that the thermal distribution of that species may solely be a function of the survey distribution cannot be ruled out. This might arise if the survey is sampling only a small proportion of the geographic and thermal range of the species.

Thermal preferences were calculated from bottom temperatures taken on all English Ground Fish Survey hauls from the 17 years including 1983-1998 and 2000. The distribution of temperatures sampled over this part of the survey ranged from 10.84 – 22.13°C and was not normally distributed: the survey encountered cooler water more often than warmer water. Across the survey grid there is a relatively low probability that cool water <12°C and that warm water >18°C are sampled (Figure 2.6.1). The northerly-distributed saithe has a greater than expected probability of occurrence in cool water between 12-15°C and a lower than expected probability of occurrence above 15 °C (Figure 2.6.2a). The southerly-distributed scaldfish has zero probability of occurrence in temperatures below 13°C and a greater than expected probability of occurrence between 15-21°C (Figure 2.6.2c).

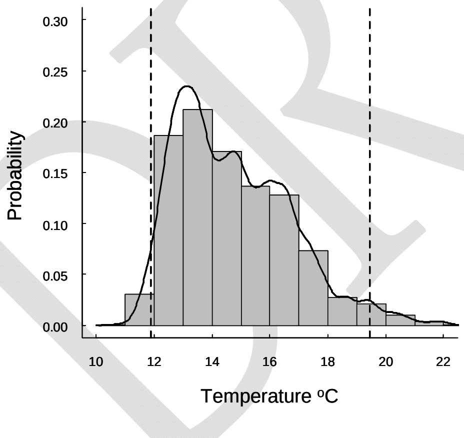


Figure 2.6.1: The probability distribution of bottom temperatures encountered by the English Groundfish Survey in autumn from 1980-2004. The mean autumn temperature during this survey is 14.4°C and the dashed lines represent the 0.025 and 0.975 quantiles in temperature (11.88-19.46°C). The histogram bars represent a 1 degree bin size whereas the line is the smoothed continuous probability distribution.

A method of visualising thermal preference may be to subtract the survey probability density function (PDF) from the fish PDF and plot the departures from the null expectation (Figure 2.6.2b & d). Positive y-axis values represent thermal preference. We summarised the thermal preference of each species using the most preferred temperature (at the peak) and the range of preferred temperatures (10th and 90th quantiles).

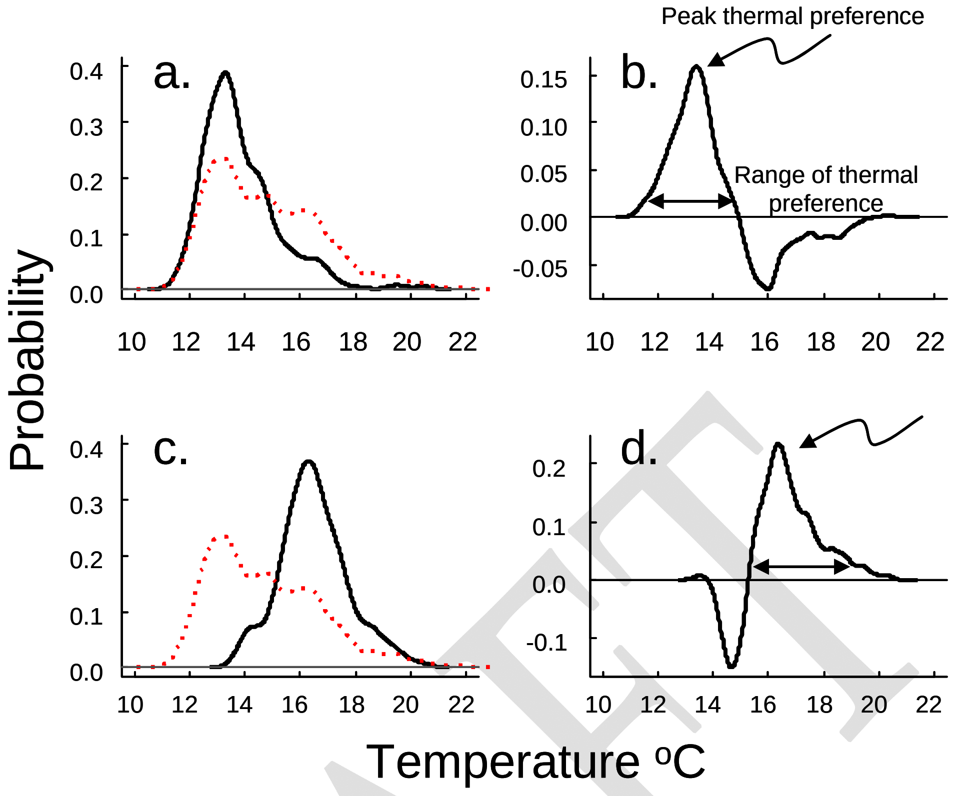


Figure 2.6.2: Autumn thermal profiles of (a,b) saithe and (c,d) scaldfish. (a,c) The probability of occurrence of each species (black line) and the probability of surveying a particular temperature (red dashed line) across the range of temperatures sampled. (b,d) Thermal preference calculated by subtracting the survey temperature pdf from the species temperature pdf.

Bullet point 2 – Linear regression analysis

Note that the ICES WGFE 2.6.3.2 Species climate variation – biogeographic distribution has is now less relevant.

The original purpose of the analysis was to determine whether certain species were more or less sensitive to climate variability and change [hereafter referred to as climate change after (Overland *et al.*, 2007)]. A standard measure of sensitivity is the slope of the relationship between climate variable and biotic response, such that a species with a steep slope is interpreted as having a higher sensitivity to climate change, and a shallower slope representing low sensitivity. The underlying simplification here is that only linear models provide a slope estimate, whereas in reality the response may be non-linear (e.g. Forcada *et al.*, 2005). The potential for non-linearity is likely to result in down-biased estimates of climate sensitivity – and hence the use of a linear descriptor of slope is likely to underestimate, rather than overestimate, climate sensitivity.

To avoid the assumption of linearity some form of multivariate method could be used, but this tests a weaker version of the hypothesis, that of *association* or *correlation* rather than *relationship*. So the original choice of hypothesis and statistical descriptor is not invalid, rather it was to address a slightly different question. Ideally both approaches would be used and if both provided consistent results would lead to greater confidence.

Bullet point 3 - Effect of NAO on species range

This is a fair point and worthy of further consideration.

Bullet point 4 – Relationship between species and temperature change

The analyses did not assume any particular mechanistic relationship between species distribution and climate change (Rindorf and Lewy, 2006). While understanding the ultimate mechanisms is a worthy and necessary exercise it may not be possible for comparative analyses of large numbers of interacting species. Indeed there may be a range of mechanisms by which climate change may influence individual species. Instead the choice was made to not assume any particular mechanistic relationship, rather than to apply a uniform assumption across all species – which is unlikely to be defensible. This correlational comparative approach offers some generality however species – environment correlations with regional climate indices tend to break down, because of the poor mechanistic understanding and non-stationarity over time (Myers, 1998; Rindorf and Lewy, 2006).

2.7 Climate change and the depth and latitudinal change in the North Sea demersal fish assemblage

European shelf seas are warming faster than the adjacent land masses and faster than the global average. Climate change impacts have been observed on individual species and species subsets, however it remains to be seen whether there are systematic, coherent assemblage-wide responses to climate change that could be used as a representative indicator of changing state.

We explore the year-by-year distributional response of North Sea bottom-dwelling (demersal) fishes to temperature change over 25 years from 1980–2004. The centres of latitudinal and depth distributions of 28 fishes were estimated from species-abundance-location data collected on an annual fish monitoring survey.

Individual species responses were aggregated into 19 assemblages reflecting physiology (thermal preference and range), ecology (body size & abundance-occupancy patterns), biogeography (northern-southern & presence of range boundaries), and susceptibility to human impact (fishery target, bycatch & non-target species).

North Sea winter bottom temperature has increased by 1.6°C over 25 years, with a 1°C increase in 1988-9 alone (refer to figures above for a spatial representation of these changes). During this period the whole demersal fish assemblage deepened by ~3.6 m decade^{-1} and the deepening was coherent for most assemblages.

The latitudinal response to warming was heterogeneous, and reflects (i) a northward shift in the mean latitude of abundant, widespread thermal specialists, and (ii) the southward shift of relatively small, abundant southerly species with limited occupancy and a northern range boundary in the North Sea.

The deepening response of North Sea bottom-dwelling fishes to climate change is the marine analogue of the upward movement of terrestrial species to higher altitudes. The assemblage-level depth responses, and both latitudinal responses, covary with temperature and environmental variability in a manner diagnostic of a climate change impact. The depth response of the demersal fish assemblage to temperature could be used as a biotic indicator of the effects of climate change in the North Sea and other semi-enclosed seas (Dulvy *et al.* in press).

2.8 RECLAIM Literature review

Throughout this chapter, WGFE made use of a review of the effects of climate on the marine biological ecosystem provided by the EU-funded project RECLAIM. The coordinator of RECLAIM unofficially requested WGFE just prior to the meeting to comment on the review, in order to make it suitable for publication as an ICES Cooperative Research Report. Due to shortage in time during the meeting, WGFE has not been able to fulfill this request. However, WGFE wishes to express that the report has been very valuable as a literature source, in particular this chapter and chapter 9, "Climate and changes in the distribution of fish".

2.9 Recommendations

In the analysis and results presented above where the changes are related to changes in hydrography and temperature were also likely influenced by other drivers confounding with the hydrographic attributes. The disentanglement with fishery effects as already mentioned for the North Sea is an example of that. WGFE recommends further work on changes in distribution and abundance of species with respect to hydrodynamics, specifically focusing on disentangling confounding drivers.

2.10 References

ACIA. 2005. Arctic Climate Impact Assessment. Cambridge University Press, 1042 pp.

Arronte, J.C.; Pis-Millan, J.A.; Fernández, M.P. and García, L. 2004. First records of the subtropical fish Megalops atlanticus (*Osteichthyes: Megalopidae*) in the Cantabrian Sea, northern Spain. Jour. Mar. Biol. Ass. U.K., 84, 1091-1092

Astthorsson, O. S., and Palsson, J. 2006 MS. New fish records and records of rare southern species in Icelandic water in the warm period 1996–2005. ICES CM 2006/C:20, pp.

Astthorsson, O.S., A., Gislason and S. Jonsson. 2007. Climatic variability and the Icelandic marine ecosystem. Deep-Sea Research II. In press

Beare, D., Burns, F., Greig, A., Jones E.G., Peach, K., Kienzle, M., McKenzie E. and Reid, D.G. 2004. Long-term increases in prevalence of North Sea fishes having southern biogeographic affinities. Mar Ecol Progr Ser, 284: 269-278.

Beare, D., Burns, F., Jones, E., Peach, K. and Reid, D. 2005. Red mullet migration into the northern North Sea during late winter. Journal of Sea Research, 53(3): 205.

Berge, J., Johnsen, G., Nilsen, F, Gulliksen, B., Slagstad, D. 2005. Ocean temperature oscillations enable reappearance of blue mussels Mytilus edulis in Svalbard after a 1000 year absence. Mar Ecol Progr Ser 303, 167-175

Bañón, R. 2000. Présence de Pteroplatytrygon violacea (Myliobatiformes, Dasyatidae) dans des eaux du Golfe de Gascogne (Nord-Est Atlantique). Ann. Soc. Sci. Nat. Charente-Maritime, VIII, 9, 1069-1074

Bañón, R. 2004. New records of two southern fish in Galician waters (NW Spain). Cybium, 28, 4, 367-368

Bañón, R.; del Río, J.L.; Piñeiro, C. and Casas, M. 2002. Occurrence of tropical affinity fishes in Galician waters north-west. Spain. Jour. Mar. Biol. Ass. U.K., 82, 877-880

Bañón, R.; A. Garazo and A. Fernández. 2008. Note about the presence of the lumpsucker Cyclopterus lumpus (Teleostei, Cyclopteridae) in Galician waters (NW Spain). J. Appl. Ichthyol., 24, 108–109

Bañón, R., T.W. Pietsch and C.G. Piñeiro. 2006. New record of Linophryne coronata (Lophiiformes, Linophrynidae) from the north-eastern Atlantic Ocean. Cybium, 30, 385-386

Bañón, R. and C. Sande. 2008. First record of the red cornetfish Fistularia petimba (Syngnathiformes: Fistularidae) in Galician waters: a northernmost occurrence in the eastern Atlantic. J. Appl. Ichthyol., 24, 106–107

Bjoernsson, H., Palsson, O.K. 2004 MS. Distribution patterns and dynamics of fish stocks under recent climate change in Icelandic waters. unpublished

Callaway, R., Alvsvåg, J., de Boois, I., Cotter, J., Ford, A., Hinz, H., Jennings, S., Kröncke, I., Lancaster, J., Piet, G.J., Prince, P., and Ehrich, S. 2002. Diversity and community structure of epibenthic invertebrates and fish in the North Sea. ICES Journal of Marine Science 59: 1199-1214.

Crawshaw, LI & O'Connor, C. 1997. Behavoural compensation for long-term thermal change. In Global Warming: Implications for freshwater and marine fish (eds C.M. Wood & D.G. McDonald), pp. 351-376. Cambridge University Press, Cambridge.

Corten, A., van de Kamp, G. 1996. Variation in the abundance of southern fish species in the southern North Sea in relation to hydrography and wind. ICES J Mar Sci 53: 1113–1119

Cotton, P.A., Sims, D.W., Fanshawe, S & Chadwick, M. 2005. The effect of climate variability on zooplankton and basking shark (*Cetorhinus maximus*) relative abundance off southwest Britain. *Fisheries Oceanography* 14: 151-155.

Daan, N., Gislason, H., Pope, J. & Rice, J. 2005. Changes in the North Sea fish community: evidence of indirect effects of fishing? *ICES Journal of Marine Science*, 62: 177-188.

Drinkwater, K.F. 2006. The regime shift of the 1920s and 1930s in the North Atlantic. Progr Oceanogr 68, 134-151

Dulvy, N. K., Rogers, S. I., Jennings, S., Stelzenmüller, V., Dye, S. R. & Skjoldal, H. R. (In press) Climate change and deepening of the North Sea fish assemblage: a biotic indicator of regional warming. *Journal of Applied Ecology*.

Edwards, M., John, A.W.G., Hunt, HG., Lindley, J.A. 1999. Exceptional influx of oceanic species into the North Sea late 1997. J Mar Biol Assoc UK 79:737–739

Engelhard, G. H., J. R. Ellis & J. K. Pinnegar. 2008. Climate and the distribution, abundance and productivity of Red mullet *Mullus surmuletus.* RECLAIM.

Engelhard, G. H., M. R. Heath2 & J. K. Pinnegar. 2008. Climate and the distribution, abundance and productivity of Cod *Gadus morhua* RECLAIM.

Forcada, J., Trathan, P. N., Reid, K. & Murphy, E. J. 2005. The effects of global climate variability in pup production of antarctic fur seals. *Ecology*, 86, 2408-2417.

Harris, M.P., Beare, D., Toresen, R., Nøttestad, L., Kloppmann, M., Dörner, H., Peach, K., Rushton, D.R.A., Foster-Smith, J., Wanlass, S. 2007. A major increase in snake pipefish (*Entelurus aequoreus*) in northern European seas since 2003: potential implications for seabird breeding success. *Marine Biology.*15: 973-983.

Heath, M.R. 2007. Responses of fish to climate fluctuations in the Northeast Atlantic. In: L. E. Emery [Ed], The Practicalities of Climate Change: Adaptation and Mitigation. Proceedings of the 24th Conference of the Institute of Ecology and Environmental Management, Cardiff, 14-16 November 2006. p102-116.

Heath, M.R., Henderson, E.W., Slesser, G., Woodward, E.M.S. 1991. High salinity in the North Sea. *Nature*, 352:116

Heath, M.R. and 22 others. 2003 MS. An operational model of the effects of stock structure and spatio-temporal factors on recruitment. Final report of the EU-STEREO project. FAIR-CT98-4122. 1 December 1998 – 28 February 2002. Fisheries Research Services Contract Report No 10/03. 360pp.

Hedger, R., McKenzie, E., Heath, M., Wright, P., Scott, B., Gallego, A. & Bridson, J. 2004. Analysis of the spatial distributions of mature cod (*Gadus morhua*) and haddock (*Melanogrammus aeglefinus*) abundance in the North Sea (1980-1999) using Generalised Additive Models. *Fisheries Research*, 70: 17-25.

Heessen, HJ.L., Hislop, J.R.G., Boon, T.W. 1996. Short communication. An invasion of the North Sea by blue-mouth, *Helicolenus dactylopterus* (Pisces, Scorpaenidae). ICES J Mar Sci 53(5):874–877

Houghton, J.D.R., Doyle, T.K., Davenport, J., Hays, G.C. 2006. The ocean sunfish *Mola mola*: insights into distribution, abundance and behaviour in the Irish and Celtic Seas. *Journal of the Marine Biological Association of the United Kingdom*, 86: 1237-1243.

Hutchinson, W.F., Carvalho, G.R., & Rogers, S.I. 2001. Marked genetic structuring in localised spawning populations of cod *Gadus morhua* in the North Sea and adjoining waters, as revealed by microsatellites. *Marine Ecology Progress Series*, 223: 251-260.

ICES. 2007a. Report of the Working Group on Fish Ecology. ICES CM 2007/LRC:03; 217 pp.

ICES. 2007b. Report of the Working Group on Ecosystem Effects of Fishing Activities. ICES CM 2007/ACE:04; 162 pp.

ICES. 2008a. Report of the Study Group on Working Hypotheses Regarding Effects of Climate Change. ICES CM 2008 ACOM:43, 34 pp.

ICES. 2008b. Report of the Working Group on Oceanic Hydrography. ICES CM 2008 OCC:01.

Jennings, S., Lancaster, J., Woolmer, A. & Cotter, J. 1999. Distribution, diversity and abundance of epibenthic fauna in the North Sea. - Journal of the Marine Biological Association of the United Kingdom 79: 385-399.

Kirby R.R., Johns, D.G., Lindley, J.A. 2006. Fathers in hot water: rising sea temperatures and a Northeastern Atlantic pipefish baby boom. *Biology Letters*, 2, 597–600.

Kloppmann, M.H.F. & Ulleweit, J. 2007. Off-shelf distribution of pelagic snake pipefish, *Entelurus aequoreus* (Linnaeus, 1758), west of the British Isles. *Marine Biology,* 151: 271-275.

Knijn RJ, Boon TW, Heessen HJL, Hislop JRG (eds) 1993 Atlas of North sea Fishes. In: ICES Cooperative research report, ICES, Copenhagen, pp. 194-268.

Lindley, J.A., Kirby, R.R., Johns, D,G., and Reid, P.C. 2006. MS. Exceptional abundance of the snake pipefish (*Entelurus aequoreus*) in the north-eastern Atlantic Ocean. ICES CM 2006/C:06.

Mamie, J.C.J., Beare, D.J., Jones, E.G., Kienzle, M., Dobby, H., Heath, M.R., Reid, D.G. 2007. Aspects of the distribution and growth of bluemouth (*Helicolenus dactylopterus,* Delaroche 1809) since its invasion of the northern North Sea in 1991. Fisheries Oceanography, 16, 85-94.

Myers, R. A. 1998. When do environment-recruitment correlations work? *Reviews in Fish Biology and Fisheries,* V8, 285-305.

Overland, J., Alheit, J., Bakun, A., Hurrell, J., Mackas, D. & Miller, A. 2007. *Climate controls on marine ecosystems and fisheries*. *Journal of Marine Systems,* in press.

Perry, R.I. and Smith S.J. 1994. Identifying habitat associations of marine fish using survey data: n application to the Northwest Atlantic. Can. J. Fish. Aquat. Sci 51:589-602.

Perry, A. L., Low, P. J., Ellis, J. R. & Reynolds, J. D. 2005. Climate change and distribution shifts in marine fishes. Science, 308, 1912-1915.

Petitgas, P. 2008. Case study Bay of Biscay Anchovy RECLAIM

Poulard, J.-C., Blanchard, F. 2005. The impact of climate change on the fish community structure of the eastern continental shelf of the Bay of Biscay. ICES Journal of Marine Science, 62: 1436-1443.

Punzón, A. and Serrano, A. 1998. First record of Ijimaia loppei Roul, 1922 (Lampriformes: Ateleopodidae) in the Iberian Peninsula waters. Cybium, 22, 297-299

Quero, J.C., Du Buit, M.H. and Vayne., J.J. 1998. Les observations de poissons tropicaux et le rechauffement des eaux de l'Atlantique europeen. *Oceanologica Acta* 21:345-351

Rijnsdorp A. D. 2008. Climate and the distribution, abundance and productivity of sole, *Solea solea* (L.) RECLAIM.

Rindorf, A. & Lewy, P. 2006. Warm, windy winters drive cod north and homing of spawners keeps them there. *Journal Of Applied Ecology,* 43, 445-453.

Rose G.A. 2005. On distributional responses of North Atlantic fish to climate change. *ICES Journal of Marine Science*, 62, 1360-1374.

Sánchez, F. and Gil, J. 2000. Hydrographic mesoscale structures and Poleward Current as a determinant of hake (*Merluccius merluccius*) recruitment in southern Bay of Biscay. *ICES Journal of Marine Science*, 57, 152-170

Sánchez, F.; Blanco, M. and Gancedo, R. 2002. Atlas de los peces demersales y de los invertebrados de interés comercial de Galicia y el Cantábrico. Otoño 1997-1999. Publ. Instituto Español de Oceanografía, 158 pp

Sánchez, F., F. Velasco, J.E. Cartes, I. Olaso, I. Preciado, E. Fanelli, A. Serrano and J.L. Gutiérrez-Zabala. 2006. Monitoring the Prestige oil spill impacts on some key species of the Northern Iberian shelf. *Marine Pollution Bulletin*, 53, 332-349

Sánchez, F.; Gándara, F. and Gancedo, R. 1995. Atlas de los peces demersales de Galicia y el Cantábrico. Otoño 1991-1993. *Publ. Esp. Ins. Esp. Oceanogr.*, 20, 1-100

Sánchez, F. and Serrano, A. 2003. Variability of groundfish communities of the Cantabrian Sea during the decade of the 1990s. *ICES Marine Science Symposia Series*, 219, 249-260

Serrano, A., Sánchez, F. and García-Castrillo, G. 2006a. Epibenthic communities of trawlable grounds of the Cantabrian Sea. *Scientia Marina*, 70S1, 149-159

Serrano, A., F. Sánchez, I. Preciado, S. Parra and I. Frutos. 2006b. Spatial and temporal changes in benthic communities of the Galician continental shelf after the Prestige oil spill. *Marine Pollution Bulletin*, 53, 315-331

Serrano, A., I. Preciado, E. Abad, F. Sánchez, S. Parra and I. Frutos. 2007. Spatial distribution patterns of demersal and epibenthic communities on the Galician continental shelf (NW Spain). *Journal of Marine Systems*, 71 (in press)

Sims, D.W., & Quayle, V.A. 1998. Selective foraging behaviour of basking sharks on zooplankton in a small-scale front. *Nature,* 393, 460-464.

Sims, D.W., Southall, E.J., Quayle, V.A. & Fox, A.M. 2000. Annual social behaviour of basking sharks associated with coastal front areas. *Proceedings of the Royal Society, London B,* 267: 1897-1904.

Sims, D.W., Southall, E.J., Richardson, A.J., Reid, P.C. & Metcalfe, J.D. 2003. Seasonal movement and behavior of basking sharks from archived tagging: no evidence of winter hibernation. *Marine Ecology Progress Series*, 248: 187-196.

Stebbing, A.R.D., Turk, S.M.T, Wheeler, A. and Clarke, K.R. 2002. Immigration of southern fish species to south-west England linked to warming of the North Atlantic (1960–2001). *Journal of the Marine Biological Association of the United Kingdom*, 82: 177-180.

Swaby, S.E., Potts, G.W. 1999. The sailfin dory, a first British record. J Fish Biol 54:1338–1340.

Toresen, R., Østvedt, L.J. 2000. Variation in abundance of Norwegian spring-spawning herring (Clupea harengus, Clupeidae) throughout the 20th century and the influence of climatic fluctuations. *Fish and Fisheries* 1, 231-256.

van Damme, C.J.G. & Couperus A.S. 2006 MS. Mass occurrence of snake pipefish: result of a change in climate? ICES CM2006/C:17

van Keeken, O.A., van Hoppe, M., Grift, R.E., Rijnsdorp, A.D. 2007. Changes in the spatial distribution of North Sea plaice (*Pleuronectes platessa*) and implications for fisheries management. *Journal of Sea Research* 57: 187–197.

Wright, P.J., Neat, F.C., Gibb, F.M., Gibb, I.M. 2006. Evidence for metapopulation structuring in cod from the West of Scotland and North Sea. *Journal of Fish Biology,* 69 (Supplement C): 181-199.

Zühlke, R., Alvsvåg, J., de Boois, I., Cotter, J., Ehrich, S., Ford, A., Hinz, H., Jarre-Teichmann, A., Jennings, S., Kröncke, I., Lancaster, J., Piet, G.J., and Prince, P. 2001. Epibenthic diversity in the North Sea. Senckenbergiana maritima 31: 269-281.

3 Review of ICUN decline criteria and comparison with ICES reference points

Term of reference and background

b) revisit and update the 2004 work of WGFE on the IUCN red listing criteria for marine fish in order to answer a request to:

- review the IUCN criteria and evaluate suitability and usefulness of using these on marine fish species. Compare the criteria with the ICES series of reference points.
- evaluate whether the risk assessment criteria used by IUCN are suitable for marine fish species and, if necessary, other and more appropriate methods and procedures.
- discuss and suggest procedures within ICES to gather useful biological information about non-commercial species that may improve the red list evaluation of these species.

ICES WG Fish Ecology has addressed these issues over the past three years. We provide a brief overview of the IUCN categories and criteria before addressing the TOR in detail and outline some background to threat and extinction risk in marine fishes (Appendix - section 3.4).

3.1 Review the IUCN criteria and evaluate suitability and usefulness of using these on marine fish species

A comprehensive review of threat criteria was undertaken by WGFE (ICES, 2004 p. 71-99). The conclusion was that IUCN and IUCN-like criteria while initially devised for application to mammals, birds and plants, are suitable and useful for evaluating threat in marine fish species. Here we briefly remind the reader of the structure of the criteria and categories, further detail can be sought from the comprehensive 2004 review. The IUCN Threat criteria can be used to hierarchically categorise species as one of nine categories using five methods (IUCN 2001, 2004). All taxa (except micro-organisms) can be placed in one of nine categories: Not Evaluated (NE), Data Deficient (DD), Least Concern (LC), Near Threatened (NT), Vulnerable (VU), Endangered (EN) or Critically Endangered (CE), Extinct (EX) or Extinct in the wild (EW) (Figure 3.1). The hierarchy works like a taxonomic key, the evaluator first determines whether the species is Data Deficient or otherwise working upward through the hierarchy until the final status is reached.

A species is Extinct when there is no reasonable doubt that the last individual has died and is Extinct in the Wild when it is known only to survive in cultivation, in captivity or as a naturalized population (or populations) well outside the past range.

A species is presumed EX or EW only when exhaustive surveys have been carried out:

- throughout its historic range;
- in known and/or expected habitat;
- at appropriate times (diurnal, seasonal, annual); and
- over a time frame appropriate to the species' life cycle and have failed to record an individual.

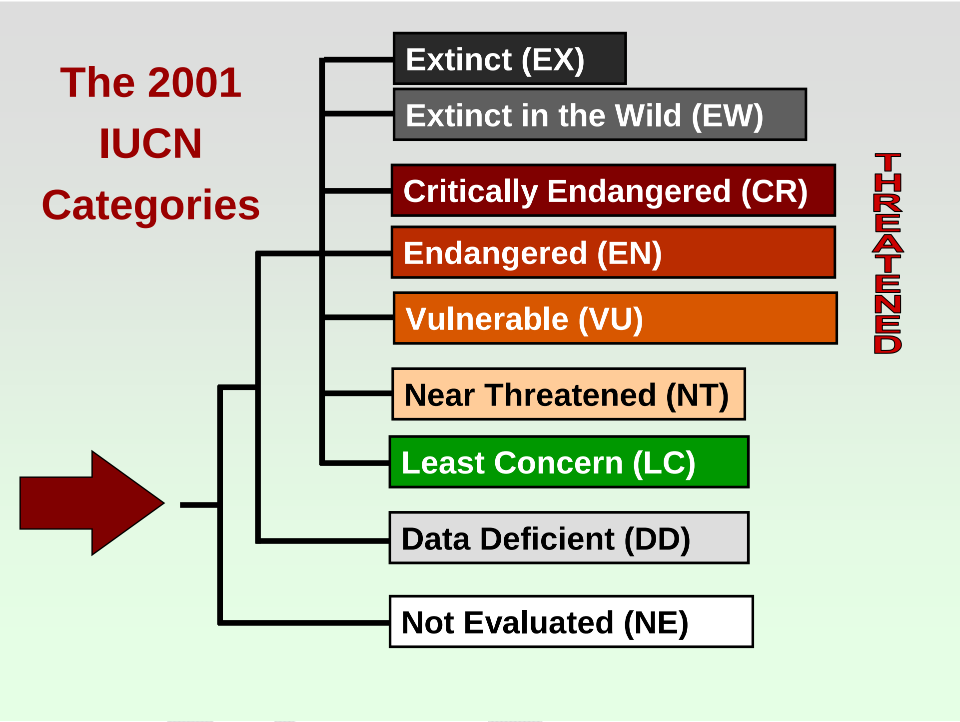


Figure 3.1: Hierarchy of the 2001 IUCN Red List assessment categories.

A species is Threatened when the best available evidence indicates that it meets any of the criteria for either Critically Endangered, Endangered or Vulnerable. When species meet these three threat categories (CE, EN, VU) they are considered to face;

- CE - an extremely high risk of extinction in the wild,
- EN - a very high risk of extinction in the wild, and
- VU - a high risk of extinction in the wild.

A species that has been evaluated against the criteria and does not qualify for a threatened category and is not Extinct or Extinct in the Wild is either:

- Near Threatened - when it is close to qualifying for or is likely to qualify for a threatened category in the near future,
- Least Concern - when it does not qualify for a threatened category or Near Threatened. Widespread and abundant taxa are included in this category,
- Data Deficient - when there is inadequate information to make a direct, or indirect, assessment of its extinction risk, and
- Not Evaluated - when it has not yet been evaluated against the criteria.

Species categorised as Data Deficient (DD) and Not Evaluated (NE) should not be treated as not threatened:

- DD only indicates that more information is required,
- future research may show that taxa currently listed as DD or NE are threatened, and
- it may be appropriate (especially for DD) to give these taxa the same degree of protection as some threatened species.

Species are categorised into the nine categories according to five criteria:

- Criterion A population reduction,
- Criterion B restricted geographic range,
- Criterion C small population size & decline,
- Criterion D very small or restricted population, and
- Criterion E quantitative analysis.

In practice most exploited species have been evaluated using the IUCN A decline criterion, such as Atlantic cod (*Gadus morhua*), bluefin tuna (*Thunnuis maccoyi*) and bocaccio rockfishes (*Sebastes paucispinus*). The use of IUCN A criterion dominates the listings of marine fishes fishes, for example, to date, of the 591 chondrichthyans assessed globally 126 species are considered Threatened and all were listed using the IUCN A decline criterion, i.e. on the basis of population declines over ten years or three generation spans, whichever is greater (Dulvy *et al.* 2008).

It is important to note that the IUCN threat categories are accorded an interpretation in terms of the relative extinction risk faced by species. This has been a point of considerable debate in the fisheries science community. There is increasing agreement that a threat-listed marine fish may face a relatively elevated risk of extinction compared to a non-threat-listed fish. However, the question remains unanswered as to whether a Critically Endangered marine fish is as close to extinction as a Critically Endangered mammal, bird, amphibian or plant. This may well be more likely and defensible for species with relatively low productivity, such as elasmobranchs, however this may not be known for relatively productive broadcast spawning fishes that previously supported large fisheries. This issue is not addressed here and threat criteria are not interpreted in term of the absolute risk of extinction.

The development and purpose of threat criteria and fisheries reference points

Threat criteria have been developed by conservation agencies and organizations largely independently of fisheries management agencies and their scientific bodies. Over the last decade a number of commercially exploited fishes have been listed using threat criteria and a lively debate between both scientific communities has ensued (Rice & Legacé 2007). Increasing numbers of jurisdictions have legislation requiring mandatory conservation measures in response to designation of a species or population being at risk of extinction and marine species are considered within these frameworks (Canadian Species at Risk Act; Australian Environmental Protection and Biodiversity Conservation Act, U. S. Endangered Species Act and Appendix 1 of CITES) (Rice & Legacé 2007).

Fisheries management can devise harvest strategies to ensure highest productivity by managing exploited fish stocks close to *target* reference points, such as MSY, or to reduce the risk of serious harm by keeping stocks above limit reference points (Caddy

& Mahon 1995). The long history of European fisheries means that few stocks are at or close to MSY, restricting the management goal to limiting serious harm and consequently stocks are typically managed to stay above a *limit* reference point. Serious harm is considered to have occurred if a stock is reduced to a size at which productivity is likely to be impaired (good recruitment is unlikely) or when the likelihood of poor productivity is increased substantially. The spawning stock biomass limit reference point associated with 'serious harm' is B_{lim}, in practice to account for assessment uncertainty managers use a precautionary reference point B_{pa} biomass (Fig 3.2), and setting catches corresponding to the fishing mortality limit reference points F_{lim} and F_{pa}.

The fisheries management goal may permit spawner biomass to drop below the limit reference point risking serious harm, in exchange for the continued social and economic benefits from ongoing fisheries. Threat criteria are intended to flag those populations that are sufficiently small as to compromise recovery and with some risk of becoming extinct. Heavily overexploited populations facing 'serious harm' may also trigger threat criteria. So there is potential for overlap and consistency in the categorization of small exploited populations that face 'serious harm' and are threatened (Jennings 2007).

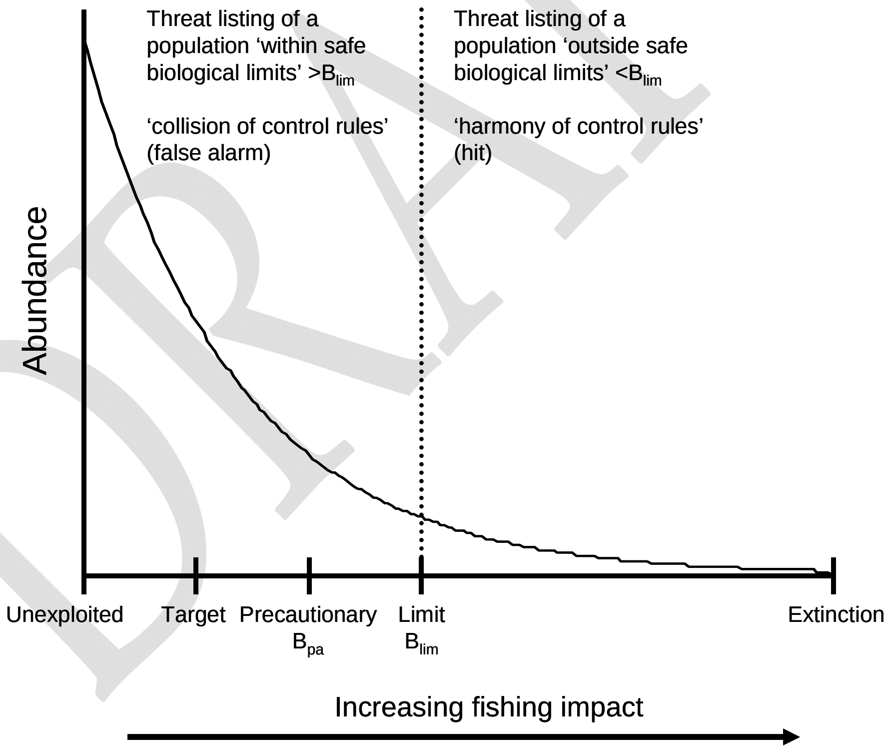


Figure 3.2: Performance of ICES fisheries references points compared to IUCN threat criteria. If the threat criteria are triggered before the population has declined below Blim then there is overlap in management triggers and the potential for a collision of control rules. This is referred to as a false alarm – where a stock is incorrectly designated as threatened yet it sustainably exploited within safe biological limits. Where threat criteria are triggered after the stock declines beyond Blim the control rules are harmonious (hit). Note while the rank order of reference points is correct, the specific location of the reference points is arbitrary and was chosen to minimise overlap in text. In reality Bpa and Blim may be closer together and lie further to the left.

3.2 Comparison of the criteria with the ICES series of reference points

The suitability and usefulness of applying IUCN A & D threat criteria have recently been evaluated and reviewed in WGFE CM 2004 G:09, pages 71–99 and in peer-reviewed literature (Dulvy *et al.* 2004; Rice & Legacé 2007). Both analyses score false alarms and hits as described in Figure 3.2.

Rice and Legacè (Rice & Legacé 2007) gathered limit reference points (B_{lim} in 2006) for 56 ICES, NAFO, DFO, NMFS assessed stocks and converted them to approximate numbers N_{lim}. These N_{lim} values were then compared to the IUCN qualifying thresholds for the A (population decline), B (restricted geographic range) and D (very small or restricted population) criteria.

Comparisons of the threat criteria with fisheries reference points could only be made for two criteria, A (population decline) and D (very small or restricted population), because the biomass reference points could not be converted to a spatial extent relevant to the B criterion. This study had three main findings:

The A population decline criterion is triggered for declines in adult abundance ≥ 50, 70 or 90% over a time spanning the greater of 10 years or three generations resulting in Vulnerable, Endangered and Critically Endangered categorisations. The maximum rate of decline over any 15-year period was calculated as an approximation to the three-generation time span (assuming all species have a 5-year generation span). Most fish stocks would qualify as vulnerable at least at some point in their documented exploitation history (Fig. 3.3). More than a quarter of stocks would qualify as Critically Endangered. The scientific advice for such stocks rarely recommended closing directed fisheries, most recommended the harvest was sustainable (Rice & Legacé 2007).

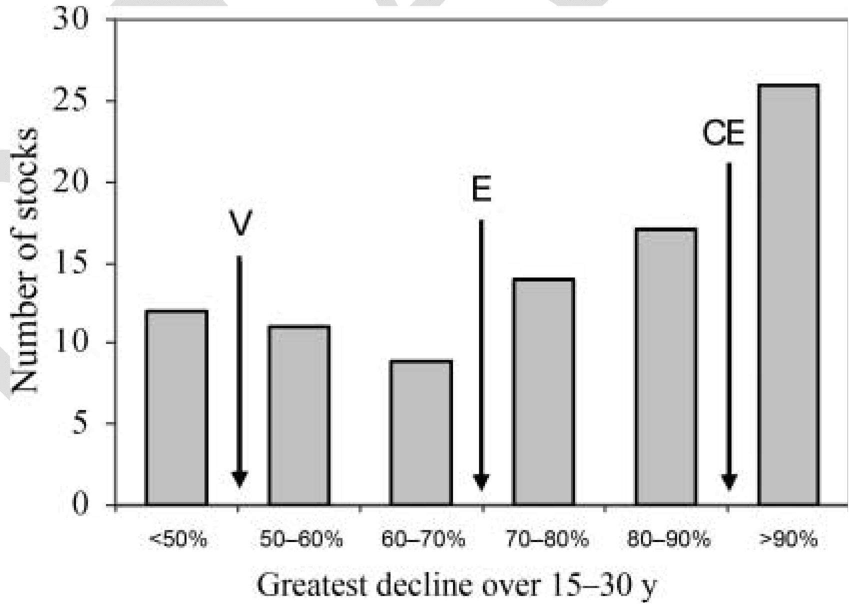


Figure 3.3: Histogram of the maximum percentage decline for 89 species within the time-series of SSB (≥35 y) from ICES, NAFO, DFO, and NMFS. Arrows represent the percentage decline corresponding to IUCN categories: V, Vulnerable; E, Endangered; CE, Critically Endangered (following Rice & Legacè 2007).

There was no overlap in N_{lim} and the D criterion thresholds (250-10,000 adults). N_{lim} values tend to be 100 times bigger than the D criterion population thresholds that would trigger a threat listing. So a declining population would tend to breach Blim thresholds well before breaching D criterion thresholds (250-10,000 adults) (Figure 3.4).

The triggering thresholds for the B criterion range from 100-20 000 km², the typical ICES rectangle size is at least 3000 km². A stock would have to occupy seven or fewer rectangles to qualify. A few stocks in the English Channel and Kattegat/Skagerrak may trigger this IUCN threat criterion, such as Clyde and Blackwater herring. However, most ICES stocks are far greater in extent of occurance than the IUCN B criterion thresholds.

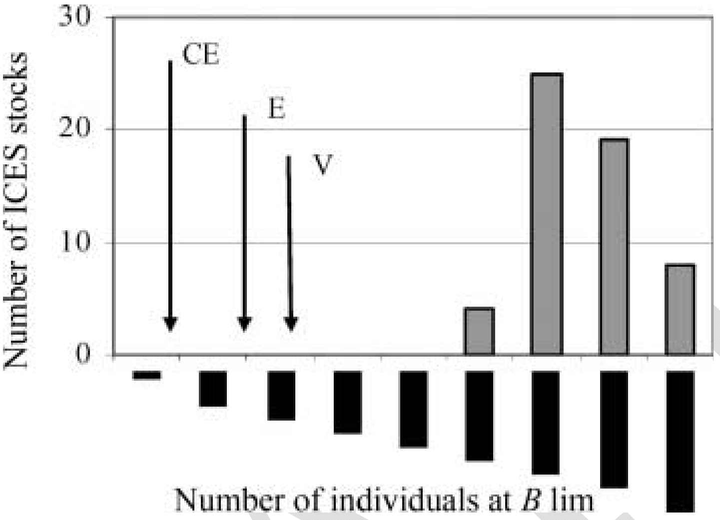


Figure 3.4: Histogram of the number of (ICES) stocks with Blim converted into approximate numbers of Nlim (x-axis in arbitrary bins). V, Vulnerable; E, Endangered; CE, Critically Endangered (following Rice & Legacè 2007).

Where comparisons between criteria can be made these findings suggest consistency (Table 3.1, Figure 3.2 'hit zone') between ICES reference points and the IUCN D (very small or restricted population) criterion. However, evaluation of the A population decline criterion uncovers many threat-listing qualifying declines for populations for which management advice was largely categorized as sustainable (Table 3.1, Fig 3.2 'false alarm zone'). This was interpreted as a conflict between fisheries management control rules and the IUCN decline criterion rules (Rice & Legacé 2007). It would be interesting to examine just the portion of these results that pertain only to the ICES area.

Using similar stock assessment data from just the ICES region, Dulvy *et al.* made a direct comparison at one point in time between IUCN A1 criteria to ICES limit reference points (B_{lim}) for up to 76 commercially exploited North-East Atlantic stocks, comprising 21 species (62 stocks) of fish and three species (14 stocks) of invertebrates. The threat and exploitation status were compared using a hits, misses and false alarms framework (Table 3.1).

Table 3.1. Performance evaluation framework for comparing the performance of threat criteria to the exploitation status of stocks reported by the Advisory Committee on Fisheries Management of the International Council for the Exploration of the Sea (Adapted from Rice 2003).

	STOCK MEETS EXTINCTION RISK CRITERION	STOCK DOES NOT MEET EXTINCTION RISK CRITERION
Exploited within safe biological limits	False alarm	Hit (true negative)
Exploited outside of safe biological limits	Hit (true positive)	Miss

This study found that a total of 46 (67%) stocks were outside safe biological limits (exploited unsustainably) and 23 (33%) were inside safe biological limits (exploited sustainably). The IUCN A1 criterion was applied to all 76 stocks, and 12 stocks declined at a sufficient rate to qualify as threatened. In a direct comparison, this study determined whether any false alarms were generated – where stocks were both classified as inside safe biological limits AND as threatened (a false alarm). IUCN A1 decline criterion did not raise false alarms when applied to ICES stocks: none of the threatened stocks was classed as 'exploited within safe biological limits' (Table 3.2). Stock status was also evaluated using the IUCN E quantitative analysis criterion using simple population viability analysis. These methods provided threat classifications broadly consistent with the decline criteria and these also did not generate false alarms (Table 3.1).

Table 3.2. The proportion (%) of stocks meeting each of four possible outcomes (true positive hit, true negative hit, miss and false alarm) and the total number of stocks for which both stock status and threat status were available. IUCN E^D was calculated using the Dennis, Munholland & Scott (1991) method and IUCN E^D was calculated using the Holmes (2001) method.

THREAT CRITERIA	FALSE ALARM	HIT (POSITIVE)	HIT (NEGATIVE)	MISS	NUMBER OF STOCKS COMPARED
IUCN A1	0	16%	36	48	64
AFS	0	2	36	62	58
IUCN ED	0	36	45	18	11
IUCN EH	0	50	25	25	12

This second study concluded that threat criteria and fisheries stock assessments provide comparable information on the status of populations exploited by North-east Atlantic European fisheries. These results lead to two conclusions. First, in every case stocks classified as threatened are unsustainably exploited 'outside safe biological limits' (Figure 3.2, 'hit zone'). Secondly, stocks categorized as 'outside safe biological limits' may, in some cases, also be considered threatened. Thus management advice to reduce fishing mortality on stocks outside safe biological limits is consistent with the requirement to reduce the risk of extinction, and reductions in fishing mortality will meet the concerns of both fisheries and conservation interests (Dulvy *et al.* 2005; Jennings 2007).

These two studies initially appear contradictory; however they focus on a differing range of management systems and different time periods. The first finding (Rice & Legacè 2007) relies on an indirect comparison among a range of fisheries management systems (ICES, NAFO, DFO & NMFS) over a range of time periods. Whereas the second finding (Dulvy *et al.* 2005) suggests that when direct comparisons between threat and exploitation status are made in the same time period (2001–2), for a consistent fisheries management framework (ICES only) then there is no evidence for conflict between or a collision of the different control rules. The former work warns that conflict and overlap of the two criteria systems may arise at earlier points in exploitation history, but for a heavily exploited system the treat criteria are consistent with fisheries management control rules. While it appears that IUCN A1 decline criteria are broadly consistent with ICES fisheries advice, it remains to be determined whether IUCN criteria are generally compatible to other fisheries management reference point frameworks, e.g. as used by DFO, NAFO & NMFS, and over different time points in the evolution of fisheries.

For historically overexploited species IUCN threat criteria may suffer a relatively high miss rate –near locally extinct species may not qualify as threatened. A number of historically overexploited species that declined prior to the implementation of the current management system, such as Atlantic sturgeon, Atlantic halibut, meagre, common skate, angel shark, Atlantic bluefin tuna (in the North Sea). For such species the IUCN A criterion qualifying decline rates occurred long before quantitative scientific observation began. Some such species may still be in decline but, declining slow enough not to trigger IUCN decline criteria may not necessarily trigger listing under the IUCN small population B criterion. The rarity of these species means that encounter rate of current surveys may be at the limit of detection of the species and trends will be difficult to detect due to the high variance of population estimates (Maxwell & Jennings 2005). The assessment of such species poses two particular problems. The IUCN A criterion may be difficult to apply because the main decline occurred before modern surveys and low capacity to detect current trends. The IUCN B small population criterion may also be difficult to apply because of very poor estimates of absolute abundance with wide confidence intervals. There may be consensus between fisheries and conservation scientists that such species may be 'in trouble' or in need of further attention but assigning threat status may be difficult.

Where a threat-listing invokes a management response from a conservation agency there has been considerable discussion of the desire to move towards harmonising fisheries management reference points and IUCN criteria and categories. IUCN-like criteria are implemented by the Committee on the Status of Endangered Wildlife in Canada [COSEWIC]. COSEWIC is the body charged with assessing the risk of extinction of species for potential protection under the 2003 Species at Risk Act (SARA). Perfect harmony – threat listing when B_{lim} is reached – is not necessarily desirable as one of the management systems may be redundant.

In conclusion, population decline rate criteria, such as IUCN A, tend to be most often used for evaluating marine fishes. When populations are exploited and decline down to comparatively small population sizes then the objectives and reference points of both fisheries managers and conservation agencies will be in agreement. The IUCN A population decline criterion is most widely used to assess marine fishes. The performance of IUCN A criterion is consistent and in harmony with ICES advice (ICES, 2005).

3.3 Evaluate whether the risk assessment criteria used by IUCN are suitable for marine fish species and, if necessary, other and more appropriate methods and procedures

This is largely a repetition of the first clause of the first question. Risk assessment criteria used by IUCN have been qualitatively (ICES, 2004, p. 72-99) and quantitatively (Dulvy *et al.* 2005; Rice & Legacé 2007) compared to other methods and procedures, including the Texel-Faial criteria, and have been found to be the most carefully-specified methodology, the most widely used and superior in performance compared to other methods e.g. AFS. These criteria have been in development for 40 years and while the application to exploited fishes was not without controversy, the IUCN criteria have been modified to better evaluate exploited marine fishes. This is not to say that all concerns have been eliminated. In particular, it is difficult to operationalise some of the narrative guidelines into a set of management rules (Rice & Legacé 2007) and there is concern that application of current qualifying decline rates to newly-opened fisheries on near virgin populations may not be appropriate (Anonymous 2005).

The original criticism of threat criteria focused on the difference in life histories of fecund broadcast-spawning wide-ranging marine fishes compared to mammals, birds and plants. It is now clear that the annual rate of intrinsic increase does not differ between these groups (Dulvy *et al.* 2003; Hutchings 2000b; Reynolds *et al.* 2005). While fish may differ in life histories they tend to have similar population dynamics (r_{max}) to mammals and birds.

3.4 Discuss and suggest procedures within ICES to gather useful biological information about non-commercial species that may improve the red list evaluation of these species

The most important information for evaluating species is a population or species scale time-series of distribution and abundance in appropriate habitats (e.g. research vessel catch rate). While there is relatively good sampling on soft-sediment habitats, other potentially important habitats may do unsurveyed, such as rocky habitat. In the absence of direct abundance information, an understanding of essential habitat can be combined with surveys of the extent and quality of habitat has been used in IUCN threat assessment of terrestrial organisms. Information on life histories, in particular related to generation time, e.g. growth rates and maturation schedules (Dulvy *et al.* 2004; ICES 2004). Estimates of sources of mortality, such as bycatch, would aid evaluation. For some deep-sea fishes the spatial extent and boundary of populations is unclear and molecular identification of population substructure might be required. The potential for using less formal information such as mammal strandings and public observer programmes might be considered.

3.5 Appendix: background information on extinction risk

Are there any marine fish extinctions?

Fisheries scientists often ask whether it is actually possible that a marine fish can be overexploited to the point where recovery is compromised or extinction occurs. The available evidence suggests many fish populations have undergone severe declines, in spawner biomass of 50–90% over a time period three generation lengths, and tend not to exhibit recovery, even when fishing pressure is reduced (Hutchings 2000a, 2001). While there have been few global marine extinctions, and these are mainly of mammals, birds and molluscs. However there have been numerous extinctions of marine populations including fish (Dulvy *et al.* 2003).

Non-recovery in declining fish populations

Spawner abundance data from 230 exploited fish populations reveal a median reduction in spawner populations of 83% from known historic (not necessarily unexploited) population sizes (Hutchings & Reynolds 2004). Few of these populations have recovered rapidly, most exhibited little or no increase in abundance 15 years after a collapse (defined as a greatest 15-year decline in population size) (Hutchings & Reynolds 2004). Recovery 5, 10 or 15 years after collapse was negatively associated with the magnitude of collapse across 90 populations from 38 species in 11 families. Only 12% of stocks recovered fully after collapse with 40% experiencing no recovery (Hutchings & Reynolds 2004). This analysis provides two lessons: first, many severe declines and collapses (as defined in this study) have occurred despite considerable management effort; second, these collapses are measured relative to recent historic maxima, whereas many of these fish stock have been exploited for centuries prior to the start of scientific data collection.

A brief overview of known marine extinctions

There is unequivocal evidence for 12 global marine extinctions – and these are mainly large birds and mammals hunted by early explorers, such as Steller's sea cow (*Hydrodamalus gigas*), the Great Auk (*Alca impennis*), and Pallas's cormorant (*Phalacrocorax perspecilliatus*) and four molluscs that suffered the ill effects of habitat loss (Carlton *et al.* 1999). An additional four mammal and bird species are listed as extinct by the World Conservation Union (IUCN) Red List including Japanese sea lion (*Zalophus japonicus*), small and large St. Helena petrels (*Bulweria bifax, Pterodroma rupinarium*) and the Tasman booby (*Sula tasmani*) (Baillie *et al.* 2004). Other 'species' (such as the Atlantic gray whale) may have gone extinct, but there is a great deal of uncertainty on whether they were taxonomically valid species or subspecies or clinal variants (Carlton *et al.* 1999). It is not clear whether species - such as the Canary Islands oystercatcher (*Haematopus meadewaldoni*), are hybrids or aberrant specimens (Carlton *et al.* 1999). In many cases there is little museum reference material to work with, so it is unlikely that this problem can be resolved easily. Excluding these uncertain records, there is good evidence that between 18-21 marine species have become globally extinct in the last 300 years.

Extinction of marine fish populations

More biodiversity has been permanently lost than might be inferred from this relatively low number of known marine species extinctions. A recent survey has highlighted that numerous population extinctions have occurred at the local and regional level and there may be more impending global-scale extinctions (Dulvy *et al.* 2003). The survey focussed on local and regional population-scale extinctions for four reasons. First, populations are often morphologically and genetically distinct (Carlton *et al.* 1999; McIntyre & Hutchings 2004; Ruzzante *et al.* 2000). Second, source populations may also rescue other sink populations and thus have the capacity to contribute to the resilience of the species as a whole (Smedbol *et al.* 2002). Third, population extinctions usually precede global extinction (King 1987; Pitcher 1998). Finally, impacts and management typically occur at the population scale.

It is difficult to imagine that a marine fish could be driven to extinction by anthropogenic activity. Species-level marine fish extinctions are suspected however they appear to be rare. However, numerous marine populations have disappeared and remain absent for a considerable period of time. The continuous absence is long enough to suggest these populations are extinct. This survey uncovered evidence for 133 local, regional and global extinctions, some of which are fishes. Local and regional scale extinctions cover the scale of small semi-enclosed seas such as the Irish Sea up to the Mediterranean Sea and ocean quadrants. There was evidence for eight new possible global extinctions of fishes, corals and algae, including:

- the green wrasse (*Anampses viridis*) endemic to Mauritius,

- the endemic Galapagos damselfish (*Eupalama azurina*) which has not been seen since the 1982-3 ENSO bleaching event despite concerted searches by ichthyologists (Edgar *et al*. submitted),
- the marine New Zealand grayling (*Prototoctes oxyrhinchus*),
- an endemic sea anemone (*Edwardsia ivella*),
- two eastern Pacific corals (*Millepora boschmai* and *Siderastrea glynni*),
- two Australian algae Bennett's seaweed (*Vanvoorstia bennettiana*) and Turkish towel algae (*Gigartina australis*) (Dulvy *et al*. 2003).

One criticism of this analysis is that these populations have truly become extinct. The unlikely alternative is that they represent shifts in dynamic geography whereby reduced abundance is associated with reduced spatial occupancy (Del Monte-Luna *et al.* 2007; MacCall 1990). Temporary extinction and recolonisation events may be particularly likely at the edge of a species' geographic range (Hanski 1998); however permanent range contractions at local scales are the stepping stones toward species extinction (King 1987). These population extinctions are not temporary: local extinctions have persisting for 64 years on average, and we have known about global extinctions for 77 years on average. Comparatively less time has elapsed for regional extinctions – 33 years – largely due to the inclusion of numerous recent fish population extinctions.

While some authors have questioned the veracity of these extinctions (Del Monte-Luna *et al.* 2007), since the Dulvy *et al.* (2003) review was published no new evidence has come to light to suggest recovery or recolonization of any of the 133 extinctions originally reported. Indeed population extinctions are turning up with greater frequency. Instead of recoveries and reappearances, evidence for additional population-level extinctions is mounting, particularly on coral reefs, including:

- the disappearance of the rainbow parrotfish (*Scarus guacamaia*), from the coastline of Brazil (Ferreira *et al.* 2005),
- population extinctions in the world's largest parrotfish, the giant bumphead parrotfish (*Bolbometopon muricatum*) from Guam and Marshall Islands and parts of Fiji (Bellwood *et al.* 2003; Donaldson & Dulvy 2004; Dulvy & Polunin 2004; Hamilton 2003),
- the local and near-global extinction of two coral-dwelling gobies (*Gobiidae*) (Munday 2004),
- probable global extinction of the endemic Galapagos stringweed (*Bifurcaria galapagensis*), and possible local extinction of another seven species (Edgar *et al.* submitted).

These newly discovered extinctions may reflect an elevated marine extinction rate. This is plausible due to accelerating threats such as exploitation, climate change and habitat loss (Edgar *et al.* 2005). However, it is likely that the detection of disappearances is increasing due to greater awareness of the potential for marine extinctions (Dulvy *et al.* 2003).

While population of extinctions of tropical invertebrates and fishes may be of little concern to the European fisheries science and policy community there is considerable evidence for the decline and extinction of numerous NE Atlantic and Mediterranean populations, e.g. skates and angel shark (Cavanagh & Gibson 2007; Rogers & Ellis 2000). There is molecular evidence for the extinction of one of the four main Atlantic cod populations in the North Sea due to exploitation (Hutchinson *et al.* 2003) and there is evidence for the extinction of Icelandic spring-spawning herring due to exploitation and habitat loss (Jakobsson 1980).

To summarise, fish population that have undergone severe declines generally show little or no recovery and there is evidence that fishing can cause the extinction of marine fish populations at least at local and regional scales.

3.6 References

Anonymous. 2005. Report on the Marine Fish Workshop and Recommendations for COSEWIC, Halifax, Nova Scotia.

Baillie, J. E. M., Hilton-Taylor, C. & Stuart, S. 2004. 2004 IUCN Red List of Threatened Species: A global species assessment. World Conservation Union (IUCN), Gland, Switzerland. pp. 217.

Bellwood, D. R., Hoey, A. S. & Choat, J. H. 2003. Limited functional redundancy in high diversity systems: resilience and ecosystem function on coral reefs. Ecology Letters, 6, 281-285.

Caddy, J. F. & Mahon, R. 1995. Reference points for fisheries management. Food and Agriculture Organisation, Rome. pp. 83.

Carlton, J. T., Geller, J. B., Reaka-Kudla, M. L. & Norse, E. A. 1999. Historical extinctions in the sea. Annual Review Of Ecology and Systematics, 30, 525-538.

Cavanagh, R. & Gibson, C. 2007. Overview of the Conservation Status of Cartilaginous Fishes (Chondrichthyans) in the Mediterranean Sea. IUCN, Gland, Switzerland. pp. 42.

Del Monte-Luna, P., Lluch-Belda, D., Serviere-Zaragoza, E., Carmona, R., Reyes-Bonilla, H., Aurioles-Gamboa, D., Castro-Aguirre, J. L., Próo, S. A. G. D., Trujillo-Millán, O. & Brook, B. W. 2007. Marine extinctions revisited. Fish and Fisheries, 8, 107-122.

Donaldson, T. J. & Dulvy, N. K. 2004. Threatened fishes of the world: Bolbometopon muricatum (Valenciennes, 1840) (Scaridae). Environmental Biology Of Fishes, 70, 373.

Dulvy, N. K., Baum, J. K., Clarke, S., Compagno, L. V. J., Cortés, E., Domingo, A., Fordham, S., Fowler, S., Francis, M. P., Gibson, C., Martínez, J., Musick, J. A., Soldo, A., Stevens, J. D. & Valenti, S. 2008. You can swim but you can't hide: the global status and conservation of oceanic pelagic sharks. Aquatic Conservation - Marine and Freshwater Ecosystems.

Dulvy, N. K., Ellis, J. R., Goodwin, N. B., Grant, A., Reynolds, J. D. & Jennings, S. 2004. Methods of assessing extinction risk in marine fishes. Fish and Fisheries, 5, 255-275.

Dulvy, N. K., Jennings, S. J., Goodwin, N. B., Grant, A. & Reynolds, J. D. 2005. Comparison of threat and exploitation status in Northeast Atlantic marine populations. Journal Of Applied Ecology, 42, 883-891.

Dulvy, N. K. & Polunin, N. V. C. 2004. Using informal knowledge to infer human-induced rarity of a conspicuous reef fish. Animal Conservation, 7, 365-374.

Dulvy, N. K., Sadovy, Y. & Reynolds, J. D. 2003. Extinction vulnerability in marine populations. Fish and Fisheries, 4, 25-64.

Edgar, G. J., Banks, S. A., Brandt, M., Bustamante, R. H., Chiriboga, A., Earle, S. A., Garske, L. E., Glynn, P. W., Henderson, S., Hickman, C. P., Miller, K. A., Rivera, F. & Wellington, G. M. (submitted) Fisheries interact with El Niño and animal grazers to threaten Galapagos marine life. Conservation Biology.

Edgar, G. J., Samson, C. R. & Barrett, N. S. 2005. Species extinction in the marine environment: Tasmania as a regional example of overlooked losses in biodiversity. Conservation Biology, 19, 1294-1300.

Hamilton, R. 2003. The role of indigenous knowledge in depleting a limited resource - a case study of the bumphead parrotfish (Bolbometopon muricatum) artisanal fishery in Roviana

Lagoon, western Province, Solomon Islands. Putting Fishers' Knowledge to Work (eds N. Haggan, C. Brignall & L. Wood), pp. 68-77. University of British Columbia, Fisheries Centre Research Reports.

Hanski, I. 1998. Metapopulation dynamics. Nature, 396, 41-49.

Hutchings, J. A. 2000a. Collapse and recovery of marine fishes. Nature, 406, 882-885.

Hutchings, J. A. 2000b. Numerical assessment in the front seat, ecology and evolution in the back seat: time to change drivers in fisheries and aquatic sciences? Marine Ecology Progress Series, 208, 299-302.

Hutchings, J. A. 2001. Influence of population decline, fishing, and spawner variability on the recovery of marine fishes. Journal Of Fish Biology, 59A, 306-322.

Hutchings, J. A. & Reynolds, J. D. 2004. Marine fish population collapses: consequences for recovery and extinction risk. Bioscience, 54, 297-309.

Hutchinson, W. F., van Oosterhout, C., Rogers, S. I. & Carvalho, G. R. 2003. Temporal analysis of archived samples indicates marked genetic changes in declining North Sea cod (Gadus morhua). Proceedings of the Royal Society of London, Series B, 270, 2125-2132.

ICES. 2004. Report of the ICES Working Group of Fish Ecology. ICES CM 2004/G:09. pp. 246.

ICES. 2005. Report of the ICES Working Group of Fish Ecology. ICES CM 2005/G:05. pp. 214.

IUCN. 2001. IUCN Red List Categories and Criteria: version 3.1. IUCN Species Survival Commission. Gland, Switzerland. pp. 30.

IUCN. 2004. Guidelines for using the IUCN Red List categories and criteria. IUCN Species Survival Commission. Gland, Switzerland. pp. 50.

Jakobsson, J. 1980. Exploitation of the Icelandic spring and summer-spawning herring in relation to fisheries management. Rapports et Procès-Verbaux des Réunions du Conseil International pour l'Exploration de la Mer, 177, 23-42.

Jennings, S. 2007. Reporting and advising on the effects of fishing. Fish and Fisheries, 8, 269-276.

King, F. W. 1987. Thirteen milestones on the road to extinction. The road to extinction (eds R. Fitter & M. S. Fitter). IUCN, Gland, Switzerland.

MacCall, A. D. 1990. Dynamic geography of marine fish populations. University of Washington, Seattle, Washington.

Maxwell, D. L. & Jennings, S. 2005. Power of monitoring programmes to detect decline and recovery of rare and vulnerable fish. Journal Of Applied Ecology, 42, 25-37.

McIntyre, T. M. & Hutchings, J. A. 2004. Small-scale temporal and spatial variation in Atlantic cod (Gadus morhua) life history. Canadian Journal Of Fisheries and Aquatic Sciences, 60, 1111-1121.

Munday, P. L. 2004. Habitat loss, resource specialization, and extinction on coral reefs. Global Change Biology, 10, 1-6.

Pitcher, T. J. 1998. A cover story: fisheries may drive stocks to extinction. Reviews in Fish Biology and Fisheries, 8, 367-370.

Reynolds, J. D., Dulvy, N. K., Goodwin, N. B. & Hutchings, J. A. 2005. Biology of extinction risk in marine fishes. Proceedings of the Royal Society of London, B, 272, 2337-2344.

Rice, J. C. & Legacé, E. 2007. When control rules collide: a comparison of fisheries management reference points and IUCN criteria for assessing risk of extinction. International Council for the Exploration of the Seas Journal of Marine Science.

Rogers, S. I. & Ellis, J. R. 2000. Changes in the demersal fish assemblages of British coastal waters during the 20th century. International Council for Exploration of the Seas, Journal of Marine Science, 57, 866-881.

Ruzzante, D. E., Wroblewski, J. S., Taggart, C. T., Smedbol, R. K., Cook, D. & Goddard, S. V. 2000. Bay-scale population structure in coastal Atlantic cod in Labrador and Newfoundland, Canada. Journal Of Fish Biology, 56, 431-447.

Smedbol, R. K., McPherson, A., Hansen, M. M. & Kenchington, E. 2002. Myths and moderation in marine 'metapopulations'. Fish and Fisheries, 3, 20-35.

Webb, T. J., Noble, D. & Freckleton, R. P. 2007. Abundance-occupancy dynamics in a human dominated environment: linking interspecific and intraspecific trends in British farmland and woodland birds. Journal Of Animal Ecology, 76, 123-134.

4 Modelling approaches useful for addressing fish community EcoQOs

4.1 Introduction

Over several years, WGECO and WGFE have developed the EcoQO for North Sea fish communities to its current form, defined in 2007 by WGECO (ICES, 2007a) as "the EcoQO for restoration/conservation of the size-structure of the fish community of the North Sea should be 'the proportion (by weight) for fish greater than 40cm in length should be greater than 0.3.'" The ToR that WGFE were asked to address now focuses on the provision of advice – what management action is required in order to achieve this EcoQO.

WGFE were therefore asked to assess the various modelling approaches, which WGECO (ICES, 2007) noted were in development, to determine which, if any, of these models might also be suitable to address this advice issue, and to this end apply the MSVPA in forecast mode (MSFOR) to complete the matrix below.

FISHING MORTALITY AVERAGED ACROSS THE SEVEN MAIN DEMERSAL SPECIES (COD, HADDOCK, WHITING, SAITHE, SOLE, PLAICE, NORWAY POUT)	ECOQO INDICATOR PROPORTION (BY WEIGHT) OF FISH > 40CM IN LENGTH	TIME TO REACH INDICATOR TARGET
0.85Fpa	0.27	?
	0.30	?
	0.33	?
1.00Fpa	0.27	?
	0.30	?
	0.33	?
1.15Fpa	0.27	?
	0.30	?
	0.33	?

To address this ToR WGFE first reviewed the results of the work carried out by WGSAM in October 2007, which was directed towards attempting to fill in the question marks in the table above. The "ecosystem/foodweb" models currently available or in development, that could also be used to complete the table are then reviewed. At present none of these models can be directly applied to complete the table, but some can be used to provide "generic" insight into the likely behaviour of the proportion of large fish (PLF) metric under different fishing scenarios. One species-specific size based model for North Sea fish is currently at an advanced stage of development and this model may be capable of being used to complete the table above, or some similar table, within the next year or so.

Necessary conditions for a model to adequately address the North Sea fish EcoQO

The question posed dictates the kind of model required. A model suitable for addressing the Term of Reference in its current state must have the following attributes.

Size-based: Since the EcoQO is size-based, involving length and weight measures, a model is required that is size-based for, at the very least, those fish species involved in the calculation of the indicator.

Dynamic: The question posed involves time-scales of changes in the indicator with fishing pressure, so a dynamics model is required.

Multi-species: The model must take account of (but not necessarily resolve individually) multiple species since the EcoQO is a community indicator. Since we are looking at community-level responses to changes in fishing, the model should capture the main processes driving ecosystem structure.

Representation of fishing: We require a representation of fishing mortality and one which is alterable in the model for the purposes of investigating the impact of changes in fishing on the community structure.

North Sea specific: Since the EcoQO has been specifically formulated for the North Sea demersal fish community, if the model is to be used as an advice tool then it must be capable of representing this specific community at some specific time.

Predictive: The model is required to make community-level predictions with appropriate uncertainty bounds.

4.2 Review of size-based fish community models

The following is a short literature review of the main size-based multispecies models, including the Hall *et al.* (2006) and Pope *et al.* (2006) models specifically mentioned by WGECO07. Their suitability for addressing the ToR is discussed. For ease of comparison, a summary of their relevance to the ToR is made in Table 4.1.

Hall *et al.* 2006

LeMANS (length-based multispecies analysis by numerical simulation) is a length-based model of a 21 species fish community loosely based on the George's Bank ecosystem; parameters have been chosen to roughly represent different species but mainly with the aim of capturing an appropriate range of fish life history characteristics. The model has not been "tuned" to George's Bank data and the authors do not expect the model to accurately represent or forecast trends in species abundances; rather, the model is designed for "investigating the dynamics of emergent properties of fish communities".

Growth. Fish are characterised by length in discrete length-classes and grow according to the von Bertalanffy formulation with empirically derived species asymptotic length and growth rate. Growth is deterministic: fish are assumed to always meet their food requirements for growth. Fish weights are calculated from the standard length-to-weight power law.

Recruitment. Recruitment is related to spawning stock biomass according to the Ricker formulation. The proportion mature at length is calculated by assuming a logistic form of maturation lengths with species-dependent empirically derived length at which 50% of individuals are mature, L_{M50}, and steepness parameter, κ. New recruits enter the smallest length-class at a defined seasonal spawning time-step during model runs.

Mortality. Fishing mortality rate is a function of length but not species according to a logistic model with L_{F50} set at 25 cm for all fish species. "Natural" mortality (encompassing sources of mortality not due to predation or fishing) is a function of length using the u-shaped beta probability function parameterised to give mortality which is high for young fish, low for intermediate length-classes and increasing for larger fish. Predation mortality inflicted by each size-class of predator species is assumed to be constant and determined by the required food for growth over the time-step. The required food is a function of weight change from one size class to the next, calculated from the von Bertalanffy growth curve, and assimilation efficiency. The efficiency is also taken to be a function of weight so that the smallest fish have an efficiency of about one third and the efficiency tends to zero as fish approach their asymptotic weight W_{∞}.

Predation. The fixed total predation mortality by length-class of predators is divided up among prey species dependent on prey abundance and "suitability" of the prey for the predator. The suitability is a vector of a binary diet variable (1 if the prey type is eaten and 0 otherwise), which defines the ecosystem topology, and a length-based preference. The preference is a log-normal function of the predator to prey length ratio which is assumed to be species-independent, i.e. all species have the same feeding patterns. Because the predation on individual species depends on prey abundance but the total food intake required by a predator for growth is assumed constant, a contribution is required from what is termed "other food". This is used to scale the predation mortality rates to empirical data.

This model is size-based, dynamic and multi-species and has a flexible representation of fishing mortality. It could potentially be modified to represent the North Sea demersal assemblage and fitted to data. For the purpose of addressing the TOR, the model may lack flexibility: the deterministic growth formulation might make it unlikely to predict changes in community structure with changes in fishing pressure, but this would require further investigation.

Pope *et al.* 2006

The length-based fish community model of Pope *et al.* 2006, was designed to explore the impact of changes in fishing pressure on the shape of the biomass-size spectrum.

The model represents what they describe as an "imaginary" fish assemblage: they model an abstract community comprising 13 species with different asymptotic lengths varying from 10 cm to 130 cm in 10 cm increments. All other life history parameters are based on or derived from this defining characteristic. The modelling strategy recognises size as a key factor driving fish ecosystem processes.

The model is steady-state and is therefore not capable of modelling transient dynamics such as may occur, for example, after a change in fishing pressure. The key output from the model is the biomass size spectrum obtained by summing the biomass at size of all species.

Growth. Growth is assumed to follow the von Bertalanffy curve with species dependent asymptotic length and growth rate. The growth rate k is assumed to be related to L_{∞} by a power law $k = \alpha L_{\infty}^{\beta}$ with the parameters α and β derived empirically from a mixture of literature values and a log-linear regression of k on L_{∞} values taken from FishBase. Uncertainty remains in these parameters. Species populations are divided into discrete 1 cm length-classes with new recruits entering the population at 5 cm for all species. The survival of individuals from one size class to the next in a year depends on length, species life history characteristics and the mortality rate Z, which varies with length and species as described below.

Mortality. In line with traditional methods (e.g. MSVPA), mortality rate is divided into components of fishing mortality F, "natural" or "background" mortality M1 and predation mortality M2. The latter encompasses predation by fish within the modelled community and predation from other sources is included in the background mortality.

The overall fishing mortality F inflicted upon the assemblage is modified by length and species based parameters. The fishing pressure varies with length according to a logistic model with 50% pressure at 33% of the species L_∞ and the same slope for all species. In addition, there is an "effort multiplier" that varies linearly with L_∞ and is centred on the species with L_∞ of 70 cm. The authors explain that this formulation is designed to capture the main features of a realistic fishery on different sizes of fish. Obviously, a different fishing scheme could easily be imposed on the modelled assemblage.

The natural mortality is constant for each species and does not depend on length.

Conversely, the predation mortality depends on length but not directly on species. The predation mortality on each length class of each species is calculated from a weighted sum of the biomass of all fish in larger length classes. The length-based weighting multiplier is a tuning parameter set to give values of the predation mortality in line with those suggested by MSVPA simulations. The biomass values are also weighted by a preference function of the ratio of predator to prey length.

Recruitment. The number of new recruits is assumed to be related to spawning stock biomass (SSB) by a power law with power ϕ that determines the strength of compensation in recruit numbers at low levels of SSB. SSB is calculated assuming a maturation length of Lmat and summing up the biomass of all fish larger than this. The authors carry out model runs for a variety of values of ϕ.

This model is size-based and multi-species and has a carefully thought out fishing representation. It is not dynamic. It is not fitted to the North Sea or any "real" ecosystem but potentially could be.

Benoit and Rochet, 2004

Benoit and Rochet (2004) present a mass balanced dynamic community biomass spectrum model for investigating the stability of and impact of fishing on the size spectrum. Species are not represented individually; the distribution of numbers (and biomass) of fish with weight is modelled with time. Energy flow through the model is governed by size-based predation. A continuous model is derived, which is then discretised in weight classes and driven by a plankton model and a "larger animals" model at the upper and lower ends of the weight spectrum, respectively. Dynamics in the mid-range, corresponding to the weight range of the fish community, is governed by predation alone. The overriding assumption is that size is the main structuring force of fish ecosystems.

Growth. Departing from the models described above, growth is not prescribed by the von Bertalanffy growth curve. Fish grow according to food intake (see below) multiplied by a constant assimilation efficiency. However, the growth curve for an individual fish predicted by the model is found to closely match the von Bertalanffy form. Realistic life expectancies as a function of fish size are also an emergent property of the model.

Predation. Within a time step, predators are assumed to search a volume of water related allometrically with their weight. During the search, the predators encounter prey of different weights, which they are assumed to consume with a probability which is related to the ratio of predator weight to prey weight. This “preference” has a peak at a specified predator to prey weight ratio and is set to 0 for prey larger than the predator. Integrating over the distribution of number of fish gives the total prey biomass consumed as a function of predator weight.

Mortality. Mortality derives from predation, “natural” mortality and fishing. The predation mortality is calculated as a function of prey size according to the description above. The natural mortality increases with body size and abundance, giving density dependent mortality. Fishing mortality is taken to be a linearly increasing function of log weight with zero fishing at a specified lower weight. The rate of increase and lower limit weight are derived empirically from ICES data. The authors vary fishing mortality to explore the impacts of fishing on the community biomass spectrum.

The model is dynamic and size-based and takes account of the full fish community, but is not resolved into individual species. There is a representation of fishing. The model is not set up for or fitted to any particular region, but could potentially be altered to represent the North Sea fish community.

Maury *et al.* 2007

The continuous biomass size spectrum model of Maury *et al.* 2007 is essentially similar to Benoit and Rochet (2004). Again, the ecosystem is not broken down into species but is modelled as a “meta-organism” with all processes (growth, feeding, mortality, etc) governed by size.

The models differ in that Maury *et al.* 2007 have assumed a Holling II functional response by predators to prey biomass. When food consumption rates do not meet that required for growth and reproduction, the energy intake required for maintenance of body size is removed from the system as ecosystem-level starvation mortality. They have also incorporated temperature dependence in many of the ecosystem rates.

Reproduction is calculated from the energy intake.

Suitability for addressing the ToR is as for Benoit and Rochet (2004).

New models

Two new multispecies models have been developed. Because they not yet published in the literature, they are described here more fully than the previous models.

Andersen (2008): Size-based mass-balance model

This model is a size-based model with a detailed description of the energy budget of individual fish. Importantly, this means that growth is not prescribed, but depends on the availability of food. The model is a life-history model with the life-history variation being represented by groups of species with different asymptotic size. The model is therefore essentially a combination of the size-based life-history models (Hall *et al* (2006), Pope *et al* (2006)) and the mass-balance community models (Benoît and Rochet (2004); Maury *et al* (2007)). A steady-state version of the model is described in Andersen & Beyer (2006).

The model is calibrated to represent a generic ecosystem and can simulate the response of the ecosystem to changes in fishing pressure and changes in the primary productivity. The output of the model is the size spectrum as a function of individual size and of asymptotic size. It therefore predicts the number of fish with a given size and belonging to a given asymptotic size group (Fig. 4.2.1).

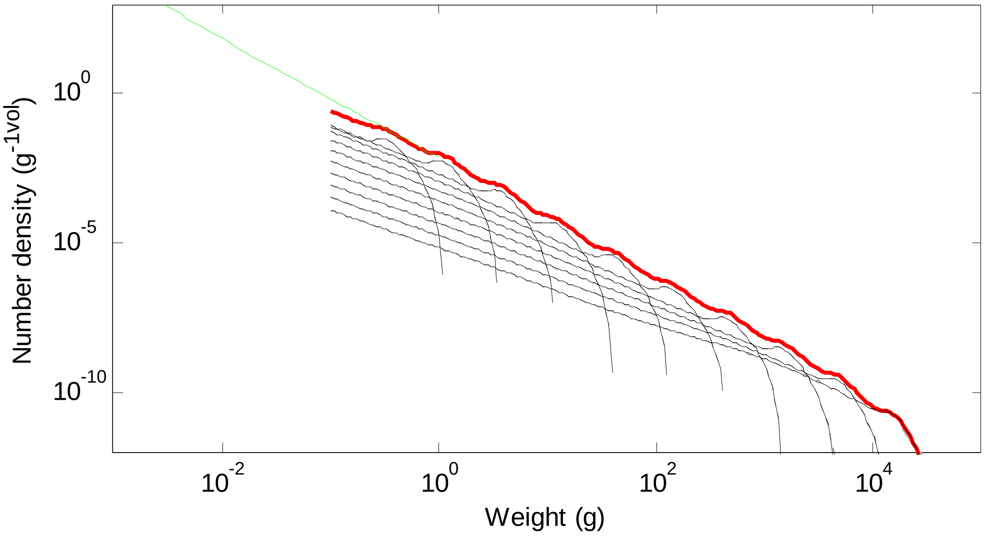


Figure 4.2.1: Output of a run with an ecosystem wide fishing level of F = 0.7 year^{-1}. The green line is the background spectrum representing zooplankton production, the black lines are spectra of species groups with different asymptotic sizes, and the red line is the total community spectrum. The "hump" in each spectrum just before the asymptotic size occurs around the size at maturation where growth slows down and individuals therefore pile up as they do not grow as fast out of a size group as they grow into it.

Growth. Encounter is modelled as in the community models. Individuals encounter prey in a given window in the size spectrum, described by a log-normal selection function with a fixed predator-prey mass ratio (150) and a fixed width (1.5). The amount of the encounter food which is eaten is described by a Holling type II functional response. Whereas the community models just assume that growth is proportional to the eaten food, here there is a detailed allocation for metabolism, growth and reproduction. Assimilated food is first used to fuel standard metabolism and the remainder is used for growth and reproduction. At constant food conditions growth curves will become similar to von-Bertalanffy growth curves, but in practice they will deviate somewhat, as they will depend on food conditions, i.e. on the state of the whole size spectrum.

Mortality. Mortality is composed of predation mortality, background mortality and fishing mortality. Predation mortality is calculated from the amount of food eaten by predators. Thus every fish eaten in the system is used to fuel growth of predators. Background mortality is typically low compared to predation mortality and effectively only acts on the largest fish which have no predators in the model. Fishing mortality can be prescribed on a size and asymptotic size basis. The specification used here is a direct adoption of the ecosystem wide fishing mortality by Pope *et al*

(2006). Here the fishing mortality on each species is a sigmoid function with an inflection point at a fraction of its asymptotic size.

Recruitment. Recruitment is fixed and the level is prescribed by the prediction for steady state by Andersen & Beyer (2006). This corresponds to the SSB being on the flat part of a Beverton-Holt recruitment curve. Following a simulation a check can be made if this is indeed the case. If it is not the case, the given species group is predicted to experience recruitment failure.

This model is dynamic, size-based and multispecies, and has a representation of fishing mortality. It models an abstract fish community not tied to any particular region but could be adapted for this purpose. It is not fitted to data

Speirs *et al.* (2008): Dynamic multi-species size-structured fish community model

A dynamic, multi-species, physiologically structured fish community model designed to investigate the impacts of changes in fishing pressure on fish diversity and community structure is being developed in collaboration between FRS and Dr Dougie Speirs and Prof Bill Gurney at the department of Statistics and Modelling Science at Strathclyde University, Glasgow, (http://www.stams.strath.ac.uk/).

The modelling structure is based on the "rhomboidal" approach suggested by deYoung *et al.* (2004): biological resolution and detail of representation is greatest for the focal species, where "focal" depends on the question being asked of the model, and increasing biological simplifications and reduction in resolution are made up and down the trophic scale away from the focal species.

For the purposes of investigating the impact of changes in fishing pressure on the North Sea demersal fish assemblage, the focal species are the dominant demersal species. Crucially, for the sort of questions the model is being developed to address, the representation of these key species is length-based. For computational convenience, the length distribution of each species from hatching to maximum length is divided into discrete length-classes.

Ancillary species are modelled dynamically but with less biological detail. Such species are defined as those acting as significant sources of predation of prey for the fish community and will include major pelagic predators and the benthos and zooplankton.

The inter-connected processes of growth, predator-prey interactions, mortality and reproduction are parameterised in the model as described below and aim to capture the major ecosystem processes shaping the fish community.

For key species, individuals are classified by length. For the purposes of modelling the effects of changes in fishing pressure on the North Sea fish assemblage, "key species" are those 10 to 20 species making up ~95% of the total biomass. For example, using the results of Fraser *et al.* (2007) to rank demersal species in order of contribution to the total biomass results in a list encompassing the following species: Norway pout, haddock, plaice, common dab, starry ray, poor cod, whiting, cod, grey gurnard, lemon sole, long-rough dab, ling and monk fish. The model structure is flexible to choice of species. The model keeps track of numbers (and biomass) of fish at length with time from larvae to maximum size for each species.

Growth. Appropriate hatching length L_H and asymptotic length L_∞ are determined from the literature. Growth is assumed to be well approximated by the von Bertalanffy formulation with species-specific growth rate γ.

Following previous authors (Gurney *et al.*, 2001), individuals are classified first in terms of a development index q that runs from 0 for newly hatched fish and tends to ∞ as fish approach their L_∞, with $\frac{dq}{dt} = \gamma$.

In order to capture the variability in size-at-age inherent in a natural population, the scheme of Gurney *et al.* 2007 is used: over a time-step of length dt, a proportion p of individuals are assumed to meet the food requirements for growth to the next length-class and (1-p) remain. By fixing the growth rate, the time-step and an appropriate coefficient of variance (c.v.) for size-at-age at a particular reference length, which is taken here as the mean length of maturity L_{mat}, the width dq of development classes and development proportion p can be determined. This gives von Bertalanffy growth with realistic variability and asymptotic length.

A fish at development stage q has a specific length L and weight W set according to the standard length-to-weight power law $W = aL^b$ with species-specific constants a and b. Additionally, the mean and s.d. of length at maturity and length of "settlement" are set for each species, giving transitions from larval to juvenile to adult stages so that maturity, mortality and diet vary with length.

Reproduction. Maturity in the model is a function of length. Each species is assumed to have a normally distributed spread in maturation lengths with species-dependent mean and s.d. From this, the proportion of individuals mature in each length-class is calculated and used to approximate the number of eggs produced by an individual over the time-step for each species and size-class so that the fecundity depends on proportion mature, ratio of females to males, the fecundity constant, which is number of eggs produced per female per gram and the time-step. The total egg production rate for each species is then calculated by summing up over all length classes. Over the time-step, eggs are subject to losses: some are fed upon by fish, some die naturally and others hatch. The egg development rate and natural mortality are assumed fixed and the predation rate is calculated from the egg and predator biomass as described below. The biomass of eggs remaining and the number hatched at the end of the time-step is calculated from these rates and new recruits enter into the smallest size-class.

Predation. The feeding rate of a fish of species in size class is calculated at the beginning of a time-step as a function of total available prey biomass according to the type II functional response and is broken down into uptake by predator species by size class from prey species by size class. Weightings are applied to different prey species, as will be described below. The maximum food uptake rate required for a fish in a size class to grow to the next size-class over the time-step depends on the weight, the weight change between size classes, the weight-specific basal metabolic cost and the assimilation efficiency. The actual food uptake rate is calculated depending on the food availability over the time-step. This gives the fraction of fish in (i,j) meeting their food requirements.

The calculation of uptake rate by predators according to the functional response depends on the prey biomass. In the model, total prey biomass is the sum of biomass of all suitable prey species weighted by a size-based "preference".

From North Sea stomach content studies published in the literature (e.g. Daan *et al.*, 1993; deGee & Kikkert, 1993) a diet matrix is defined. The list of diet items includes fish eggs as well as fish. A juvenile and an adult diet are specified and the fish move from one to the next when they reach a defined length of "settlement".

The feeding preference function assumes that some sizes of prey relative to the size of the predator are more suitable or desirable as food than others and will therefore be preferentially eaten. Studies have shown (Scharf *et al.*, 2000; Floeter *et al.*, 2005) that different fish species have different feeding tactics. For example, a large-mouthed species such as monk fish will preferentially consume relatively large prey compared with a smaller-mouthed species such as lemon sole. A function is therefore required that is flexible enough to reflect the size dependent aspect of feeding behaviour for different species. The preference is parameterised in terms of the ratio of predator to prey body length r = (predator length)/(prey length) and it is assumed that there exists a minimum r_{min}, maximum r_{max} and optimum r_{opt} suitable length ratio for each predator species. The preference is 0 for $r<r_{min}$ and $r>r_{max}$.

Using the diet matrix and preference weighting function, we integrate over all species to obtain the available prey biomass for each predator species and length class.

Mortality. As well as a possible starvation mortality calculated as above, fish suffer from a length-dependent "background" mortality μ; at the length of "settlement" (transition from larvae to juvenile fish) the mortality decreases in a step-change.

Finally, the model implements a species and size specific fishing mortality that can be varied to fit data and to explore the impacts of fishing on the community.

Unstructured resources. The zooplankton and benthos are important sources of food for the North Sea demersal fish and therefore must be included in any ecosystem model. As non-focal species, we are more interested in them as links in the food chain than in their dynamics; the aim is to capture the size composition and total biomass of these resources to provide the right amount of food of the right size for planktivorous and benthivorous fish.

The scheme is based on the empirical log-linear weight-biomass density spectrum (see e.g. Dickie & Kerr, 2001). For each resource, the size range of organisms (or, at least, the size range organisms likely to provide food for the fish) is determined to give a minimum and maximum length. Using the standard length-to-weight formulae, this gives a weight range which is divided up into n weight intervals. Each interval is treated as representing a species and the population size of each species is modelled dynamically. Individuals from each "species" do not "grow into" the next weight interval with time. In other words, the resource is represented as a community of dynamically growing unstructured (in the sense that growth of individuals is not modelled) populations.

Logistic growth of populations is assumed, modified by predation: each "species" is taken to have a characteristic weight, length, carrying capacity K and intrinsic growth rate r. K is set according to the empirical biomass-weight density spectrum with slope and intercept obtained from the literature (e.g. Jennings *et al.* (2001) for benthos). Similarly, the growth rate r is set according to the empirical relationship production to biomass spectrum. The fish predation on the unstructured resources is calculated from the resource biomass, fish biomass and functional response as outlined above.

The standard model output is a time series of the numbers and biomass at length of each species, showing how the length-distribution evolves with time. From this, the "proportion of large fish" EcoQO can be calculated. Additional output is a breakdown of the food intake into proportion from the benthos, zooplankton, fish and egg components. This aids in understanding the mechanisms controlling steady-state population sizes and length-distributions.

The model is size-based, dynamic and multispecies. It represents fishing mortality. It is North Sea specific and the authors intend to fit the model to North Sea data.

Table 4.1. Summary of the features of reviewed models with respect to the list of key attributes given in Section 3.2. Note: * fitting the model to data is work in progress, so "no" will be "yes" in the future.

	Hall *et al.*, 2006	**Pope *et al.*, 2006**	**Benoit & Rochet, 2004**	**Andersen, 2008**	**Speirs *et al.*, 2008**
Size-based	yes	yes	yes	yes	yes
Dynamic	yes	no	yes	yes	yes
Community-based	yes	yes	yes	yes	yes
Multi-species	yes	yes	no	yes	yes
Representation of fishing	yes	yes	yes	yes	yes
North Sea specific	no	no	no	no	yes
Fitted to data	no	no	no	no	no (*)
Number of parameters					

4.3 MSVPA: Review of WGSAM's work

The Working Group on Multispecies Assessment Methods (WGSAM) was tasked with running Multispecies Virtual Population Analysis (MSVPA) in forecast mode to evaluate the management action required to meet the fish community EcoQO target "*the proportion (by weight) for fish greater than 40cm in length should be 0.3*" taking multi-species interactions into account. WGSAM were asked to complete the same table given to WGFE for ToR C. Essentially therefore they were asked to assess the time required to reach the EcoQO target (±10%) under three fishing effort scenarios; at F_{pa} across the community and 15% either side of this.

MSVPA is an age-based model. To convert the model output to the length-based data required to support advice for management towards the length-based EcoQO, a set of transformations was applied. Each age group was given a mean weight in each quarter of the year. This mean weight at age was then converted to mean length at age by applying species specific length weight relationships. In effect this produced discrete length classes for each species, rather than a continuous numbers at length spectrum. This was disadvantageous as it tends to keep numbers of fish in length classes below the 40cm cusp which is the definition used in the North Sea Eco QO for a large fish.

After running the model for a 10 year period, 2006 to 2015, assuming mean weights in the sea and food rations averaged over the period 2000 to 2005, the predators and prey and the stock recruit relationships given in WGSAM's report (ICES, 2007c - tables 7.2 & 7.3), WGSAM concluded that the EcoQO could not be achieved within 10 years under either of the three fishing scenarios. However, WGFE note that the EcoQO metric was calculated across all fish species modelled in the MSVPA. In fact the metric should only relate to the demersal fish community, and so should have been calculated excluding herring and sandeels. Inclusion of these species, which never grow to exceed 40cm in length and which tend to be abundant in the whole fish community, must inevitably mean that the metric calculated in this way, is unlikely ever to reach the target of 0.3.

Examination of both the single species and multi species assessment outputs by WGFE and WGECO in 2007 (ICES, 2007a; 2007b) suggested that during the early

1980s, the reference period selected for the proportion of large fish (PLF) metric when the metric stood at around the current EcoQO target of 0.3, fishing mortality (F) averaged across the commercial species was substantially in excess of F_{PA}. This lead WGECO to believe that management action aimed at reducing F to F_{PA} across all the major commercial species would be sufficient to restore the PLF metric from its current level of approximately 0.15 to the target value of 0.3. WGSAM noted that few of the modelled species contribute to the percentage greater than 40cm in length. In fact their data suggest that only four of the MSVPA species actually grow to exceed 40cm in length; cod, saithe, haddock and plaice. Examination of the output provided by WGSAM suggested that for each of these species, current levels of fishing mortality are already lower than F_{PA}. Consequently the three fishing mortality scenarios suggested for WGSAM's ToR actually represented increases in fishing mortality above current levels. In future work therefore, it is clear that different fishing mortality scenarios should be investigated. WGSAM's work suggests that fishing mortality may have to be reduced to levels substantially lower than FPA if recovery in the PLF metric to its target level is to be achieved. They also noted that differences in fishing mortality of just $\pm 0.15F_{PA}$ were too small to have any noticeable effect on their results. In future, fishing mortality scenarios of F_{PA}, $0.8F_{PA}$, $0.6F_{PA}$ and $0.4F_{PA}$ should perhaps be tried. Furthermore, should the EcoQO target fail to be achieved at any of these fishing scenarios, the models should be used to determine what level of fishing mortality is compatible with the EcoQO target.

Despite the simulations representing increases in fishing mortality above the current relatively low levels for the four largest species modelled, WGFE note that for two of the species, cod and plaice, increases in spawning stock biomass still occurred. However, both haddock and saithe spawning stock biomass declined, and since haddock appear to be one of the most crucial species contributing to higher PLF metric values (because of their high relative abundance), this result was discouraging. For the PLF metric to show a strong increase, it seems obvious that the spawning stock biomass of all four of these species will have to increase: fishing mortality levels for these species will have to be set accordingly.

WGFE are grateful to WGSAM for the work they undertook; in particular for their adherence to the ToR despite the disappointing results. Although on first impression disappointing, their results provide considerable insight and will help to guide this work forward. Consequently WGFE acknowledge their offer to continue with this modelling work and undertake to work closely with WGSAM to see this work continued in the most effective manner possible.

4.4 Application to the North Sea EcoQO

Some preliminary experiments were carried out with the "generic" ecosystem model of Ken Andersen detailed above. The North Sea specific model of Speirs *et al.* is currently complete in structure but is undergoing rigorous validation and exploration of the ecosystem dynamics. Consequently, simulations will not be presented at this time.

The aim of the simulations with the model of Andersen is 1) to evaluate how the EcoQO indicator responds to ecosystem-wide changes in the fishing mortality and 2) to estimate the time-scale of the recovery of the ecosystem to changes in fishing mortality.

As the model is set up only for a "generic" ecosystem, we will not try to give a specific prediction for a specific ecosystem.

Method

For guidance, we use the basic setting with the growth rates scaled to give growth rates in order of magnitude of the North Sea ecosystem and the largest asymptotic size on 50 kg, corresponding to saithe in the North Sea. We stress that the results are general and not intended to give predictions specific to the North Sea demersal fish.

The model was run for 50 years with an ecosystem-wide fishing mortality on the order 0.7 year^{-1} using a scheme based on the work of Pope *et al.* (2006) (see above review). After that a steady-state was reached. The indicator was calculated as the ratio of biomass of fish larger than 40 cm to the biomass of fish larger than 10 cm. The fishing mortality was then changed either upwards or downwards for the whole ecosystem, and the development in the indicator EcoQO was calculated, i.e. a press perturbation experiment looking at the time taken to reach the new community steady-state.

Result

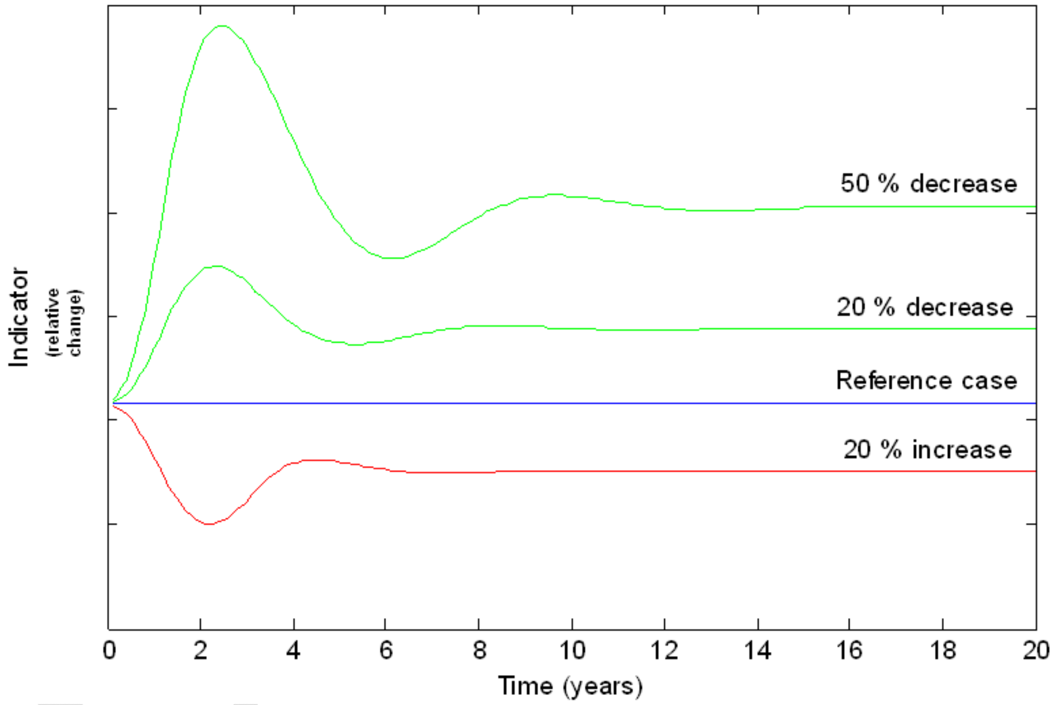


Figure 4.4.1: The development in the EcoQO indicator as a function of time after a change in the overall fishing pressure. The blue line is the reference case with a fishing pressure of 0.7 year^{-1}. The green lines are for decreases in the fishing pressure and the red line for an increase in the fishing pressure. See text for interpretation of the y-axis.

Overall the exercise confirms the validity of the EcoQO as an indicator of ecosystem-wide health and fishing pressure.

Conclusions of projecting recovery to the EcoQO

Comparing the value with the predictions from the North Sea IBTS survey should be done with care. First of all the model is sampling all fish in the community and not discriminating between demersal or pelagic fish. Secondly the IBTS has a largely unknown size-selectivity which to some degree is reflected in the value of the indicator, whereas the calculation of the indicator from the model is not subject to any size selectivity. Even considering these complications we believe that the relative changes in the indicator are comparable between the IBTS survey and the model.

The EcoQO indicator behaves as expected to a change in the fishing mortality; a decrease in fishing mortality leads to an increase in the biomass of large fish relative to small fish, and therefore to an increase in the indicator value, and vice versa. A 20 % decrease in fishing mortality leads to a 10 % increase in EcoQO and a 50 % decrease in the indicator leads to a 20 % increase.

The ecosystem does not immediately go to a new steady state following a change in the fishing mortality pattern. There is a transient that lasts from 5—10 years depending upon the magnitude of the change. More importantly there also appears an overshoot in the recovery, e.g. in the case of a 50 % decrease in fishing mortality the indicator increases by 50 % after 3 years, where after it again decreases to the new steady state. This is not a problem due the indicator per se, but rather a reflection of the complicated predator-prey interaction processes taking place within the ecosystem itself following the release of the fishing pressure.

Can these predictions be trusted? We believe that the overall pattern is fairly robust: the indicator increases with decreasing fishing pressure, there is a time delay of 5—10 years to a new steady state and there can be expected to be an “overshoot” before the final state is reached.

Future directions

Currently available models all represent an “abstract” fish community; none of the models reviewed from the literature are tuned to data from any particular region. It is not obvious how to specify the level of model detail that would be necessary for addressing the ToR or the extent to which the model would need to be fitted to data. However, it is clear that any model that could fully answer the ToR would have to be capable of making community level predictions for a specific location (that is, would have to be confronted with North Sea data) and would therefore inherently be an advance on the models currently available. The multispecies model of Speirs *et al.*, which is currently under development, is specific to the North Sea demersal fish community and will be fitted to data to North Sea data. Also, some of the other models reviewed here could be adapted to the North Sea and potentially fitted to data in order to examine their predictive capabilities. This represents a shift away from the current schematic modelling and towards the type of model required to address the Term of Reference.

4.5 References

Andersen, K. H. & Beyer, J. E. 2006. Asymptotic size determines species abundance in the marine size spectrum. The American Naturalist, 168: 54-61.

Benoît, E. & Rochet, M-J., 2004. A continuous model of biomass size spectra governed by predation and the effects of fishing on them. Journal of Theoretical Biology 226: 9-21.

Daan, N., Johnson, B., Larsen, J. R. & Sparholt, H. 1993. Analysis of the ray (Raja spec.) samples collected during the 1991 international stomach sampling project. International Council for the Exploration of the Sea. G(15).

deGee, A. & Kikkert, A. H. 1993. Analysis of the grey gurnard (Eutrigla gurnardus) samples collected during the 1991 International Stomach Sampling Project. International Council for the Exploration of the Sea. G(14).

deYoung, B., Heath, M., Werner, F., Chai, F., Megrey, B., & Monfray, P. 2004. Challenges of modelling ocean basin ecosystems. Science 304:1463-1466

Floeter, J., & Temming, A. 2005. Analysis of prey size preference of North Sea whiting, saithe and grey gurnard. ICES Journal of Marine Science, 62:897-907.

Fraser, H. M., Greenstreet, S. P. R. & G. J. Piet. 2007. Taking account of catchability in groundfish survey trawls: implications for estimating demersal fish biomass. ICES Journal of Marine Science, 64:1800-1819.

Gurney, W. S. C., Speirs, D. C., Wood, S. N., Clarke, E. D. & Heath, M. R. 2001. Simulating spatially and physiologically structured populations Journal of Animal Ecology 70: 881-894

Gurney, W., Tyldesley, G., Wood, S., Bacon P., Heath, M., Youngson A. & Ibbotson A. 2007. Modelling length at age variability under irreversible growth. Canadian Journal of Fisheries and Aquatic Science, 64:638-653

Hall, S. J., Collie, J. S., Duplisea, D. E., Jennings, S., Bravington, M. & Link, J. 2006. A length-based multispecies model for evaluating community responses to fishing. Canadian Journal of Fisheries and Aquatic Science, 63: 1344-1359.

ICES. 2007a. Report of the Working Group on Ecosystem Effects of Fishing Activities (WGECO). ICES CM2007/ACE:04, 162pp.

ICES. 2007b. Report of the Working Group on Fish Ecology (WGFE). ICES CM2007/LRC:03, 217pp.

ICES. 2007c. Report of the Working Group on Multispecies Assessment Methods (WGSAM). ICES CM2007/RCM:08, 132pp.

Kerr, S. R. & Dickie L. M. 2001.The Biomass Spectrum. Columbia University Press.

Maury, O. M., Faugeras, B., Shin, Y-J., Poggiale, J-C., Ari, T. B. & Marsac, F. 2007. Modeling environmental effects on size-structured energy flow through marine ecosystems. Part 1: The model. Progress in Oceanography. 74:479-499.

Pope, J. G., Rice, J. C., Daan, N., Jennings, S. & Gislason, H. 2006. Modelling an exploited marine fish community with 15 parameters – results from a simple size-based model. ICES Journal of Marine Science, 63: 1029-1044.

Scharf, F. S., Juanes, F., & Rountree, R. A. 2000. Predator size - prey size relationships of marine fish predators: interspecific variation and effects of ontogeny and body size on trophic-niche breadth. Marine Ecology Progress Series. 208:229-248.

Speirs, D., Gurney, W & Guirey, E. 2008. A dynamic multi-species size-structured model of the North Sea demersal fish community. *In progress.*

5 Utilisation of space by fish in relation to habitat, abundance, environment and exploitation

5.1 Introduction

Essential fish habitat

In recent years, WGFE has devoted a considerable effort in examining issues on Essential Fish Habitat (EFH), which is defined 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, reviewing the concept of essential fish habitat, 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 Benaka, 1999).

Habitat selection may be supported by the supposition such as the 'Ideal Free Distribution' theory (IFD) (Fretwell & Lucas, 1970), which suggests that areas of high resource value (e.g. in terms of food and shelter) attract a larger number of individuals than areas with fewer resources. Theory indicates that mobile fishes are most abundant in areas with the most suitable habitat characteristics for that species. The detection of areas of the seabed where fishes consistently aggregate could thus highlight areas that have a potentially important or 'essential' functional role (Hinz *et al*, 2003).

Although EFH may be characterized in general terms by the type of substrate, depth, and water characteristics (bottom temperature, salinity), in many cases the utility of these habitats for supporting fish populations relies on the presence of sessile habitat-generating organisms such as corals, sponges, kelp, rockweed, mussels, marine worms, etc. These organisms can have low rates of reproduction and be highly susceptible to habitat degradation, and hence the motivation for habitat protection.

In previous years, WGFE has analysed benthos-fish interactions, from both habitat-providing and trophic points of view, and including ToRs regarding fish habitat, concentrated on the study of habitat requirements of fish species. Those studies were focused on specific groups: gadiform and pleuronectiform fishes, deep-water fishes of Le Danois Bank, and the fish communities in Canadian waters and in the Barents Sea. In the last year report, the study on fish habitat was concentrated on the identification of nursery grounds, but also continuing with the study of the relationships between fishes and the physical and biotic environment.

Abundance-occupancy

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).

The study of abundance-occupancy relationships and essential core habitats of fish is important for a variety of reasons. If geographic range will contract as abundance declines, as in a species with a strong abundance occupancy relationship vulnerability to exploitation increases (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). Clearly, there are compelling reasons to clarify the nature of abundance-occupancy relationships within and over species and systems. The core habitat of fish is also essential to understand as it is to this habitat that a species is most likely to contract as its abundance decreases. Clearly observed utilisation of space by individuals and their core habitat will also be affected by the life stage of individuals considered in the analysis.

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).

Another important factor affecting observed spatial patterns could be fishing affects. For example an exploited fish may be depleted in certain locations for primarily socio-economic regions and the pattern remaining is largely because of the fishery rather than ecology. Furthermore, contraction to an area under these circumstances may not be to the most preferred area for a species so it would be an error to interpret it as such.

In 2007 WGFE found considerable overlap between its work on essential fish habitat and abundance-occupancy and benefits in combining this work as observed spatial patterns cannot be analysed independent of the mechanisms leading to them. Here we present three case studies relating how fish utilise space in relation to their abundance, suitable habitat area and fishing effects.

5.2 Habitat preferences of selected demersal fish species in the Bay of Biscay and Celtic Sea, North Atlantic

Data from the French annual bottom trawl survey taking place in the Bay of Biscay and Celtic Sea in autumn (Western IBTS survey) were used to assess associations between the distributions of eight demersal fish species, separately for juveniles and adults, and habitat characteristics (depth, temperature and salinity). The eight species were selected as they are well sampled by the survey gear and occur in sufficient numbers to allow quantitative analysis. Cumulative distribution functions were used (Perry and Smith 1994) to describe the general frequency distribution of environmental variables near the sea floor during the survey and their relationships with fish density. To assess the effect of abundance variations on the geographical distribution of fish, the spreading area (SA) index, adapted from Woillez *et al.* (2007), was calculated. It is an index of the area occupied by the species.

Temporal evolution of habitat conditions

Between 1992 and 2006, interannual variations of the bottom temperature and salinity were observed, especially in the 80 first meters, but with no clear time trend; see figure 4.7 and 4.8 page 138–39 in ICES 2007. In the depth range 81–200 m, median bottom temperature and salinity varied little between year; below 200 m, median temperature and salinity conditions remained constant.

Although colder, the Celtic Sea showed the same pattern with stronger interannual fluctuations occurring in the depth range 31–80 m.

Bottom temperature and salinity were both related to depth to some degree (Figure 5.1.1). In the first 100 m temperature decreased and salinity rose rapidly with increasing depth. Below 200 m temperature and salinity remained constant whatever the depth.

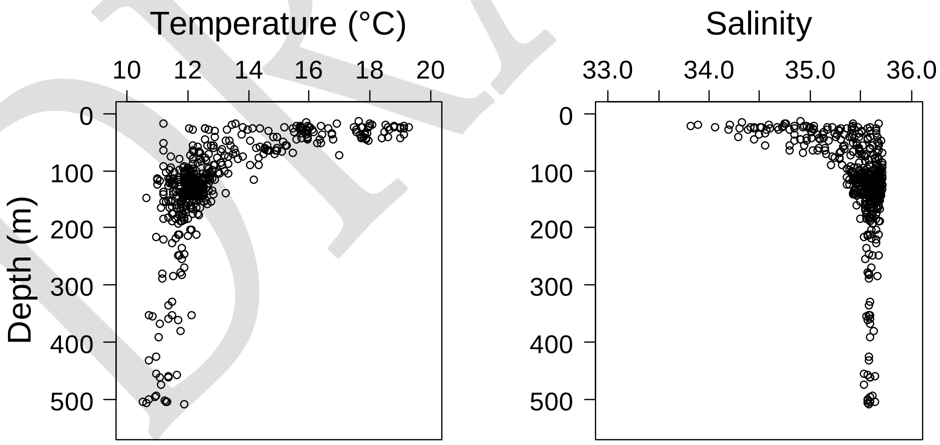


Figure 5.1.1: Relationship between bottom temperature and salinity and depth at the sampled stations in the Bay of Biscay.

Habitat associations

In the Bay of Biscay, four groups showed strong and persistent associations with specific depth conditions: juveniles of hake were associated with the range 60-180 m, juvenile and adult poor cod with 20-160 m and 20-200 m respectively and lesser-spotted dogfish with 20-335 m (Figure 5.2.2). Blue whiting, adult megrim and adult red gurnard were significantly associated with specific depth ranges, which were similar in the Bay of Biscay and the Celtic Sea (Table 5.2.1 & Table 5.2.2). The other studied species seemed to have no preference for particular depth conditions.

Associations with temperature and salinity were significant in fewer years than depth for most species (Figure 5.2.3 & Table 5.2.2). Note that even when there was a preferred temperature (in terms of a significant selection of a given temperature range) or salinity, for a given species this range differed between the Bay of Biscay and the Celtic Sea. Selected temperatures were generally colder in the Celtic Sea. Thus depth was found to be the major structuring factor, among those tested. Nevertheless, for blue whiting and red gurnard, selected depth ranges were very different between years.

Table 5.2.1. Associations between fish distribution and depth, temperature (T) and salinity (S) in the Bay of Biscay and the Celtic Sea in autumn. Ranges for environmental conditions correspond to conditions within which 95 % of individuals were found (average across years of 2.5 and 97.5 percentile points). J=juveniles (first length peak), A=adults (all other individuals). Species are ordered by preferred depth range.

		BAY OF BISCAY N=13			CELTIC SEA N=10		
Species	Group	Depth (m)	T (°C)	S	Depth (m)	T (°C)	S
Pouting	J	15-100	12-19.2	33.22-35.71			
	A	15-185	11.2-19.2	33.22-35.72			
Whiting	J	20-100	11.2-19	33.22-35-68	60-120	10-13.8	34.82-35.50
	A	15-100	11.2-19.1	33.22-35.70	55-120	10-13.8	34.76-35.41
Poor cod	J	20-160	11.1-18.5	33.82-35.71	60-180	10-13.8	34.25-35.61
	A	20-200	11-19.1	34.38-35.72	65-210	10-13.8	34.24-35.63
Lesser-spotted dogfish	all	20-335	11.2-16.3	34.79-35.71	55-260	9.7-13.8	34.21-35.63
Hake	J	60-180	10.6-17.6	34.79-35.72	65-200	10-13.8	34.15-35.59
Red gurnard	J	30-190	11.2-16	34.38-35.72	90-200	9.3-13.6	34.18-35.61
	A	55-190	11-16.3	34.93-35.72	90-260	10.2-13.6	34.24-35.69
	A	20-270	11-19.1	34.29-35.72	65-300	9.1-13.6	34.15-35.63
Megrim	A	90-425	10.7-13	35.28-35.72	75-425	9.3-13.2	34.16-35.63
Blue whiting	J	100-280	10.6-12.8	35.38-35.70	120-425	9.3-13.2	34.17-35.63
	A	100-500	10.6-13	35.42-35.71	90-330	9.7-13.2	34.21-35.63

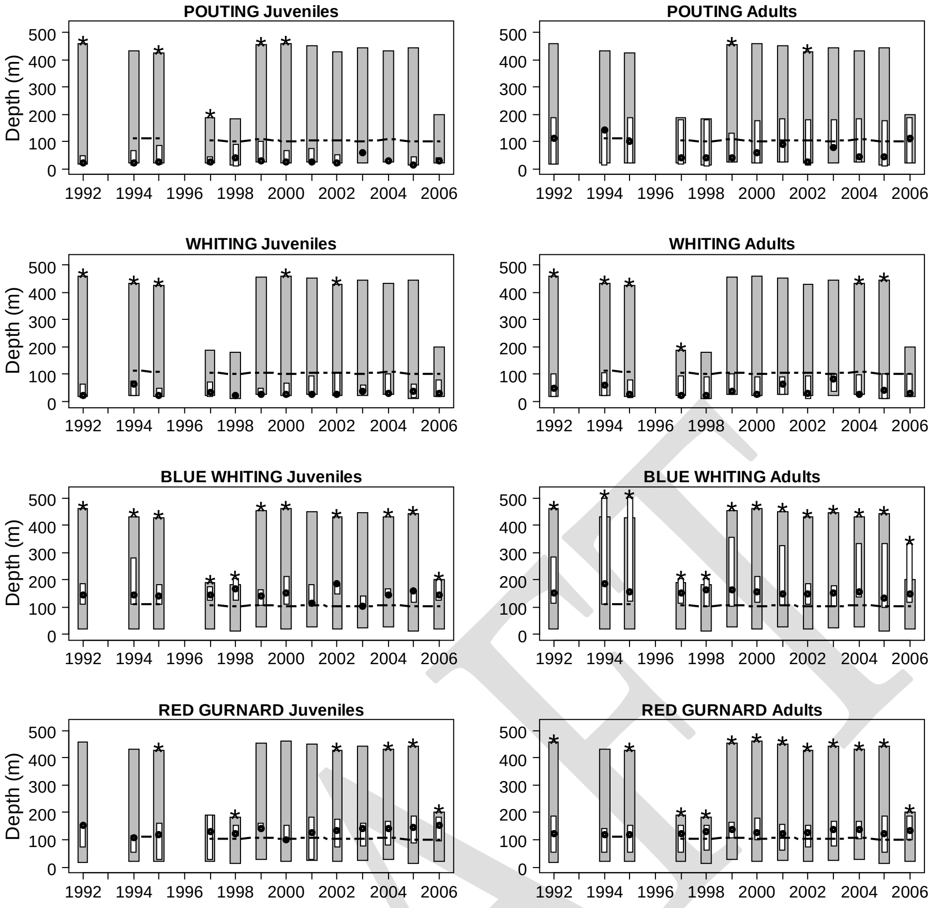


Figure 5.2.2: Associations between pouting, whiting, blue whiting and red gurnard and depth in the Bay of Biscay between 1992 and 2006. Grey bars represent 95 % of depths sampled (between the 2.5th and 97.5th percentiles), dashed lines show the median depth. White bars represent 95 % of the catches, points show the median catch. Stars indicate significant association between fish and depth (p-value <0.05).

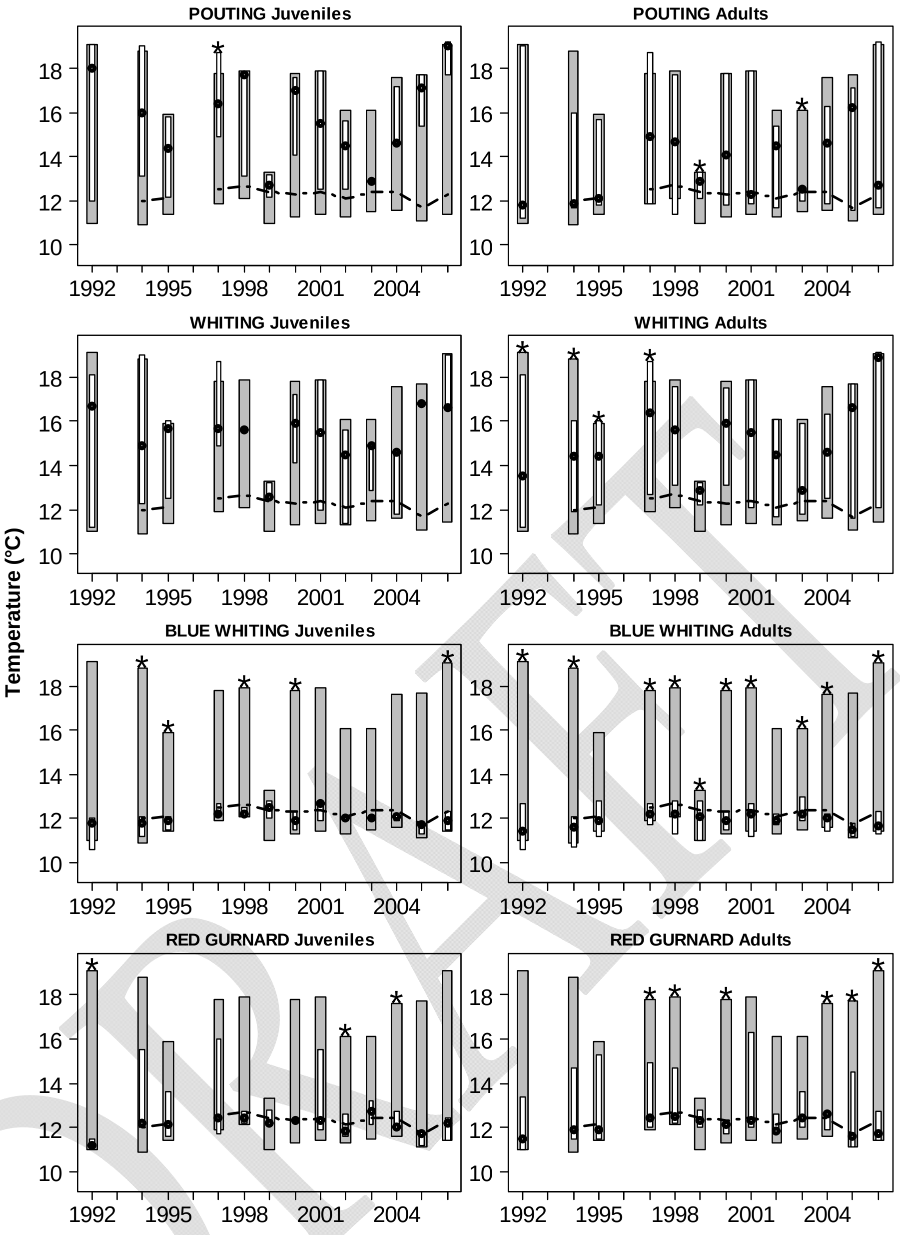


Figure 5.2.3: Associations between pouting, whiting, blue whiting and red gurnard and bottom temperature in the Bay of Biscay between 1992 and 2006. Grey bars represent 95 % of depths sampled (between the 2.5th and 97.5th percentiles), dashed lines show the median depth. White bars represent 95 % of the catches, points show the median catch. Stars indicate significant association between fish and bottom temperature (p-value <0.05).

Table 5.2.2. Significant associations between fish distribution and depth, temperature (T) and salinity (S) in the Bay of Biscay and the Celtic Sea in autumn based on cumulative distribution functions. Numbers represent the number of years for juveniles/adults when the null hypothesis of a random distribution of catches was rejected (p-value<0.05), i.e. the number of times the association between a species group and a specific habitat variable was significant. J juveniles; A other than juveniles.

	Bay of Biscay n=13			Celtic Sea n=10		
Species	D (m)	T (°C)	S	D (m)	T (°C)	S
Hake J/A	9 / 6	10 / 7	7 / 6	3 / 1	8 / 7	4 / 7
Pouting J/A	5 / 2	1 / 2	3 / 3			
Whiting J/A	5 / 6	0 / 4	2 / 6	5 / 3	0 / 1	5 / 5
Poor cod J/A	8 / 10	5 / 7	6 / 7	4 / 0	5 / 5	5 / 2
Blue whiting J/A	11 / 13	5 / 10	8 / 10	8 / 8	4 / 5	8 / 8
Red gurnard J/A	6 / 12	3 / 6	3 / 10	6 / 10	7 / 9	7 / 10
Megrim A	13	13	13	9	7	9
Lesser-spotted dogfish all	13	11	11	4	6	4

Abundance-distribution relationships

Strong abundance variations were observed for several species during the study period. However, these variations were linked to changes in area occupied only in a few cases. Adult red gurnard (whole area) and lesser-spotted dogfish (Bay of Biscay) showed a non-linear relationship between abundance and their occupied area (Figure 5.2.4). For these species, the area occupied increased with abundance, until abundances reached a certain level (2.5 million of individuals for red gurnard in the Bay of Biscay, 80 million in the Celtic Sea, and 15 million for lesser-spotted dogfish in the Bay of Biscay). At this level all suitable habitats seemed to be occupied and no further expansion possible. For blue whiting juveniles also, density-dependent effect seemed to occur, as the spreading area increased linearly with abundance, in both the Bay of Biscay and the Celtic Sea.

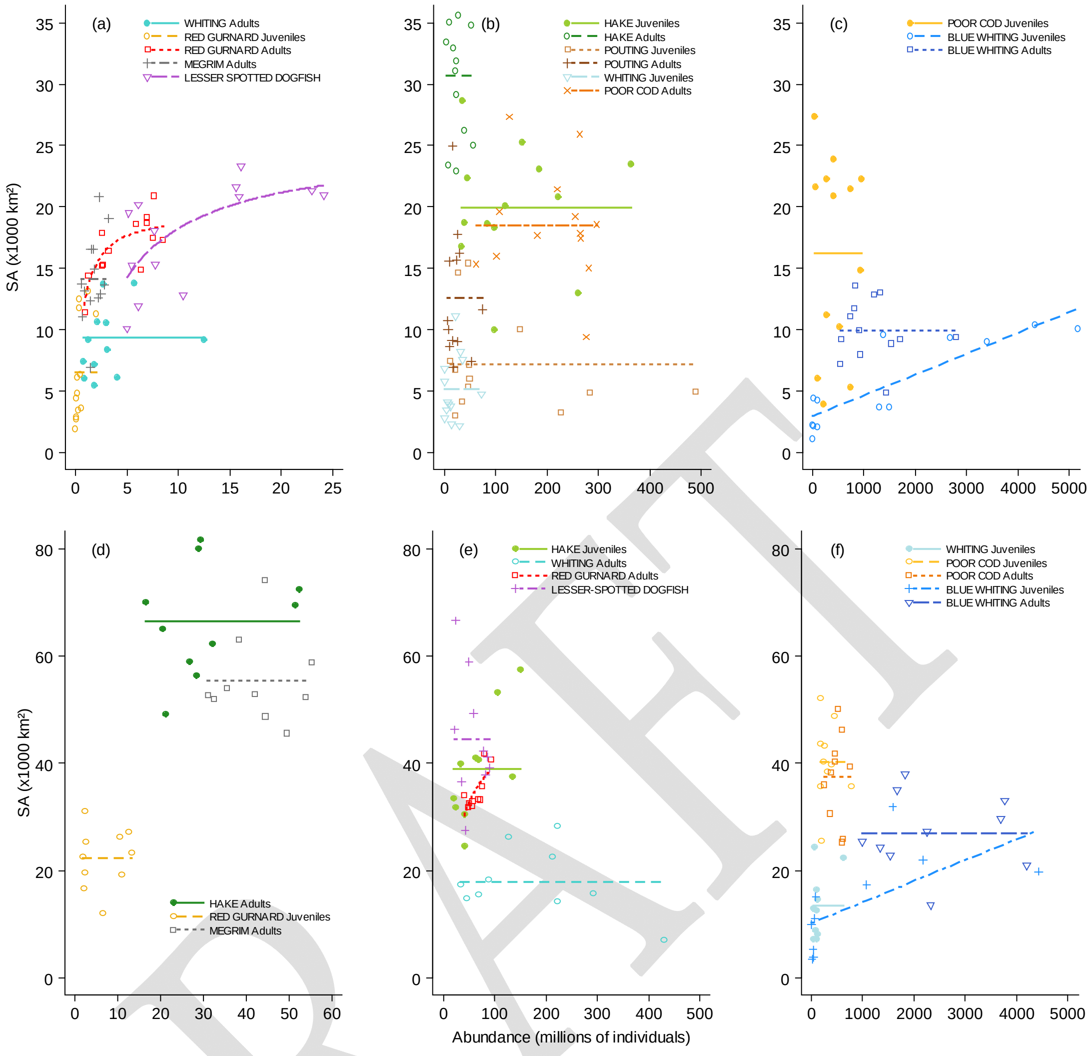


Figure 5.2.4: Relationships between abundances and the spatial spreading index in the Bay of Biscay (a to c) and in the Celtic Sea (d to f). A fitted logistic model was drawn if the b parameter in SA=a(abundance)/(b+abundance) was significantly different from 0, i.e. there was a significant curvature. A fitted linear model (SA=a(abundance)+b) was drawn when Pearson correlation coefficient was > to 0.75. Otherwise a horizontal straight line (mean(SA)=abundance) was drawn.

Relationships between available conditions, preferred conditions and abundance

Observed variations in the median of the bathymetric distribution (preferred conditions) were not linked to abundance fluctuations for any of the species. Furthermore, interannual variations in environmental conditions (available habitat) were not related to selected habitat conditions. This was established by separately ranking years in terms of median conditions in the environment and those selected by a species and calculating the correlation between the ranks of the two series.

In conclusion, local densities of juvenile hake, poor cod, blue whiting, adult red gurnard, adult megrim, and lesser-spotted dogfish were found to be significantly related to specific depth ranges in both the Bay of Biscay and Celtic Sea. Associations with bottom temperature and salinity were weaker and probably a consequence of selected depth ranges. Selected temperature ranges for the same species differed between the two regions. Associations with depth were not influenced by abundance variations but fluctuating abundances had an impact on spatial occupation for red gurnard adults, lesser-spotted dogfish and blue whiting juveniles. For some species, preferred environmental conditions changed over time, but these changes were not linked to variations in environmental conditions nor abundance. Thus apart from red gurnard adults, lesser-spotted dogfish and blue whiting juveniles there was little evidence for the studied species groups of any relationship between abundance and occupancy. Remember that the survey data were collected in autumn, which is outside the spawning period for all species apart from lesser-spotted dogfish that spawn all year round (Quéro and Vayne, 1997). Thus the conclusions apply to the feeding season during which intra-specific competition might plays less of a role. Alternatively current population abundances might be either too high or too low to observe spreading out with increasing abundance. Also, studies at other times of the year might well reveal stronger relationships between abundance and occupancy.

5.3 Recovery processes of Cantabrian Sea inner shelf fish assemblages after trawl exclusion

Introduction

Effects of trawl exclusion produced by artificial reefs on the Cantabrian Sea inner shelf fish community are described. The Cantabrian Sea shelf is very narrow and its benthic communities follow a patchy distribution as a consequence of depth and substrate type (Serrano *et al.*, 2006). On the inner shelf (off Llanes), rocky outcrops predominate, and only some finer sedimentary grounds are accessible to trawlers. As a consequence of the high biodiversity of coastal and inner shelf waters and their role as a nursery for several commercial species, bottom trawling in the Cantabrian Sea is forbidden by Spanish legislation in depths shallower than 100 m. Nevertheless, illegal trawling operations are common. To avoid this, artificial reefs (concrete blocks) were placed in 1992 by local fisheries authorities, in some of these shallow soft grounds. This measure allows us to analyse the recovery process of benthic and demersal communities in this inner shelf ecosystem. Data used in this case study were obtained from the IBTS autumn survey where one haul per year was performed since 1988 in the reef settlement area (43º28′N, 4º45′W) at a depth of 80–85 m.

These results may illustrate mechanisms that are controlling fish distribution in the area, mainly fishing pressure and habitat characteristics. Results obtained show the existence of two post-reefs periods (Fig. 5.3.1a). The first period, of 4 years, characterised by a significant increase in total biomass (Fig. 5.3.1b), can be defined as a “recovery phase”. After this period, biomass shows a new significant increase, defining a new period, a “consolidation” phase.

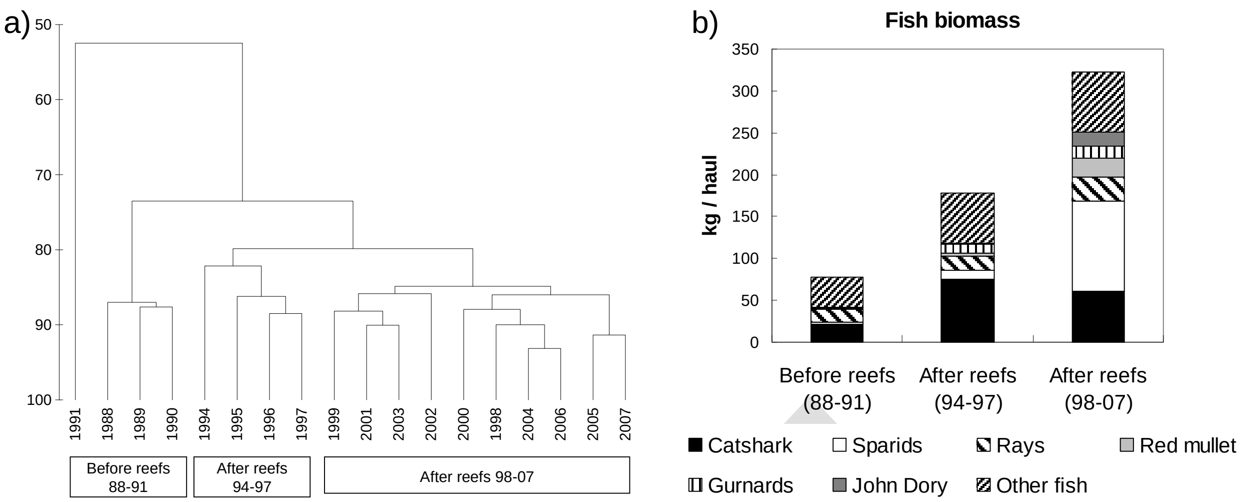


Figure 5.3.1: a) Periods resulting from interannual Bray-Curtis similarity analysis on log-transformed biomass matrix; b) Fish biomass and fish "species" composition in these periods.

Effects of trawl exclusion on species

The main species benefiting of the trawl exclusion (based both on biomass increases, Fig 5.3.1b, and SIMPER interperiod dissimilary results, not shown) are sea breams (Sparidae), elasmobranchs (dogfish and skates), red mullet and gurnards (Triglidae). On the contrary, the main commercial species (hake, anglerfish, megrim, etc) have shown a progressive decrease in abundance.

There are three patterns of response of species or species groups:

1) Progressive increase with time- slight increase from impact phase to recovery phase, but the most remarkable increase in biomass occurred between the recovery and consolidation phases. It is the pattern followed by the dominant group (sea breams). One sparid, *Pagellus bogaraveo*, practically disappeared from the soft grounds in the Cantabrian Sea, and did not appear in the Llanes reefs until the consolidation phase. This progressive increase also occurs in skates and red mullets. These species seem to need a well-structured habitat to develop their populations.

2) Increase in the recovery phase, but stabilization or slight decrease in the consolidation phase. This is the pattern followed by catshark and conger eel. These species prefer changing conditions of recovery phase. Habitat limitation may prevent further increases in some reef-associated species such as conger eel.

3) Species not affected by trawl exclusion.- the main species targeted in trawl fisheries (hake, anglerfish, megrim, etc) have shown a progressive decrease in abundance during the whole period considered (Table 5.3.1); decreases also shown by their respective stocks in the broader area outside the reef areas. The mobility of those species is high, and therefore they must not be considered as resident in the area.

Table 5.3.1. Mean biomass (kg/haul ± SD) of fish commercial species for the periods defined by cluster analysis. Empty cells mean absence of the species

	BEFORE REEFS (88-91)	AFTER REEFS (94-97)	AFTER REEFS (98-07)
Lepidorhombus spp	6.9 ± 4.2	6.9 ± 3.2	2.6 ± 1.4
Lophius spp	3.0 ± 4.2	6.7 ± 5.4	3.0 ±3.7
Merluccius merluccius	3.8 ± 2.5	3.8 ± 1.7	2.4 ± 1.3
Mullus surmuletus	0.0 ± 0.1	2.6 ± 2.8	23.0 ± 13.2
Pagellus acarne	1.8 ± 0.8	8.7 ± 6.8	72.7 ± 33.3
Pagellus bogaraveo			9.8 ± 13.8
Pagellus erithrynus		1.8 ± 1.7	5.1 ± 2.5
Trachurus trachurus	14.1 ± 23.7	3.5 ± 3.4	44.5 ± 32.4
Zeus faber	0.3 ± 0.5	1.9 ± 1.7	16.5 ± 16.7

The group 1 are constituted by characteristic inner shelf (stenobathyal) species, whereas group 2 and 3 are formed mainly by eurybathyal species. Therefore, species strongly related with the protected habitat conditions, and hence with low tendency to move to out of the closed areas presented a progressive increase. On the other hand, species that move out of the area (spillover effect) were limited in their biomass increase (group 2) or, in the case of commercial species, followed the same pattern as the wider stock (group 3). This however does not explain the evolution of catshark, since it is a philopatric species (Rodríguez-Cabello et al, 2007), with a high site fidelity. The recovery phase is characterised by changes in the community, by a species succession that is favourable to opportunistic and generalist predators, such as catsharks (Serrano et al, 2003). Later, when the habitat is fully structured, more specialized species (e.g. sparids) dominate, possibly controlling by competition the increase of the catshark population.

Effects of trawl exclusion on indicators and metrics

Several community metrics and indicators point towards a response by the inner shelf fish community to the trawling exclusion (Fig 5.3.2, Fig 5.3.3). Total biomass, and to a lesser extent species richness, were sensitive to the trawl exclusion status. The other metrics did not show this sensitivity, although no significant increases were found in all cases. Richness showed an increase in the recovery phase followed by a decrease in the consolidation phase, which is in agreement with the Intermediate Disturbance Hypothesis (IDH, Connell, 1978), under which increasing richness and diversity indices are expected for intermediate levels of disturbance, as a consequence of opportunistic competitive species.

Regarding the effects of trawl exclusion on the ratio of trophic guilds, a clear increase in the proportion of benthophagous fish was found (Fig 5.3.4). This increase implied a decrease of planktivorous species, and a relatively stable proportion of piscivorous fishes. The proportion of piscivores is expected to decrease under the impact of fishing, as piscivores are often the preferred targets of commercial fisheries (Trenkel & Rochet, 2003), but as has been described the main commercial species have followed the decreasing trend of their correspondent stocks. The species which have benefited from the trawl exclusion (sea breams, red mullet, catsharks) are benthivores.

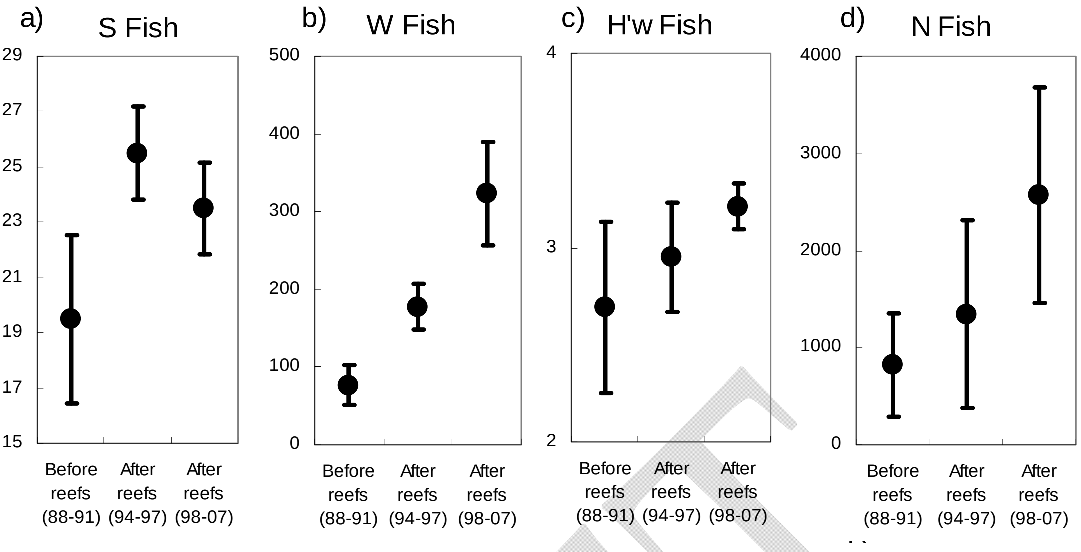


Figure 5.3.2: Shifts in community indices between periods. a) Species richness (S) of fish, b) fish biomass (W, kg/haul), c) Shannon diversity (H'w) in terms of fish biomass, d) density of fish (N, nº/haul). Mean values ± 95 % confidence levels.

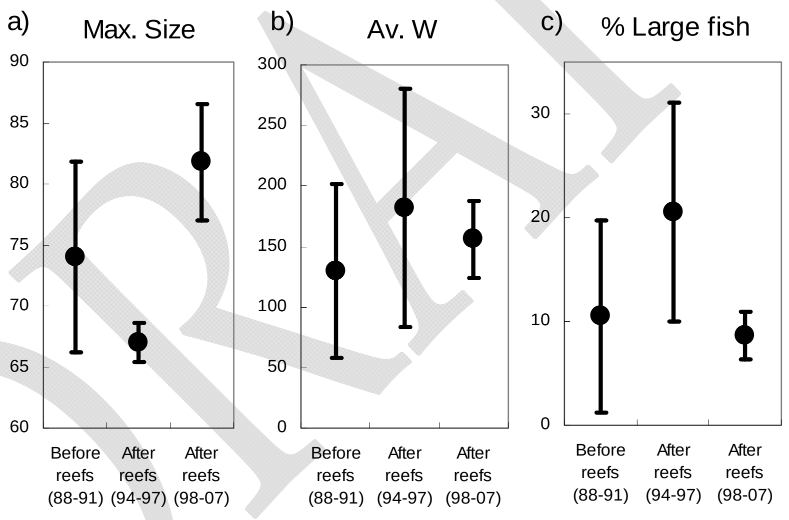


Figure 5.3.3: Shifts in size-based indicators between periods: a) average maximum size (cm), b) average weight (kg), c) percentage of fish larger than 40 cm (in biomass). Mean values ± 95 % confidence levels.

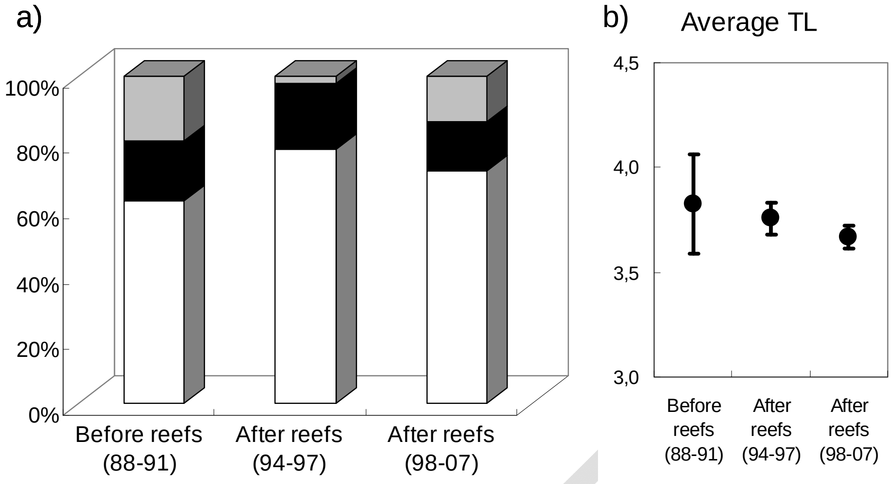


Figure 5.3.4: Shifts in trophic indicators between periods: a) percentage of trophic guilds (white: piscivores, black: benthivores, grey: planktivores), b) average trophic level values ± 95 % confidence (Trophic level data from Fishbase: Froese & Pauly, 2007)

Effects on habitats

The deployment of 270 concrete blocks in the area is not very significant in terms of surface area (2 %), but given the expected changes in habitat heterogeneity, boundary effects, and obvious settlement of hard ground species, it has to be taken in account. The increase in sparids in comparison with the more limited increase of other demersal species such as catsharks, could be explained by the higher affinity of sea breams to the new rocky habitats.

In addition, trawl exclusion has produced a noteworthy increase in invertebrate biomass, which could produce an increase in habitat complexity and trophic web complexity. Therefore, it trophic effects can be expected, since artificial reefs structures provide a hard substratum for the settlement of benthic prey, contributing to the creation of new feeding areas, and consequently increase trophic efficiency on formerly less productive, sandy seabeds (Bombace, 1989). This fact together with the increase of soft grounds invertebrates leads to an enhanced food availability at these reefs and hence to an increase in the biomass of reef fish assemblages (Leitao *et al.*, 2007).

Conclusions

Most of the Cantabrian Sea inner shelf fish species are strongly affected by fishing activities. The release of trawling pressure has lead to richer and more productive communities. Those effects are not immediate, a lapse of some years is necessary to establish a well-structured ecosystem. A differential response has been described among eurybathyal generalistic species and stenobathyal specialized species. The case of *Pagellus bogaraveo*, a species almost extinct in the area as a consequence of overexploitation, is interesting. This sea bream did not appear at the reefs area in the recovery phase, and only when the habitat was fully recovered did it re-appear in the area and present at a constant biomass until now.

Inclusion of hard substrates seem to have more influence in a trophic way, as an enhanced food resource is available to fish, than in an increase of species richness (all sampling stations were located in soft grounds).

5.4 Inter- and intra-specific abundance-occupancy relationships in the Georges Bank fish community

Introduction

Abundance-occupancy (A-O) relationships are most often analyzed for single species and interpreted in relation to fishing effects or changes in suitable habitat (ICES, 2007 and references therein). Less frequently A-O relationships are examined over many species (intraspecific) at the same time or within species groups or a large part of a community at the same time (interspecific) (Gaston *et al.*, 2000). Examining interspecific relationships for many species provides information on the community utilization of space that may be interpreted in relation to community or environmental metrics. The strength of such relationships over many species may provide insights into significant changes in the community that relate to factors that have affected the entire community (Webb *et al.*, 2007). Analyzed in this way, A-O relationships are analogous to other community indicators, such as size spectrum analysis or diversity measures.

Analyzing A-O relationships between phylogenetic groups elucidates common patterns in the relationship of density, abundance, and population range spread and contraction. If common phylogenic patterns in A-O relationships emerge, the propensity for hyper - aggregation and changes in catchability with abundance can be considered when weighing species risks of collapse under exploitation. For example, it may be possible to define an inflection point on the A-O curve to determine where species belonging to the group would be at high risk of large range contraction with further abundance reductions. If this were the case for a particular exploited species, it would raise a warning flag for management that further fishing on this species could severely contract its range size further and thus even further increase the population's vulnerability to exploitation. If this contraction in range size were accompanied by an increase in habitat fragmentation, then one might expect the A-O relationship itself to breakdown (Webb *et al.*, 2007).

Clearly, analyses over species groups could be an ecologically important community measure and have important implications for management of exploited stocks. In this case study, the A-O relationship for the Georges Bank fish community was examined for 37 species on Georges Bank over a 44 year time series. Additionally, we explore the relationship between fisheries exploitation and temperature on the slope and strength of the interspecific A-O relationship.

Methods

We analyzed data from the autumn National Marine Fisheries Service's (NMFS) bottom trawl survey from 1963-2006. The survey utilized a Yankee 36 bottom trawl equipped with a 1.27-cm mesh liner (Sosebee and Cadrin, 2006; Frisk *et al.*, 2008). Stations consist of tows 30 minutes in length at 1.95 m/s (Sosebee and Cadrin, 2006). During the autumn survey, 100-300 stations are sampled per year from the mid-Atlantic to Georges Bank and the Gulf of Maine. The analyses herein only cover's Georges Bank. The NMFS selects stations based on a depth-stratified random design (Sosebee and Cadrin, 2006; Frisk *et al.*, 2008). Only species that were captured at least 1000 times in the 1963-2006 Georges Bank autumn survey were included in the analyses.

Intraspecific Abundance-Occupancy relationships

Occupancy for each species was estimated as the percent of all stations with at least one individual present. Annual abundance was estimated as the average number of fish captured per station (catch per unit effort, CPUE). A-O relationships were estimated by fitting a logarithmic function (eq. 1) to untransformed data. In order to remove potential bias in A-O relationships, zeros were removed from the data set prior to estimation.

$$O = \alpha \cdot \log(A) - \beta \qquad \text{eq. 1}$$

Where O is occupancy = the proportion of tows in which the species was caught, A is the average abundance of species per tow, α is the fitted slope and β is the fitted intercept.

Interspecific Abundance-Occupancy relationships

We also fitted a multispecies A-O relationship to show how the community assemblage utilized space as a function of total abundance. Trends in the slope and strength of the assemblage A-O relationship can be viewed as a metric of assemblage spatial structure. To compare long-term changes in assemblage spatial structure, the slope and strength (r^2) of the A-O relationship was estimated from 1963–2006. Additionally, the total and average percent occupancy was calculated for each year.

The effects of harvest and temperature on Abundance-Occupancy relationships

The effects of harvest and average bottom temperature on the community A-O structures were explored by regression analysis. Effort and landings data were obtained for the bottom trawl fisheries on Georges Bank (Mayo *et al.*, 1992; Frisk *et al.*, in prep).

Results

Intraspecific Abundance-Occupancy relationships

A total of 37 species were captured 1000 or greater times during the autumn trawl NMFS bottom trawl survey 1963–2006 (Table 5.4.1). The average slope of the A-O relationship was 0.07 and ranged from 0.006 (Sevenspine Bay Shrimp) to 0.178 (American lobster) (Table 5.4.1). Only one species (Longhorn sculpin) has an r^2 value less then $r^2 = 0.2$ and the average was $r^2 = 0.47$. Combining all intraspecific A-O relationships indicates that the community follows a general pattern similar to patterns observed for individual species (Figure 5.4.1).

Table 5.4.1. Taxonomic order, fitted slope and strength (r^2) of the intraspecific A-O relationships for the George Bank finfish and shellfish community. A larger slope indicates a greater propensity to conserve local density as total abundance changes while the r^2 value indicates the quality of the relationship.

Order	Slope	r^2	Species
Squaliformes	0.07	0.47	SPINY DOGFISH
Clupeiformes	0.05	0.61	ALEWIFE
Clupeiformes	0.04	0.50	ATLANTIC HERRING
Shellfish and crustaceans	0.18	0.54	AMERICAN LOBSTER
Shellfish and crustaceans	0.07	0.57	ATLANTIC ROCK CRAB
Shellfish and crustaceans	0.02	0.54	LADY CRAB
Shellfish and crustaceans	0.06	0.71	SEA SCALLOP
Gadiformes	0.11	0.61	WHITE HAKE
Gadiformes	0.11	0.42	ATLANTIC COD
Gadiformes	0.07	0.30	HADDOCK
Gadiformes	0.01	0.39	LING UNCL
Gadiformes	0.08	0.40	RED HAKE
Gadiformes	0.06	0.24	SILVER HAKE
Gadiformes	0.02	0.22	POLLOCK
Ophidiiformes	0.02	0.60	FAWN CUSK-EEL
Perciformes	0.05	0.56	OCEAN POUT
Perciformes	0.03	0.66	ATLANTIC MACKEREL
Perciformes	0.04	0.72	NORTHERN SAND LANCE
Perciformes	0.08	0.46	BUTTERFISH
Pleuronectiformes	0.11	0.61	FOURSPOT FLOUNDER
Pleuronectiformes	0.03	0.43	GULF STREAM FLOUNDER
Pleuronectiformes	0.06	0.47	WINDOWPANE
Pleuronectiformes	0.08	0.47	WINTER FLOUNDER
Pleuronectiformes	0.08	0.41	AMERICAN PLAICE
Pleuronectiformes	0.03	0.50	WITCH FLOUNDER
Pleuronectiformes	0.09	0.73	YELLOWTAIL FLOUNDER
Rajiformes	0.13	0.31	LITTLE SKATE
Rajiformes	0.10	0.61	THORNY SKATE
Rajiformes	0.09	0.30	WINTER SKATE
Scorpaeniformes	0.02	0.31	ACADIAN REDFISH
Scorpaeniformes	0.02	0.26	BLACKBELLY ROSEFISH
Scorpaeniformes	0.05	0.09	LONGHORN SCULPIN
Scorpaeniformes	0.02	0.42	MOUSTACHE SCULPIN
Scorpaeniformes	0.14	0.31	SEA RAVEN
Shrimp	0.01	0.41	SEVENSPINE BAY SHRIMP
Shrimp	0.01	0.03	SHRIMP UNCL
Squid	0.08	0.61	LONGFIN SQUID
Squid	0.14	0.84	NORTHERN SHORTFIN SQUID

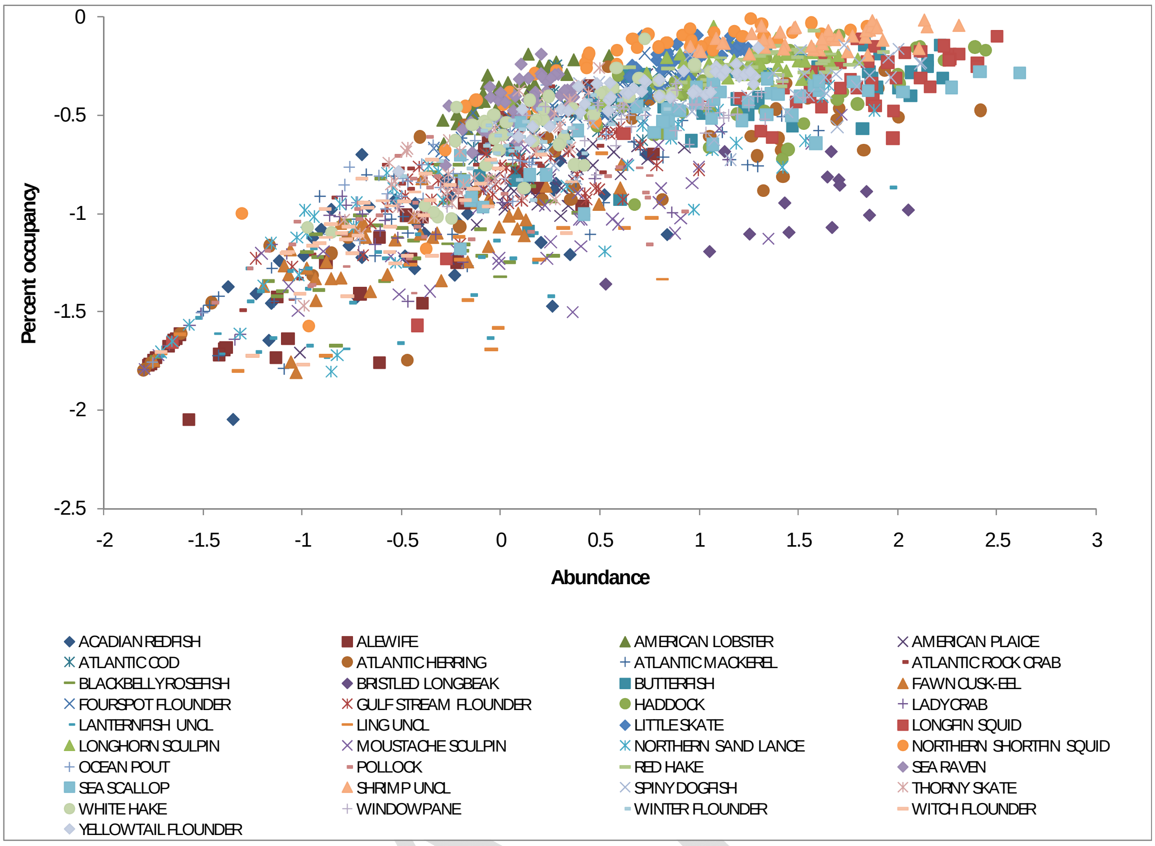


Figure 5.4.1: Intraspecific percent occupancy is regressed against abundance for 37 species on Georges Bank including species from the following groups: 27 finfish, 4 elasmobranchs, 2 crabs, 2 squids, 1 shrimp and 1 scallop. Log transformation was performed on percent occupancy and abundance.

Four orders of fish contained more than three species allowing for phylogenic comparisons including: Gadiformes, (7 species); Perciformes, (11); Rajiformes, (3); and Scorpaeniformes, (5). Additionally, crabs and shellfish were placed in a single group. No significant differences in the slope of intraspecific relationships were found between orders (Fig. 5.4.2). Additionally, the strength (r^2) of the intraspecific relationship did not differ between orders.

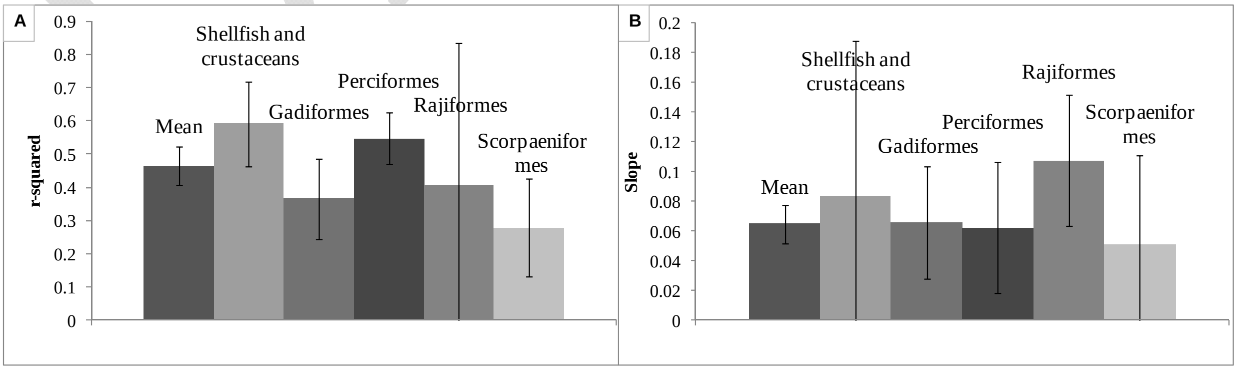


Figure 5.4.2: Panel A displays the average strength of the intraspecific A-O relationships for 5 phylogenic groups. Panel B displays the average slope of the interspecific A-O for 5 phylogenic groups.

Interspecific Abundance-Occupancy relationships

The slope of the interspecific A-O relationship significantly declined from 1963 to 2006 (Slope = -0.0007●Abundance + 1.53, $r^2 = 0.48$, $p < 0.001$) (Figure 5.4.3A). However, during 1963 to 1976 the rate of decline was steeper then rates observed in more recent years (Slope = -0.0025●Abundance + 4.94, $r^2 = 0.49$, $p = 0.005$). After 1976, the rate of decline decreased displaying a significant but weaker trend (Slope = 0.0004●Abundance + 0.923, $r^2 = 0.18$, $p = 0.017$). The strength of the interspecific A-O relationship averaged $r^2 = 0.3$ from 1963-2006. A three phase pattern is evident in the strength of the relationship. The first phase is characterized by a non-significant relatively stable trend from 1963 to 1983 (r^2 = -0.0005●year + 1.53, $r^2 = 0.00$, $p = 0.86$). The second phase is characterized by a significant rapid decline in the strength of the A-O relationship from 1984 to 1993 (r^2 = -0.0285●year + 57.03, $r^2 = 0.89$, $p = 0.00$). The third phase was non-significant and suggests a continued declining trend at a lower rate than phase two (Slope = -0.0072●Abudance + 14.98, $r^2 = 0.15$, $p = 0.23$).

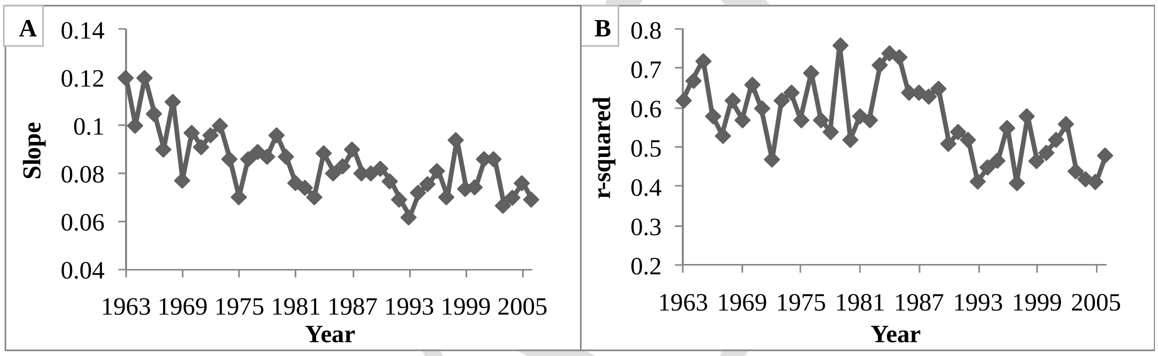


Figure 5.4.3: Panel A displays the slope of the interspecific A-O relationships regressed over years for the fall NMFS bottom trawl survey from 1963-2006. Panel B displays the strength (r^2) of the interspecific A-O relationships regressed over years for the fall NMFS bottom trawl survey from 1963-2006.

Total and average percent occupancy

Total percent occupancy showed a slight increase over the time series (Total % occupancy = 0.0341●year – 56.83, $r^2 = 0.19$, $p = 0.003$) (Figure 5.4.4A). The average percent occupancy has remained relatively constant from 1963-2006 (Average % occupancy = -0.0002●year – 0.653, $r^2 = 0.00$, $p = 0.60$) (Figure 5.4.4.B).

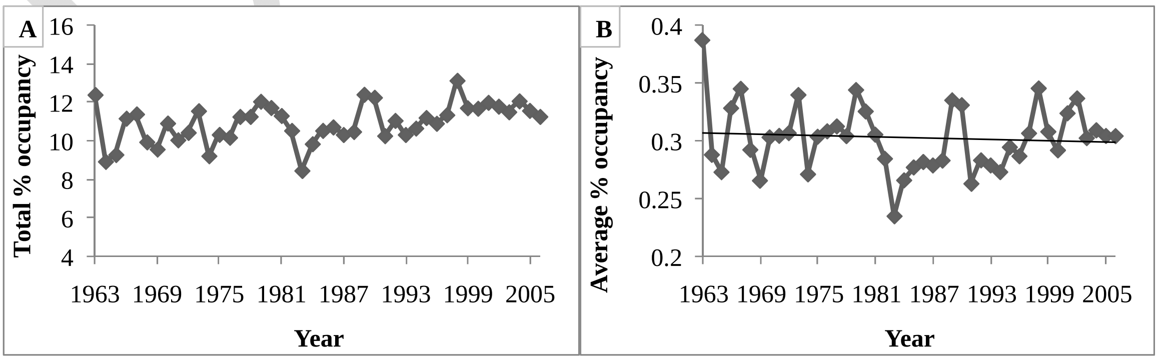


Figure 5.4.4: Panel A displays the total percent occupancy for 37 species on Georges Bank for the autumn NMFS bottom trawl survey. Panel B displays the average percent occupancy for 37 species on Georges Bank for the autumn NMFS bottom trawl survey.

The effects of harvest and temperature on A-O relationships

Fishing effort (1963-1998) and total groundfish landings (1963-1992) (TGL) were both significantly related to the slope of the interspecific A-O relationship (Slope = 0.000001●TGL – 0.0559, $r^2 = 0.39$, $p = <0.0001$; Slope = -.00001●effort – .073, $r^2 = 0.23$, $p = 0.004$).

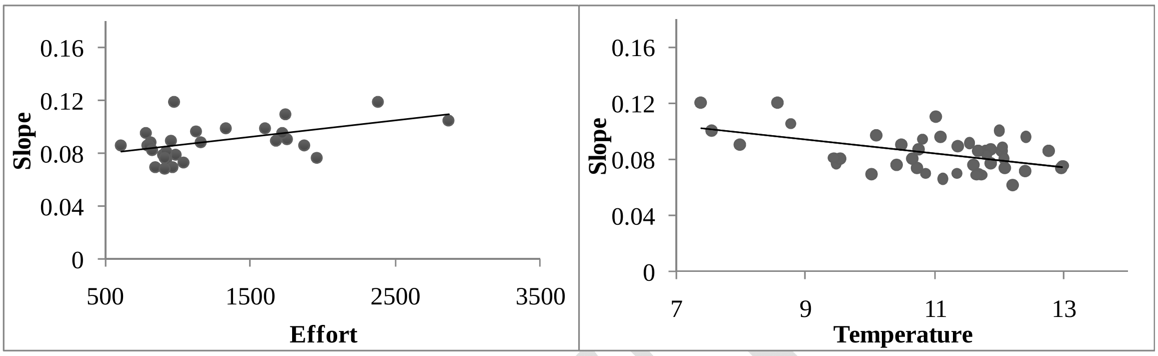


Figure 5.4.5: Panel A displays the slope of the interspecific A-O relationship regressed over fishing effort for data from 1963-1998. Effort data was measured in 1000 days fished per year (Mayo *et al.*, 1992). Panel B displays the slope of the interspecific A-O relationship regressed over average bottom temperature from 1963-2006.

Discussion

Our results found strong intraspecific A-O relationships, with only 2 species with r^2 values below 0.2 for 37 species on Georges Bank. While the models and analyses differ, previous studies of marine systems have not shown as strong relationships (Fisher and Frank, 2004). Fisher and Frank analyzed the neighbouring ecosystems of the Scotian Shelf and Bay of Fundy and found 16 or 32 species displaying significant relationships and 13 with Pearson correlations greater than 0.45 (roughly equivalent to an r^2 value of 0.2). Differences between regions may be due to macro ecological and community structure differences or spatial and temporal survey coverage.

The more interesting results were found when comparing interspecific A-O relationships. Webb *et al.* (2007) found that for birds increasing habitat fragmentation led to a decrease in the strength of the A-O relationship. They argued the weakening A-O relationship resulted from a breakdown of ecosystem structure over time (Webb *et al.*, 2007). On Georges Bank, both the slope and strength of the interspecific A-O relationship significantly declined from 1963 to 2006. We tested two potential mechanisms underlying this trend and found that annual harvest by the groundfish fishery and average bottom temperature were both significantly related to trends in the slope and strength of the A-O relationship. However, we have additional work to do to further define possible mechanisms underlying changes in the A-O relationship on Georges Bank. Community shifts, exploitation and temperature could confound fitted A-O relationships. Future work will aim at untangling possible mechanisms behind A-O relationships and also the influence of the interaction of reproductive behaviour and season. The most important result of the present analysis is showing an apparent breakdown of the interspecific A-O relationship on Georges Bank. Currently, we are working towards understanding the causes of this result and its implications for management.

5.5 References

Bombace, G. 1989. Artificial reefs in the Mediterranean Sea. *Bulletin of Marine Science*, 44: 1023–1032

Connell, J.H. 1978. Diversity in tropical rain forests and coral reefs. *Science*, 199: 1302-1310

Corten, A. 2002. The role of conservatism in herring migrations. Reviews in Fish Biology and Fisheries, 11: 339–361.

Fisher, J. A. D., and K. T. Frank. 2004. Abundance-distribution relationships and conservation of exploited marine fishes. Marine Ecology-Progress Series 279:201-213.

Freckleton, R. P., D. Noble, and T. J. Webb. 2006. Distributions of habitat suitability and the abundance-occupancy relationship. American Naturalist 167:260-275.

Fretwell, S. D. and Lucas, H. L. 1970. On territorial behavior and other factors influencing habitat distribution in birds. I. Theoretical development. Acta Biotheor. 19: 16–36.

Frisk, M. G., T. J. Miller, S. J. D. Martell, and K. Sosebee. 2008. New hypothesis helps explain elasmobranch "OutBurst" on Georges bank in the 1980s. Ecological Applications 18:234-245.

Frisk, M.G., S.J.D. Martell, T.J. Miller, and K. Sosebee (in prep). Exploring the population dynamics of winter skate in the Georges Bank region: A length-based statistical-catch-at-age model. To be submitted to Canadian Journal of Fisheries and Aquatic Sciences.

Froese, R. and D. Pauly (eds). 2007. *FishBase*. World Wide Web electronic publication. www.fishbase.org, version (10/2007)

Gaston, K. J., T. M. Blackburn, J. J. D. Greenwood, R. D. Gregory, R. M. Quinn, and J. H. Lawton. 2000. Abundance-occupancy relationships. Journal of Applied Ecology 37:39-59.

Huse, G., Railsback, S. and Fernö, A. 2002. Modelling changes in migration pattern of herring: collective behaviour and numerical dominance. Journal of Fish Biology, 60: 571–582.

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

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

Leitao, F., M.N. Santos and C.C. Monteiro. 2007. Contribution of artificial reefs to the diet of the white sea bream (Diplodus sargus). *ICES Journal of Marine Science*, 64: 473-478

MacCall, A. D. 1990. Dynamic geography of marine fish populations. University of Washington Press, Seattle, Washington. 153 p.

Mayo, R.K., M.J. Fogarty and F.M. Serchuk. 1992. Aggregate fish biomass and yield on Georges Bank, 1963-87.

McQuinn, I. 1997. Metapopulations and the Atlantic herring. Reviews in Fish Biology and Fisheries, 7: 297–329.

Morris, D. W. 1987. Tests of density-dependent habitat selection in a patchy environment. Ecol. Mono. 57: 269–281.

Paloheimo, J. E., and Dickie, L.M. 1964. Abundance and fishing success. Rapp. P.-V. Reun. Cons. Int. Explor. Mer. 155: 152–163.

Perry, R.I. and Smith, S.J. 1994. Identifying Habitat Associations of Marine Fishes Using Survey Data: An Application to the Northwest Atlantic. *Can. J. Fish. Aquat. Sci.* **51:** 589-602.

Petitgas, P., Reid, D. *et al.* 2006. The entrainment hypothesis: an explanation for the persistence and innovation in spawning migrations and life cycle spatial patterns. ICES CM 2006/B:07.

Quéro, J.C. and Vayne, J.J. 1997. *Marine fish of French fisheries*. Lausanne: Delachaux & Niestlé SA, 304 pp. (In French).

Rodríguez-Cabello, C. and Sánchez, F. and Olaso, I. 2007. Distribution patterns and sexual segregations of *Scyliorhinus canicula* (L.) in the Cantabrian Sea. *Journal of Fish Biology*, 70, 5: 1568--1586

Serrano, A., Sánchez, F. and García-Castrillo, G. 2006a. Epibenthic communities of trawlable grounds of the Cantabrian Sea. *Scientia Marina*, 70S1, 149-159

Serrano, A.; Velasco, F.; Olaso, I. and Sánchez, F. 2003. Macrobenthic crustaceans in the diet of demersal fish in the Bay of Biscay in relation to abundance in the environment. *Sarsia*, 88: 36-48

Sosebee, K. A., and S. X. Cadrin. 2006. Historical Perspectives on the Abundance and Biomass of Some Northeast Demersal Finfish Stocks from NMFS and Massachusetts Inshore Bottom Trawl Surveys, 1963-2002. NOAA/NMFS/NEFSC, Woods Hole, MA

Trenkel, V.M. and Rochet, M.J. 2003. Performance of indicators derived from abundance estimates for detecting the impact of fishing on a fish community. Canad. Jour. Fish. Aquat. Science, 60, 67-85

van der Lingen, C., Castro, L., Drapeau, L., and Checkley, D. (Eds.). 2005. Report of the GLOBEC-SPACC Workshop on Characterising and Comparing the Spawning Habitats of Small Pelagic Fish. GLOBEC report No. 21, 175pp.

Webb, T. J., D. Noble, and R. P. Freckleton. 2007. Abundance-occupancy dynamics in a human dominated environment: linking interspecific and intraspecific trends in British farmland and woodland birds. Journal of Animal Ecology 76:123-134.

Webb, T. J., Noble, D., and Freckleton, R. P. 2007. Abundance-occupancy dynamics in a human dominated environment: linking interspecific and intraspecific trends in British farmland and woodland birds. Journal of Animal Ecology, 76: 123–134.

6 Metrics for measuring the impacts of fishing on structure and function of fish communities

6.1 Introduction

One of the core activities of WGFE since its inception in 2003 has been to develop and explore the utility of fish community state indicators especially for their abilities to uncover the impacts of fishing on the total fish community. In the ICES community and elsewhere, the call for an ecosystem approach to fisheries (EAF) is growing louder which translates to the need for science to articulate what this approach should consist of operationally. It is often suggested that a good set of community indicators that reflect certain key properties of fish communities could be important for establishing EAF (Cury and Christensen 2005). Consequently, WGFE maintains a broad term of reference each year which is designed to encourage further work on developing new indicators of community state and comparing and testing established indicators. In some years much of WGFE's work consists of indicators studies (e.g. ICES 2004), while it is a smaller component of WGFE's work at other times.

Presently, this aspect of WGFE's work consists of one case study which examines the redundancy of various kinds of fish community indicators for the North Sea demersal fish community using survey data.

6.2 Redundancy in fish community metrics: a case study of the North Sea EcoQO on the proportion of large fish

ToR E: Evaluate metrics to characterise, monitor and detect changes in the structure, function and productivity of fish communities.

The EcoQO for the North Sea fish community currently focuses on the community's size structure. The chosen metric is the proportion (by weight) of demersal fish larger than 40cm (pelagic species are not currently considered in this EcoQO) and the objective for this metric for the North Sea has been set at 0.3 (ICES, 2007). Despite the EcoQO focus on fish size, other aspects of the fish community, and of the broader North Sea ecosystem, are emphasised heavily in most policy documents underpinning the development of an ecosystem approach to management. For example, the recent EU Marine Strategy Directive stresses the aim of "providing biologically diverse and dynamic oceans and seas that are safe, clean, healthy and productive", sentiments similar to those stated in Annex V of the Convention for the Protection of the Marine Environment of the North-East Atlantic and in the Convention on Biological Diversity. This then begs the question as to whether management directed specifically towards restoring the size structure of the North Sea demersal fish community will also achieve beneficial changes in respect of other characteristics of the community.

Recent studies have applied numerous different metrics to North Sea groundfish survey data; including metrics of size structure, metrics of species richness and species diversity, metrics of trophic structure and metrics conveying information on average community life history composition (e.g. Greenstreet *et al*. 1999; Jennings *et al.* 1999; Jennings *et al.* 2002; Greenstreet & Rogers 2006). It has often been said that a suite of such metrics should be used to cover all aspects of the structure and composition of natural communities that may be important ecologically, and therefore worthy of management consideration (EU Commission 2005; 2006). But implicit in such an approach is the belief that these different aspects of natural communities vary independently of one another. An alternative proposition is that these different attributes of natural communities may in fact all be linked, thus as an over-exploited fish community becomes increasingly dominated by smaller sized fish, its species richness and diversity, trophic structure and life-history trait composition might also vary in a predictable manner. Under such circumstances, the need for a suite of indices would be much reduced, since trends in any one community attribute would suggest the likely trends in other attributes, leading to redundancy among the suite of potential community metrics.

The redundancy among the suite of potential community metrics was clearly inferred from analysis on Scottish August Groundfish Survey (SAGFS) data collected in the northwestern North Sea, which examined fishing effects on fish community structure indicated by a suite of 12 different metrics (Greenstreet & Rogers 2006). However, whilst the SAGFS remains one of the longest continuous data sets in existence, enabling long-term trends in community structure and composition to be evaluated, this data set was in effect terminated in 1997 when the research vessel was changed, and simultaneously the trawl gear and trawl duration in the survey were altered. Thus while this data set can be used to assess anthropogenic impacts on the community to determine possible reference levels and objectives for different metrics, it will be necessary to analyse alternative data sets to monitor progress towards such objectives. The need for such an approach was explicitly recognised by WGECO in its formulation of the EcoQO for the proportion of large fish in the North Sea, whereby the objective value of 0.3 was determined specifically with regard to the Q1 International Bottom Trawl Survey (IBTS) (ICES, 2007). Analysis of the Q1 IBTS data revealed a marked reduction in the "proportion of large fish" metric, from the reference period in the early 1980s to its lowest point in 2001. But no corresponding systematic analysis of other community metrics calculated on the Q1 IBTS data has yet been undertaken. It is not known therefore how other attributes of the demersal fish community have changed over the corresponding time period. Without such an analysis, it is not possible to even begin to speculate how these other attributes of the fish community might change in response to management actions designed to bring about a recovery on the proportion of large fish. Given the political importance placed on biodiversity, it is certainly worth knowing that recovering large fishing should also help to enhance biodiversity among North Sea demersal fish.

In this section we examine the level of redundancy among ten community metrics calculated on Q1 IBTS data. The extent to which the different metrics show similar trends is examined. We start by calculating the two size-based metrics suggested in the Bergen (2002) element of ecological quality for the North Sea fish community; the proportion of large fish in the community, defined following ICES (2007) as the proportion by weight of fish greater than 40cm in length, and the mean weight of fish in the community. This latter metric was dropped from the EcoQO because of its sensitivity to recruitment events mainly associated with environmental rather than anthropogenic influences. However, the metric still performs an important sentinel role providing early warning of impending environmental influences on the proportion of large fish metric. Three species richness and species diversity metrics are calculated. Species richness is simply the count of species obtained from all IBTS samples collected in each year. Hill's N_1 and N_2 indices provide measures of species diversity that take into account the evenness of species abundances within the IBTS samples; the former is more strongly influenced by the abundances of rare species, while the latter is very much a measure of dominance with the samples. Three life-history trait indicators were calculated, the average ultimate body length and growth rate (L_∞ and k in the von Bertalanffy growth equation) and the average length at maturity. Two simple abundance measures were also determined, the numbers and biomass per hour of survey effort in each year.

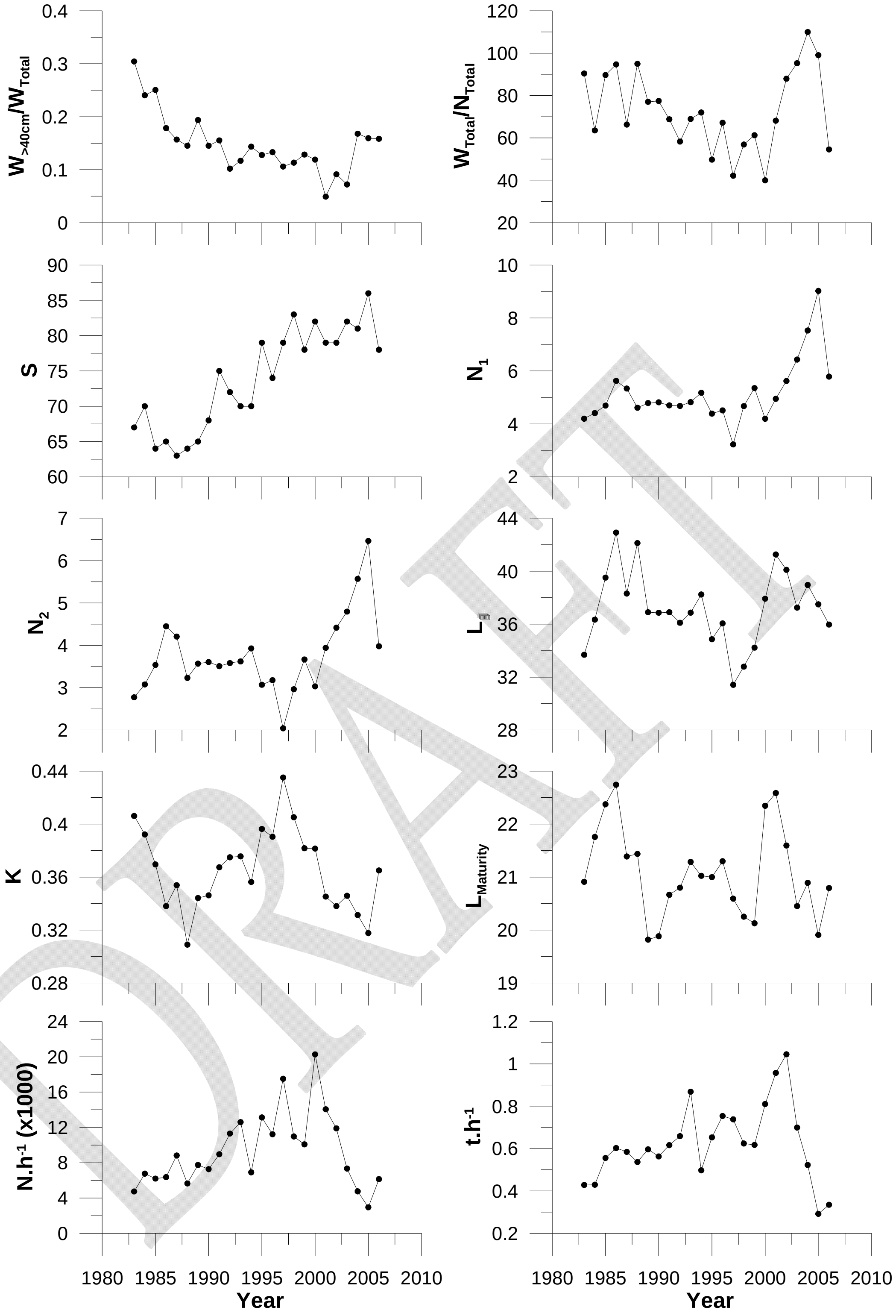


Figure 6.1: Trends in ten metrics calculated for the North Sea demersal fish assemblage as sampled by the Q1 IBTS (Proportion of large fish by weight $W_{>40cm}/W_{Total}$, mean fish weight W_{Total}/N_{Total}, Species richness S, Hill's N_1 and Hill's N_2 diversity indices, average ultimate body length L_∞, average growth rate K, average length at maturity $L_{Maturity}$, abundance $N.h^{-1}$ and biomass $t.h^{-1}$).

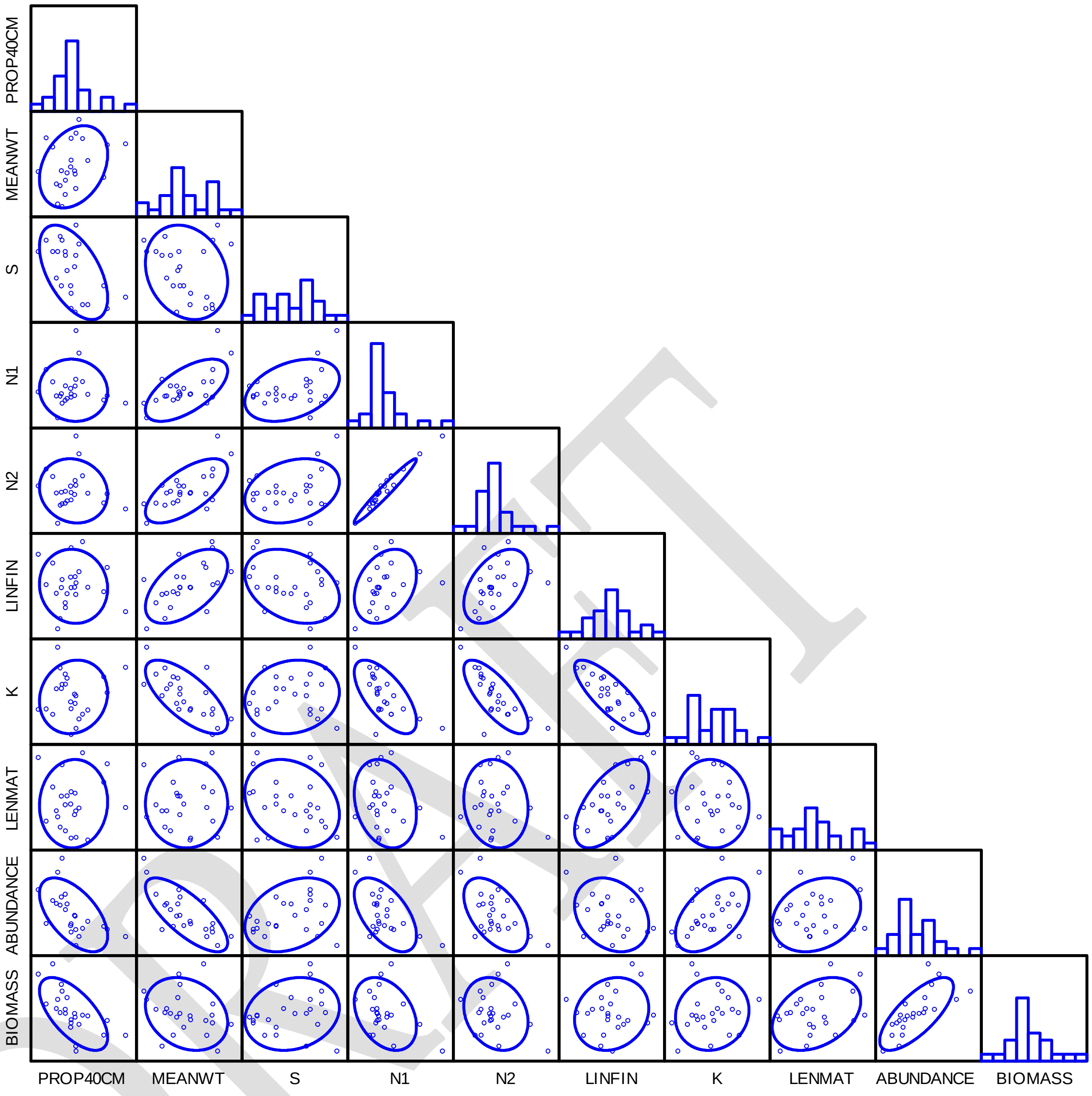


Figure 6.2: Scatterplot Matrix showing the relationships between each of the ten community metrics analysed.

Variation in all ten of these metrics is shown in Figure 6.1. These plots suggest that the level of redundancy among the different metrics was surprisingly low. Figure 6.2 shows a Scatterplot Matrix (SPLOM) relating variation in each metric to variation in all other metrics. Correlation coefficients are given in Table 6.1, with significance levels indicated. The major "take-home" messages are:

1) The two size-based metrics show the discrepancy in behaviour that lead to "mean fish weight" being dropped from the EcoQO. The influence on this metric of the larger than usual 1999 haddock cohort is clearly evident. The correlation between the two metrics is low and not significant statistically. This is evidence of the extent to which the "proportion of large fish" indicator, as currently defined for the North Sea demersal fish assemblage, has been divorced from immediate environmentally-related recruitment events. The eventual influence of the 1999 haddock cohort on the "proportion of large

fish" metric is indicated by the increase in the indicator value in 2004, thus exemplifying the sentinel role that the "mean fish weight" metric can perform.

2) The two Hill's species diversity indices are very tightly correlated suggesting that there is no requirement for both indices. Given the degree of dominance prevalent within the demersal fish community, Hill's N2 might be considered to be the most appropriate indicator, as variance in dominance is the most obvious way in which the community has changed. In reality, either metric would do.

3) Correlations between the index of species richness and each of the two species diversity indices were weak and not statistically significant. Species richness of the demersal fish assemblage has varied over the thirty year period for which data were available independently of variation in species relative abundance. There is clearly therefore a need for both a species richness and a species diversity metric to monitor all the changes that can take place within the demersal fish community.

4) The relationships between both species diversity indices and the "mean fish weight" metric were significantly positive. This can again be interpreted as indicating how the "mean fish weight" metric is influenced by single species recruitment events; the increases in the abundance of single species also resulting in increased dominance and consequent reductions in both species diversity indices.

5) Perhaps the most important result here is the strong and statistically significant negative correlation between the "proportion of large fish" metric and the species richness counts. The implication here is that, if these two traits in the fish assemblage are indeed linked in a phenomenological way, managing to restore fish size within the community could have important biodiversity consequences. The biological significance of this result therefore needs immediate investigation so that the most appropriate management actions can be identified.

6) The two von Bertalanffy growth parameters were strongly negatively correlated as expected from life-history theory (Andersen & Beyer 2006). There was clearly therefore no need to characterise the community using both these life-history metrics. Length at maturity and the growth parameter were not correlated, but length at maturity was correlated with L_∞. These results suggest that the combination of LMaturity and k might provide the most useful indicators of change in the life-history trait composition of the North Sea demersal fish community.

7) The von Bertalanffy growth parameter k was correlated with both indices of species diversity, suggesting that high species diversity communities were characterised by individuals with lower growth rates on average. Length at maturity was not correlated with either of the species diversity indices and none of the three life-history metrics was correlated with species richness.

8) Variation in the three life-history metrics was not correlated with variation in the "proportion of large fish" metric, but both L_∞ and k were correlated with variation in "mean fish weight". When the mean weight of fish in the community was lower, the community tended to be characterised by fish with smaller L_∞ and higher growth rate.

9) Measures of abundance tended to be negatively correlated with both metrics of fish size: when there were more fish in the area, their mean weight and the proportion of large fish in the community tended to be lower. Both species diversity metrics were also negatively correlated with the index of numerical abundance; high abundance communities were less species diverse.

To conclude, using the Q1 IBTS data set to monitor change in the structure and function of the demersal fish community of the North Sea would require employing a relatively high number of metrics to capture adequately all aspects of change in the community. The list of metrics examined here is not "exhaustive". Most might be considered to be metrics of "State" in that they describe the condition of the community. Some, such as the life-history trait metrics, hold implications with regard to functioning within the fish community. But there are other characteristics of the fish community, such as its trophic structure and productivity, for which we have not examined any metric. Taking account of these attributes of the fish community will require the derivation and analysis of additional appropriate metrics. Among the metrics examined, the degree of redundancy was relatively low. A list of the metrics needed is given in Table 6.2.

Table 6.1. Correlation coefficients an significance levels (* P<0.05, ** P<0.01, *** P<0.001) for correlations between each of the ten community metrics calculated on the Q1 IBTS data collected over the period 1983 to 2006.

	PROP40CM	MEANWT	S	N1	N2	LINFIN	K	LENMAT	ABUNDANCE	BIOMASS
PROP40CM	-									
MEANWT	0.304	-								
S	-0.540**	-0.199	-							
N1	-0.060	0.627***	0.380	-						
N2	-0.108	0.661***	0.276	0.972***	-					
LINFIN	-0.062	0.556**	-0.319	0.316	0.470*	-				
K	0.116	-0.704***	0.129	-0.665***	-0.751***	-0.809***	-			
LENMAT	0.045	0.003	-0.245	-0.235	-0.109	0.613***	-0.055	-		
ABUNDANCE	-0.572**	-0.754***	0.333	-0.561**	-0.526**	-0.257	0.515**	0.251	-	
BIOMASS	-0.643***	-0.237	0.175	-0.344	-0.222	0.158	0.104	0.365	0.740***	-

Table 6.2. List of metrics required to capture various aspects of change in the structure and composition of the demersal fish community of the North Sea.

Aspect of change	Metric
Abundance	Sample biomass (t.h-1)
Size Composition	Proportion of large fish
	Mean fish weight
Species Richness	Total species count
Species Diversity	Hill's N2
Life-history trait	Von Bertalanffy growth (k)
	Length at maturity

6.3 References

Andersen KH, Beyer JE (2006) Asymptotic Size Determines Species Abundance in the Marine Size Spectrum. The American Naturalist, 168, 54-61.

Cury, P. M., and V. Christensen. 2005. Quantitative ecosystem indicators for fisheries management. ICES J. Mar. Sci. 62:307-310.

EU Commission. 2006. Report of the Scientific, Technical and Economic Committee for Fisheries (STECF) - Subgroup for Research Needs, SGRN-06-01: Data Collection Regulation Review.

Greenstreet, S. P. R. & Rogers, S. I. 2006. Indicators of the health of the fish community of the North Sea: identifying reference levels for an Ecosystem Approach to Management. ICES Journal of Marine Science, 63, 573-593.

Greenstreet, S. P. R., Spence, F. E. & McMillan, J. A. 1999. Fishing effects in northeast Atlantic shelf seas: patterns in fishing effort, diversity and community structure. V. Changes in structure of the North Sea groundfish assemblage between 1925 and 1996. Fisheries Research, 40, 153-183.

ICES. 2007a. Report of the Working Group on Ecosystem Effects of Fishing Activities (WGECO). ICES CM2007/ACE:04, 162pp.

Jennings, S., Greenstreet, S. P. R., Hill, L., Piet, G. J., Pinnegar, J. & Warr, K. J. 2002. Long-term trends in the trophic structure of the North Sea fish community: evidence from stable isotope analysis, size-spectra and community metrics. Marine Biology, 141, 1085-1097.

Jennings, S., Greenstreet, S. P. R. & Reynolds, J. 1999. Structural change in an exploited fish community: a consequence of differential fishing effects on species with contrasting life histories. Journal of Animal Ecology, 68, 617-627.

7 Methods for comparing maps

7.1 Introduction

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

Geographic visualization and geo-spatial analysis of fish, fishery and environmental data are increasingly more important components of fisheries research. Evaluation of temporal changes in spatial distributions however, can lead to the production and comparison of many maps. For example, in the 2007 report of WGFE, average maps for three time periods were used for visually detecting changes in fish distributions in European waters which might be related to climate change or abundance-occupancy drivers (ICES, 2007a, Sect. 2.1). While a map can efficiently present an existing pattern (and replace voluminous descriptive text), visual comparisons between maps are subjective. Under this ToR, we evaluate quantitative methods for comparing maps.

Several methods exist in the literature and have been used in different domains for comparing two or several maps. A brief overview is given with emphasis on the underlying assumptions and questions that can be studied with each method. Some of the methods will then be compared for a common data set. The methods fall broadly into two categories, those assuming that the data (either presence-absence or a quantitative measure) is on a regular grid or not. If gridded data are necessary, this can be available either by design or after processing of the data – binning or point aggregation, i.e. averaging per statistical rectangle. Alternately, interpolation yields a continuous surface derived from the point data, i.e. potential mapping in SPANS (Kulka *et al.* 2005), kriging (Rivoirard *et al.* 2000) or linear interpolation between neighbouring cells such as triangulation (Delaunay triangles, Voronoi polygons) (Kulka *et al.* 2005).

Earlier studies by WGEF illustrated temporal shifts in fish distributions by mapping the geographical centroids of various species by year (see Section 7.3 of ICES (2004)). These centroids (mean central point location and latitudinal and longitudinal quartiles) identified the areas of peak distribution for fish species distributed around a single, central area, but did not reflect the true mean distribution of fish with skewed or patchy distributions (as is usually the case for marine fish). When several concentrations exist, centroids may occur where there are few fish or even over land. Hence results from this method can be misinterpreted.

7.2 Comparing two maps: visualising areas that are different

Testing differences in log-ratios in grid cells

A method for comparing two maps to elucidate the areas where they differ significantly was used by Fraser et al (in press). In each grid cell they calculated the log-ratio of the indices log (index2/index1). If two indices are identical, the log-ratio is 0, positive values mean that index2 was larger and negative values that index1 was larger in a given grid cell. Thus, if the indices are densities in period 1 and 2, positive values mean that densities were higher in the second period. To remove small random noise, the log-ratios are smoothed with a two-dimensional spline, assuming normal errors and an identity link. The fitted values and the standard errors from the smooth are used to test, for each grid cell, whether the log ratio is significantly different from 0. The authors plotted the smoothed log ratios with contours corresponding to pointwise significance levels of 0.0001, 0.001, 0.01, and 0.05. For example, the 0.05 contour lines show where the indices in the two maps differ at a pointwise significance level of 5%. Alternatively the p-values can be plotted directly on the grid level instead of the contour lines.

Matrix modelling in SPANS, involving the overlaying (comparing) of one to many interpolated surfaces can be used to determine degree of change between surfaces. This method provides a higher resolution picture than gridding as it provides a continuous result. Refer to Fig. 2.4.1.x for examples of this method.

In certain cases it might be preferable not to smooth the maps, in particular if the grid is rather coarse (sampling positions are widely spaced). One option would then be to calculate p-values for the null hypothesis H0: log-ratio=0 assuming the estimated log-ratios follow Normal distributions. The variance is estimated as follows, using the delta method for the second step

$$\hat{V}[(index2/index1)] = \hat{V}[index2]/(index1)^2 + (index2)^2\,\hat{V}[index1]/(index1)^4$$

$$\hat{V}[\log(index2/index1)] = \hat{V}[index2/index1]/(index2/index1)^2$$

The variance for each index is estimated by the sampling variance. For example if the indices are mean densities, they correspond to the sampling variances of the mean.

As a first example for the method consider the spatial distribution of three biodiversity indices in the Bay of Biscay in two time periods about ten years apart. The indices are species richness (N0), exp (Shannon index) (N1) and delta1 (probability that two individuals taken at random from the community are different species). The indices for each bottom haul (quarter 4 Western IBTS survey) have been averaged by grid cells across three years for period 1 (1992, 1994 and 1995) and period 2 (2004 to 2006) and then plotted as maps (Fig. 7.2.1). No smoothing was carried out and no correction for catchability difference was applied. In the first period, species richness was somewhat higher in the Northern part, while delta1 was higher in the South.

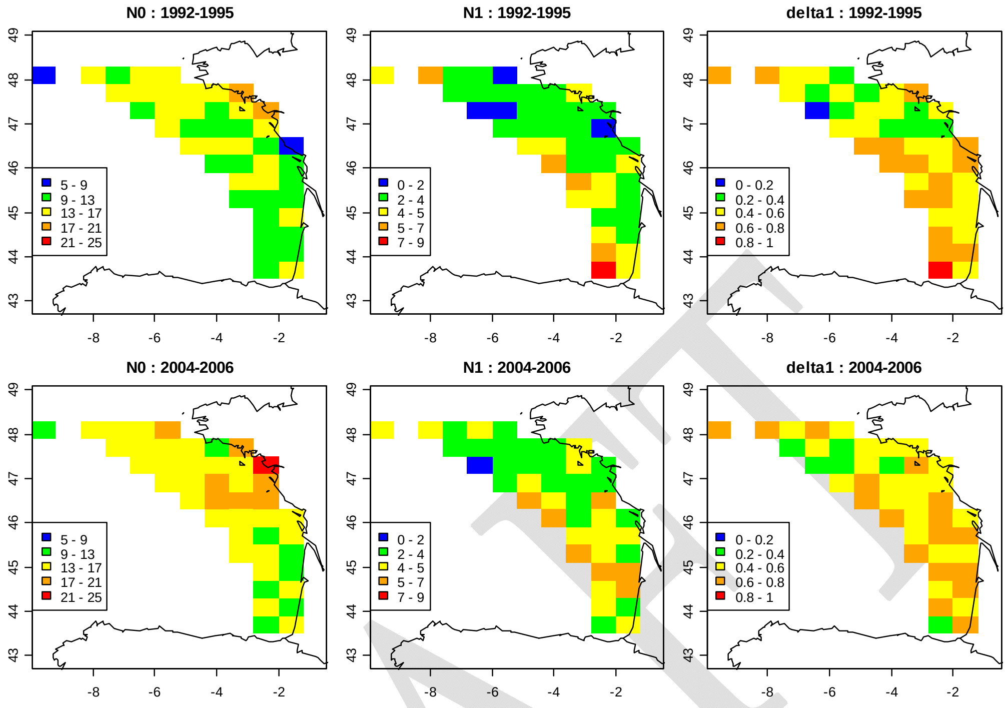


Figure 7.2.1: Diversity indices for two time periods in the Bay of Biscay averaged by grid cell. N0 species richness, N1 exp (Simpson index) and delta1 (probability two individuals are different species). The colour scale corresponds to quartile ranges.

Next the log-ratio of the two maps was calculated, log (period2/period1) and a pairwise test carried out for each grid cell (Fig. 7.2.2). The test results show that only in a few grid cells did diversity increase significantly ($p<0.05$). The cells with changes were not the same for the three indicators but always either coastal or situated on the shelf edge in the Northern part of the bay. Note that p-values would be plotted as negative if the log-ratio was negative; however, only one of the negative changes was significant.

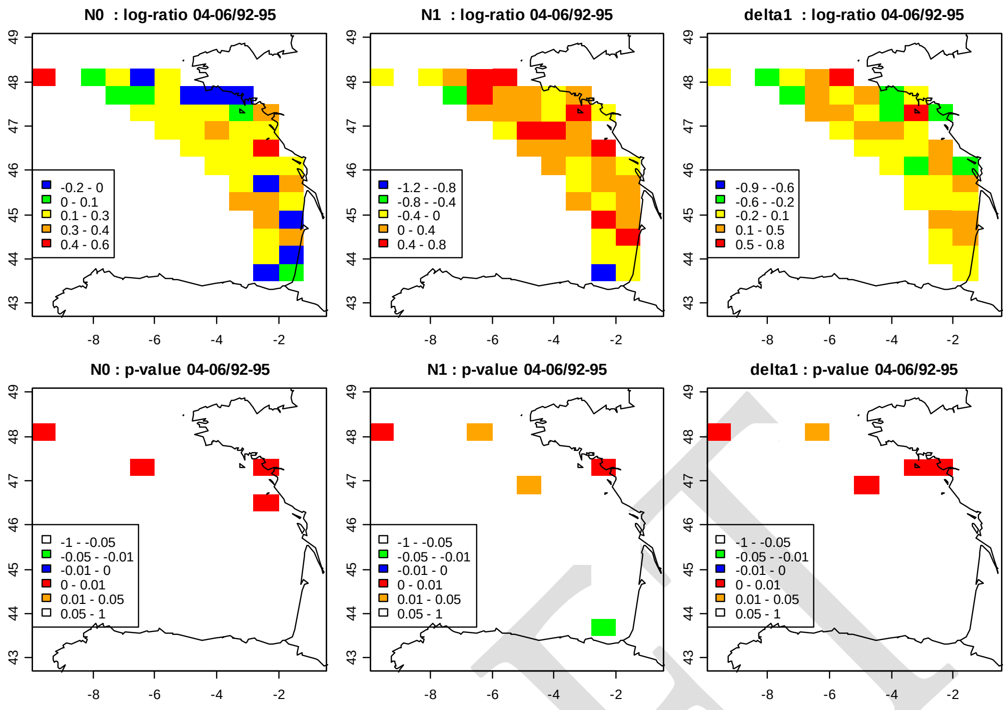
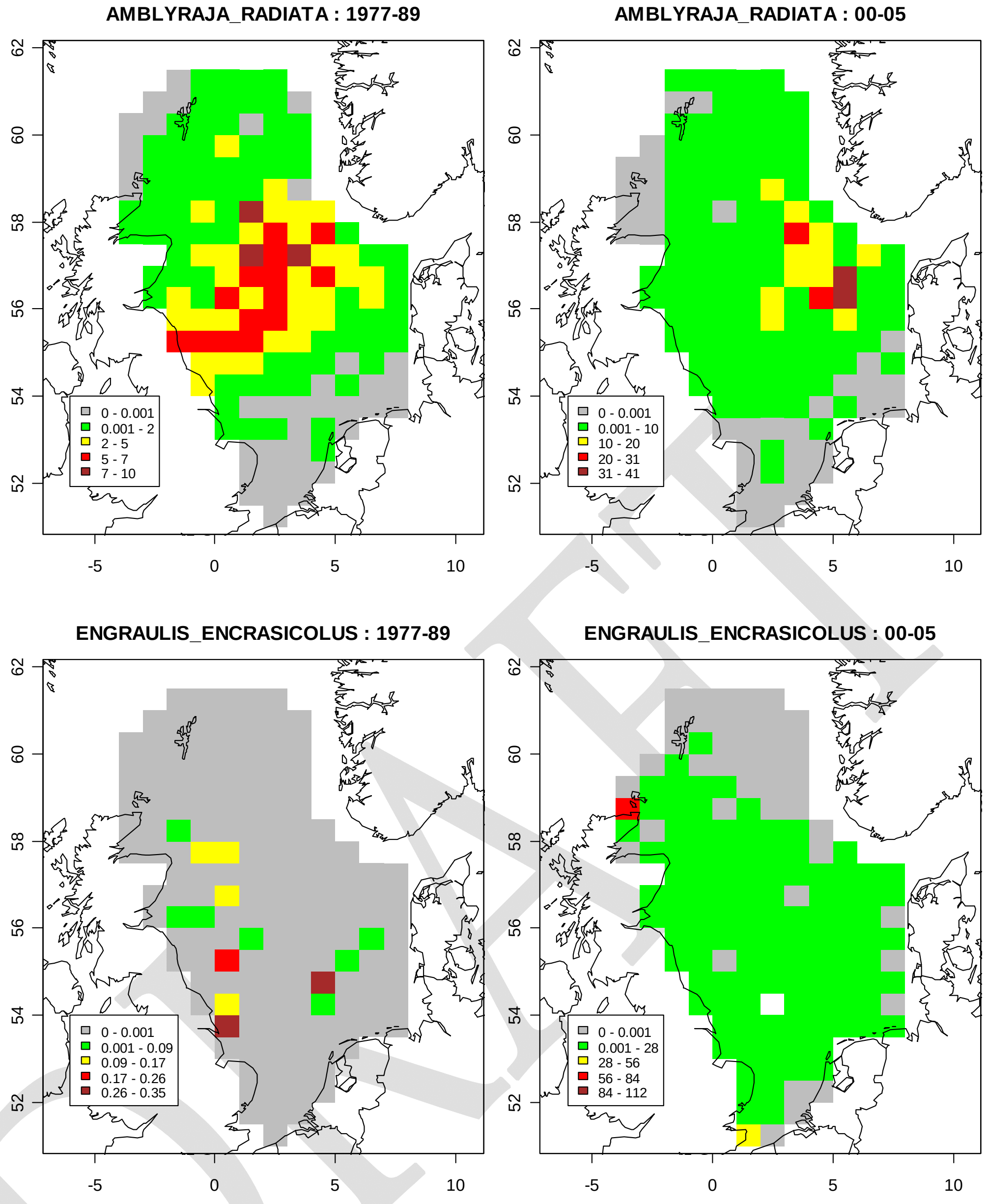


Figure 7.2.2: Test for difference in diversity indices between period 1 and period 2. Top row: log-ratios by grid cell log (period2/period1). Bottom row: p-values for hypothesis test H0 log-ratio=0.

The second example concerns species densities in the North Sea. Maps of average densities per haul per rectangle for two time periods, 1977-1989 and 2000-2005) are shown in figure 7.2.3. Large changes between periods are visible with the disappearance of grey rectangles, indicating spatial spreading across a larger fraction of the survey area, as for anchovy and red mullet, and in contrast the appearance of grey rectangles indicating contraction for haddock.



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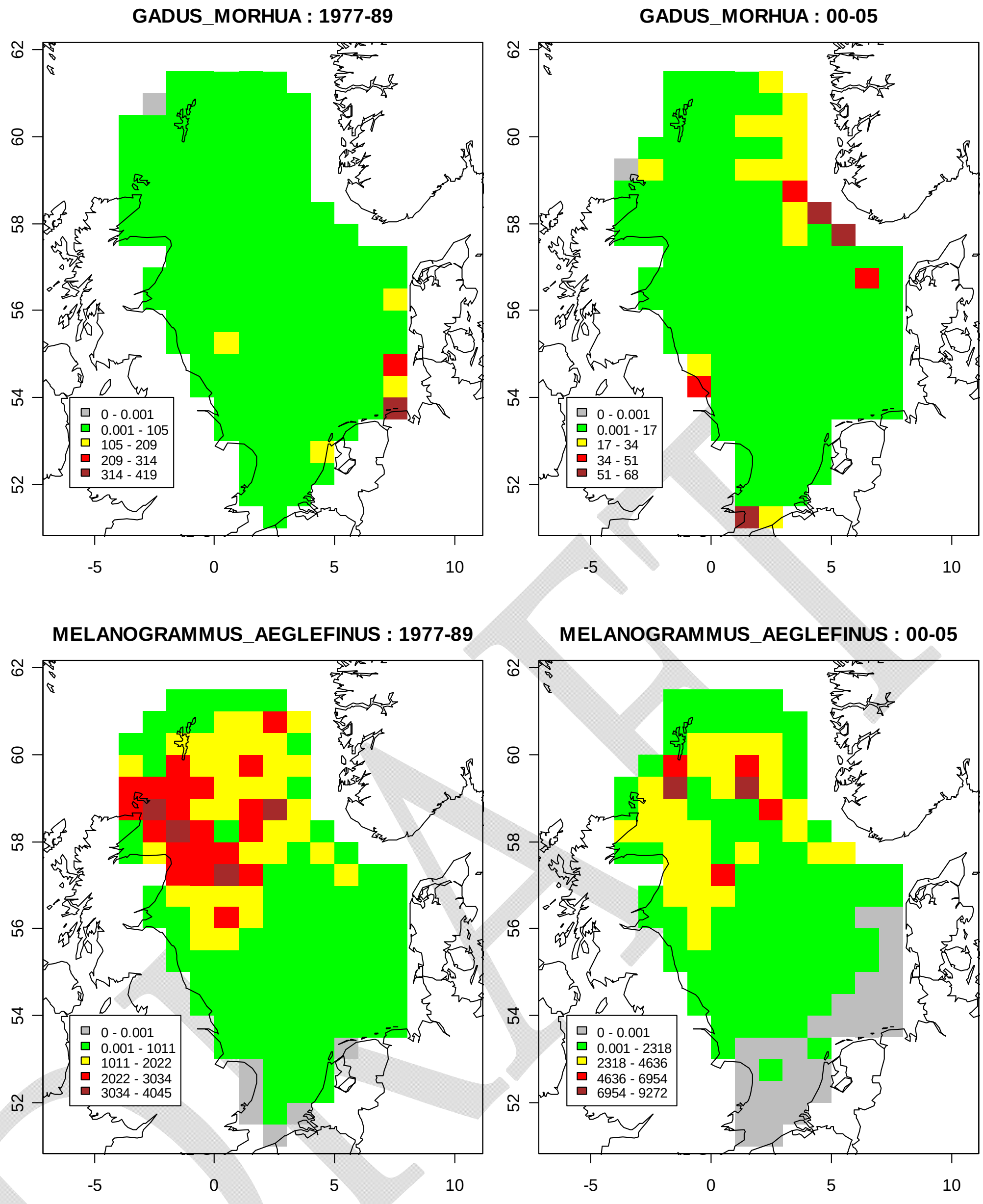

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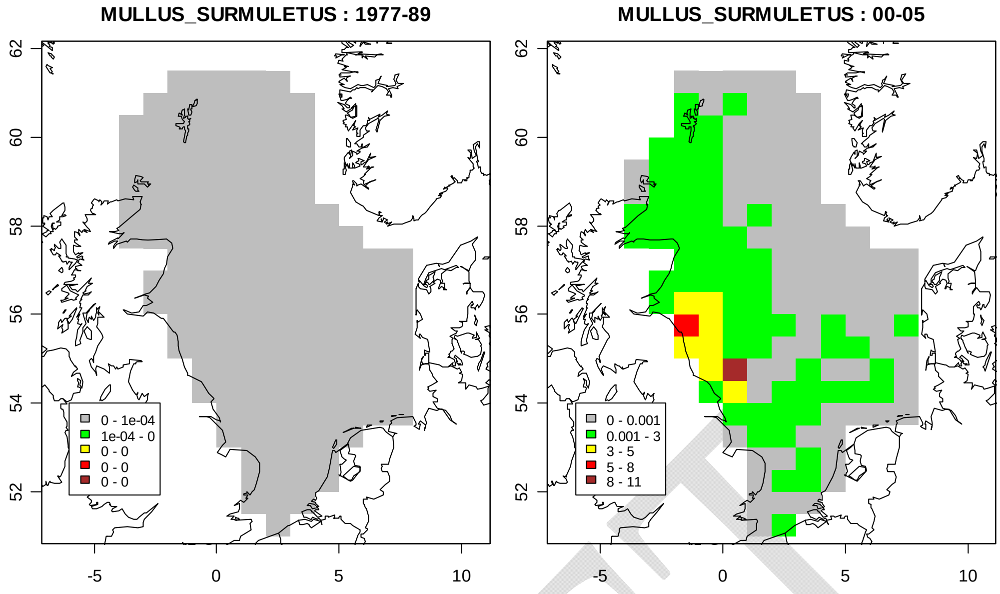
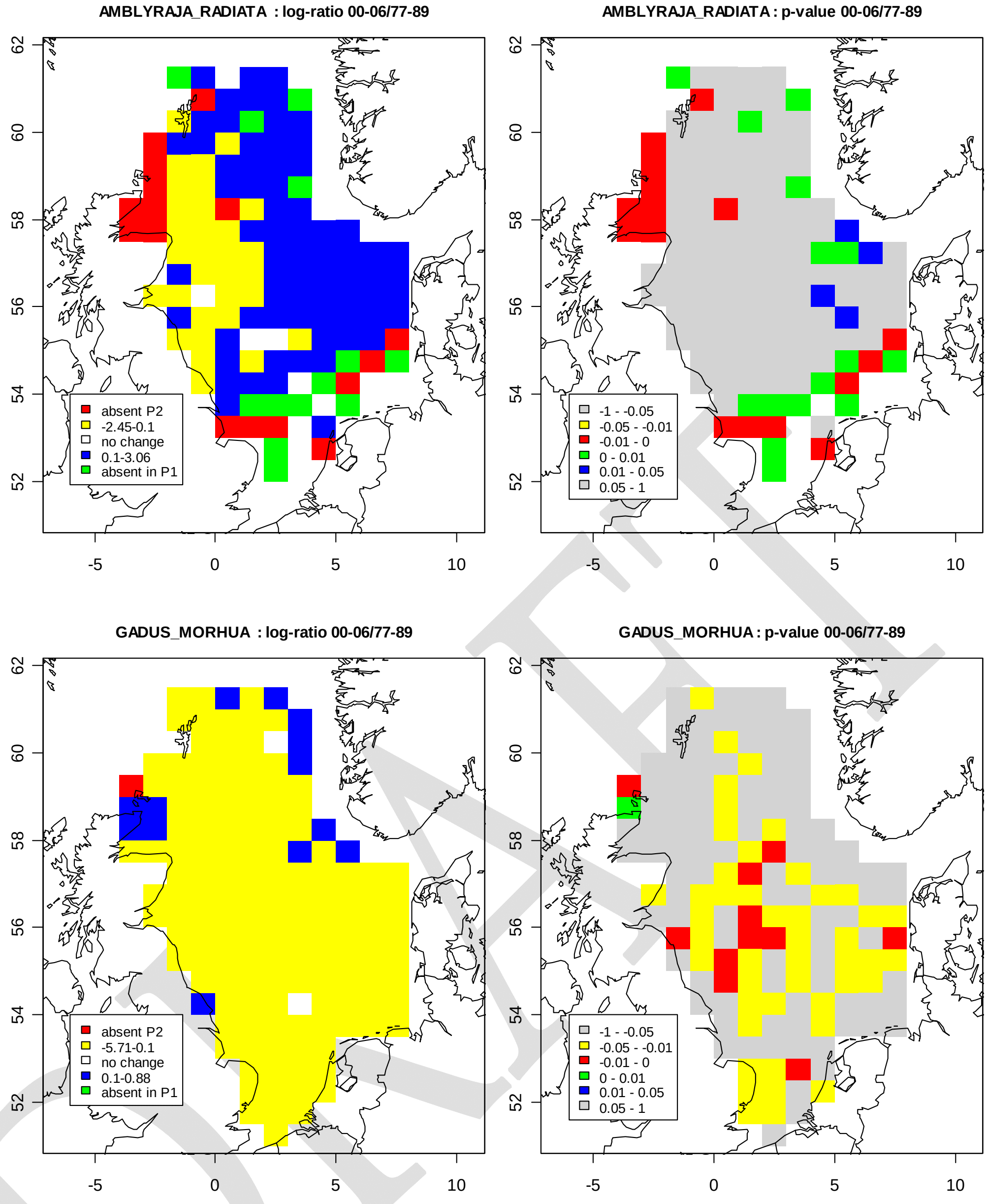


Figure 7.2.3: Maps of average density per haul per rectangle across years from quarter 1 IBTS data. The colour scale corresponds to quartile ranges of density. Green 0-25%, yellow 25-50%, red 50-75% and brown 75-100% density.

The log-ratios and corresponding p-values for the null hypothesis of no change are shown in figure 7.2.4. Note that in the case of a species not being present in both periods, the log-ratio can not be calculated and no test can be carried out. In this case the change was noted as significant (small p-value).



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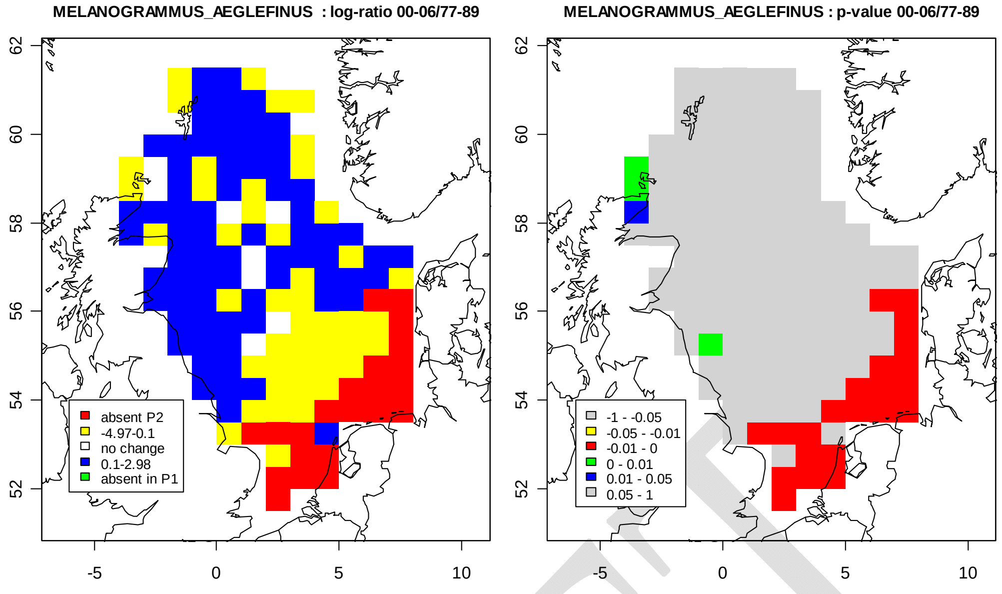

Figure 7.2.4: Maps of log-ratios of mean densities for 2004-06/1992-95 using quarter 1 IBTS data on left and p-values for test of significant change on right. If species was absent in one of the periods, the p-value is set to be significant (red when species disappeared from cell and green if it appeared).

The advantages of the method are that a statistical test for significant changes can be carried out which allows taking small scale spatial and temporal variability into account. The disadvantage is that the log-ratio cannot be calculated if the index is zero in any one of the maps. The data was not smoothed in the examples presented, given the coarse scale of the grid. However, for a smaller scale grid this should probably be done. Another variant of comparing values in grid cells is to calculate percentage differences rather than log-ratios.

Comparing periods with grids: the example of North Sea cod

A method of quantifying the change in patterns of abundance between two periods of time is presented below. Average abundances for Period 1 (1977-1989) and for Period 3 (2000-2005) are contrasted and the changes over the time span of about 30 years are quantified. This procedure allows for a first inspection, especially in order to look for the magnitude of change that occurred, or to use the spatially resolved mapping to identify the regions with most pronounced changes. It involves the following steps: (1) Transformation of (abundance) point data from surveys – in this case the IBTS – to gridded data. The value assigned to a cell represents the mean value for all measurements obtained within the respective time period and grid cell. The choice of grid cell size depends on the spatial resolution of the original data and was here chosen to be roughly ¼ of an ICES rectangle. (2) Plot of maps representing the average or typical situation for the periods in question (Figure 7.2.5). (3) Visualization of the magnitude of change (Figure 7.2.6). In this case, values were calculated as percentage differences relative to the prior of the two time periods.

The maps provide an easy means to visualize changes such as the decline in the abundance of North Sea cod over the last 30 years, which occurred over the entire area of the North Sea, and particularly in the southern part. The maps also allow for quantification of changes, of course independently of an evaluation of possible causes. As pointed out in section 2, the decline in cod abundance was paralleled by an increase in temperature. However, detailed and multivariate analyses are required in order to estimate the effect of temperature change (or other factors, such as fishing activities) on the observed changes.

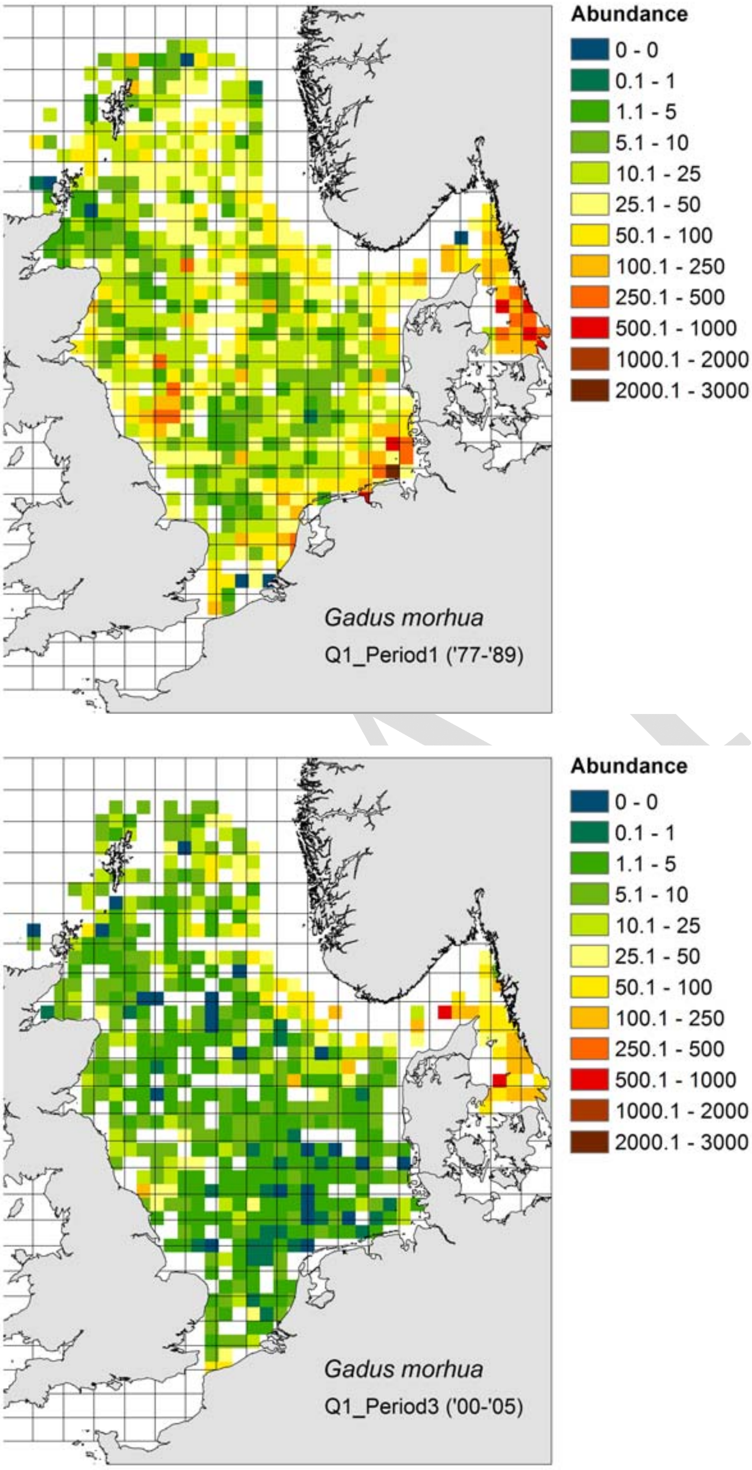

Figure 7.2.5: Gridded averages of abundance [ind per 1-hour haul] in all hauls of Q1, periods 1 (top panel) and 3 (bottom panel).

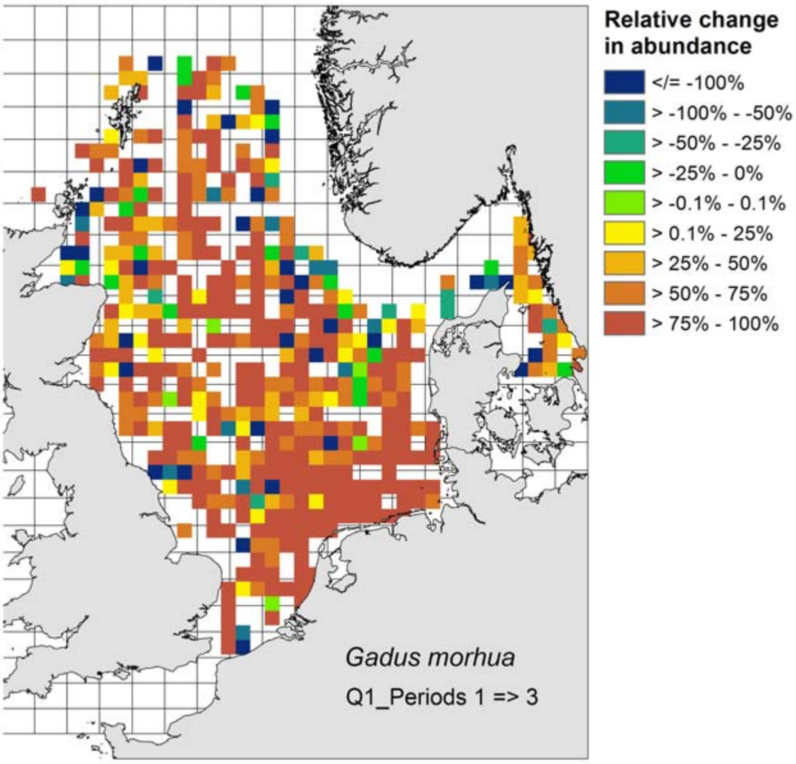

Figure 7.2.6: Gridded relative difference in cod abundance in the North Sea in quarter 3 between the periods 1 and 3. Red colours symbolize a decrease in abundance, blue colours an increase.

Comparing methodology: North Sea cod example

To compare the methods described in the previous two sections, plus the method used in chapter 2 (2.4), changes in the spatial distribution of cod in the North Sea were evaluated using: I= log-ratio test for grid cells, II=Relative change in density using matrix modelling; III= Difference in maps of gridded data. Quarter 1 IBTS bottom trawl data was used for period 1 (1977-89) and 3 (2000-05).

Changes in the abundance of cod have been previously discussed. Our results are not meant to reiterate the status of the stock. Rather they are meant to illustrate the spatial dimension of changes at a finer resolution, within the OSPAR area II. The three methods all show that densities decreased in between the two periods but show that the changes were spatially complex and uneven (Figure 7.2.7).

Features of the individual methods:

i) The matrix overlay map, a spatial modelling approach in the GIS (Fig.7.2.7, upper right) illustrates degree of change in density of fish between periods over a continuous surface (see Sect 2.4.x for a detailed description of this method). The advantage of this approach is that change can be observed on a highly resolved (virtually continuous) spatial scale and degree of change can be quantified at any location or area (user defined within the GIS). For example, the reduction in density of cod was highest to the south-east along the Dutch coast (boxed area in Fig.7.2.7, upper right), where density decreased by about 53 times (average of 274 fish/tow in the early period vs. 5 fish/tow in the later period), while a limited increase (about double) was observed along the north-eastern fringe. The reduction is density in the south-east was the main contributor to the overall reduction in abundance of cod. However, density at the north-eastern fringe was very low during both periods and so the increase there had little effect on overall change in abundance. An increase in density was observed over 11% of the survey area (but where density was low during both periods) and a decrease over 87% (including the south-east where degree of change was greatest). Within the North Sea, a northward shift in the mean latitudinal distribution of cod occurred (what was determined to be an increase in density along the north and eastern periphery of the survey area, a fairly constant decrease over the rest of the area except a very large decrease to the southeast fringe).

ii) The hypothesis test of the log-ratio method (Fig.7.2.7, upper left) has the advantage of testing for significant differences. The test indicated that this change was only significant in the centre of the Southern North Sea and in a narrow strip in the North. The test for high density cell in the southeast corner was not significantly different between periods. However, an examination of the raw data there shows that the data are highly skewed, (same area as discussed in (I) above as an area of large change) and remains highly skewed (does not become normally distributed) when transformed. Thus, the test is unable to discern a change. The change in that area to the southeast is significant and large. Examining the raw data, numbers of data points are n=78 in 1977-89 and n=32 in 2000-05 with very different ranges of values (0-5004 individuals/tow in 1977-89 and 0-5.1/tow in 2000-05): 88% of sets in the early period exceeded the maximum value (5.1) observed in the latter period. The large difference was not as a result of outliers.

iii) The difference between maps of gridded data provides grid-by-grid values of change, presenting the raw data without any further interpolation. The advantage is that an unbiased picture is created which can also reveal local outliers or areas of high variability (in the map below visible as incidences where red and blue grid cells directly neighbour each other). Additional maps visualizing the degree of variation in the data within grid cells could advance this method. The spatial resolution of grid cells can be chosen by the user, i.e. be adapted to the amount of data points available per area and time period.

In conclusion, this example brings out the importance of choosing the approach used to compare maps. It also illustrates that plotting and comparing not only mean values but also higher moments, such as the variance and skewness and also median values would be of value as they also indicate properties of changes in the spatial distribution. The issue of smoothing the initial observations or the comparison (log-ratio or difference) also deserves further investigations.

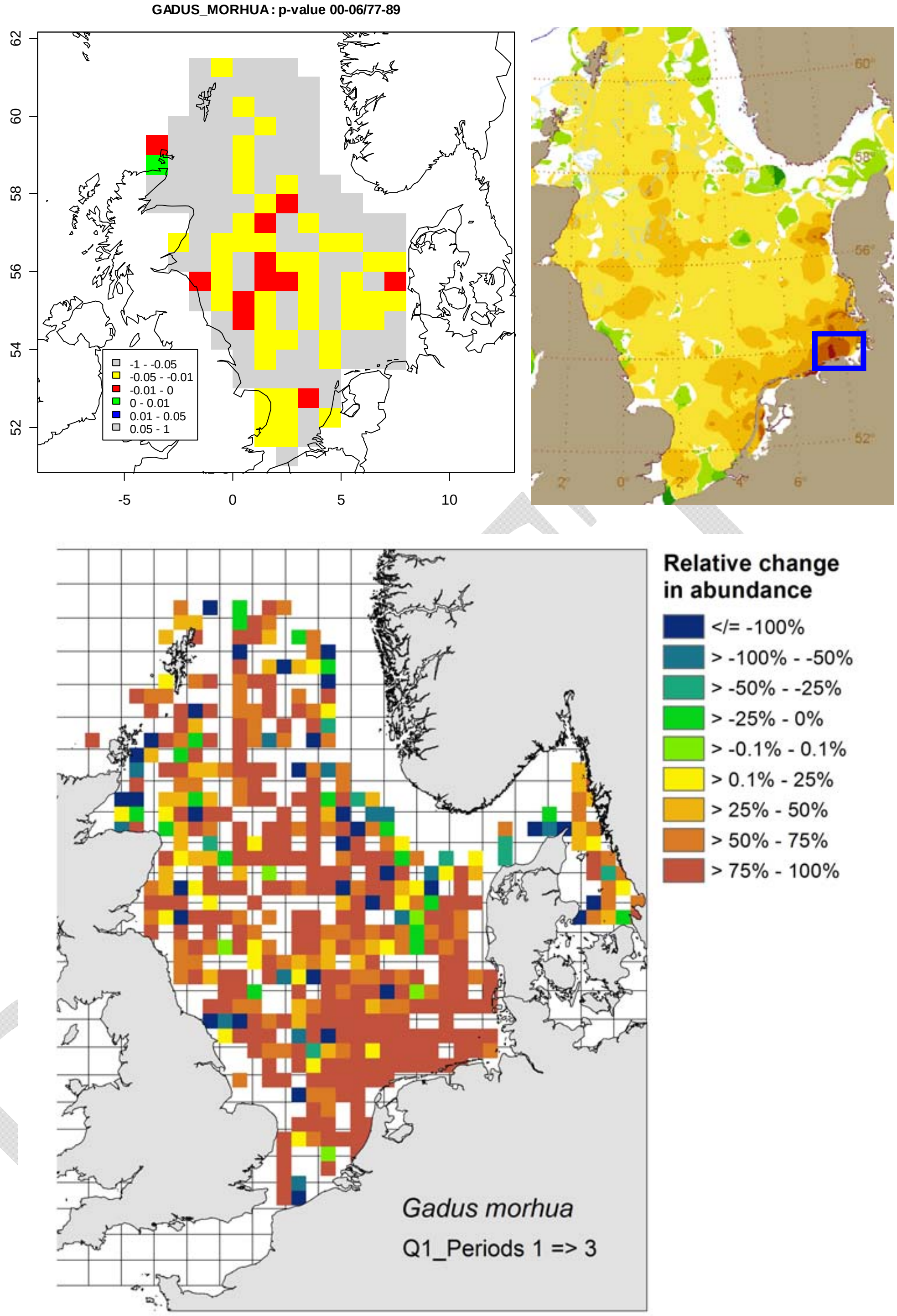

Figure. 7.2.7: Comparison of three methods for detecting changes in the spatial distribution of cod (*Gadus morhua*) in quarter 1 between two periods (1977-'89) and (2000-'05); comparison between mean values for each period. Blue box on the upper right map shows where the change in density calculation was performed, where there was a 53x decrease in density between 1977-89 and 2000-05.

7.3 Using empirical orthogonal functions (EOF) to extract principal scales in fish spatial distributions and their temporal changes: case study for anchovy in the Bay of Biscay

In the context of habitat mapping and understanding changes in the spatial distributions of fish populations, characterising the constant average features of the distributions over time as well as the variability between years in these distributions and relating this interannual variability to some explanatory variables, e.g. environmental forcing, behaviour and learning, density-dependence is useful (ICES, 2007a, b). To achieve this, empirical orthogonal functions (EOFs) provide a suitable mathematical framework for characterising principal scales (eigenvectors) in the variation around the average spatial distribution and how these can be combined to produce the annual maps. When these principal components can be related to explanatory variables, future spatial distributions can be predicted by appropriate combination of the principal scales for different scenarios of explanatory variables. The application of this method is demonstrated for the case of anchovy in the Bay of Biscay using time series of gridded fish spatial distributions estimated from survey data. Principal components in the spatial distribution are characterised and the inter-annual variability modelled as a weighted sum of these components via time-dependent amplitude coefficients. The amplitudes for two components are related to population biomass. Maps of the population distribution are then predicted for low/high biomass scenarios, showing contraction / expansion in the distribution.

EOF decomposition of space-time variability in distribution

Empirical Orthogonal Functions (EOFs: Preisendorfer, 1988) have long been used in meteorology and oceanography to decompose the time and space variability of a time series of gridded maps into principal scales and their amplitudes. The decomposition is a linear factorisation of spatial components (eigenvectors) that are constant in time and amplitudes (principal components) that are variable in time. The distribution map in each year is thus modelled as the sum of spatial components that are weighted by their annual amplitudes:

$$Z(t,s) = \bar{Z}(.,s) + \sum_{m} \lambda_m \, u_m(t) \, e_m{}^T(s) \qquad [1]$$

$Z(t,s)$ density at time t and location with spatial coordinates s

$\bar{Z}(.,s)$ density averaged over time at location s

λ_m non-zero eigenvalues

$e_m{}^T(s)$ eigenvectors or EOFs (principal scales)

$u_m(t)$ principal components (amplitudes) scaled to unity (as in [1]) or to their corresponding λ_m

To achieve the decomposition, the method proceeds as follows. Z(t,s) is a matrix containing each gridded (density) map in each line, thus having t = 1,…n lines and s = 1,….k columns. Each grid cell must have a number. In case of missing information interpolation is needed or use of a coarser grid. From each grid cell, its average is subtracted giving the matrix of the gridded map anomalies is: X(t,s)= $Z(t,s) - \bar{Z}(.,s)$. $S = X^T X / n$ is the the covariance in space over time and $S_a = XX^T / k$ is the covariance in time over space. A principal components analysis (PCA) of matrix S (or equivalently

S_a) is then performed which gives the eigenvalues λ_m , eigenvectors $e_m(s)$ and principal components $u_m(t)$.

Note that in the EOF decomposition, no space-time cross-covariance term is considered such as $X(t,s)X(t+\delta,s+h)$. The underlying geostatistical structural model of an EOF decomposition is a spatial model that varies with time but that has no temporal structure formulated (the temporal structure being in the time series of the amplitudes). The variogram at time t would be: $\gamma_t(h) = \sum_m \alpha_m(t)\,\gamma_m(h)$. The EOF decomposition is therefore less suited to situations in which patches change their locations in time. To consider cross space-time terms, so called extended EOFs have been developed which relate to more advanced geostatistical models and also Min/Max auto-correlation factors (MAFs).

Also note that with the EOF decomposition, the time variation in each grid cell is characterised independently from the other cells. If the maps contain spatial structure, neighbouring cells will respond in a similar way and thus the spatial structure will be taken into account implicitly and show up in the EOFs without explicit formulation.

Bay of Biscay anchovy

Georeferenced anchovy density values (tonnes n.m.$^{-2}$) in Biscay were derived from the IFREMER acoustic survey series PelGas (one value per n.m. along E-W transects spaced 12 n.m. apart). These values were averaged in cells of 0.4° latitude by 0.2° longitude. The time series (2000-2007) of the gridded density maps (93 cells each) was organised in matrix form (Z(t,x) with 8 lines and 93 columns). Instead of absolute density maps, percentage biomass $P(t,s)=100Z(t,s)/\sum_s Z(t,s)$ was used. $P(t,s)$ was then centred by column (grid cell) to form matrix X.

The biomass time series ($B(t)=\frac{1}{k}\sum_s Z(t,s)$; Fig. 5.3.1a) showed first three years of high biomass and then a marked decrease. The mean spatial distribution of percentage biomass $\overline{P}(.,s)$ shows two preferred habitats (Figure 7.3.1b): one coastal off the Gironde estuary (45.5°N, 1.5°W) and one on the outer shelf off Les Landes (44°N, 2°W).

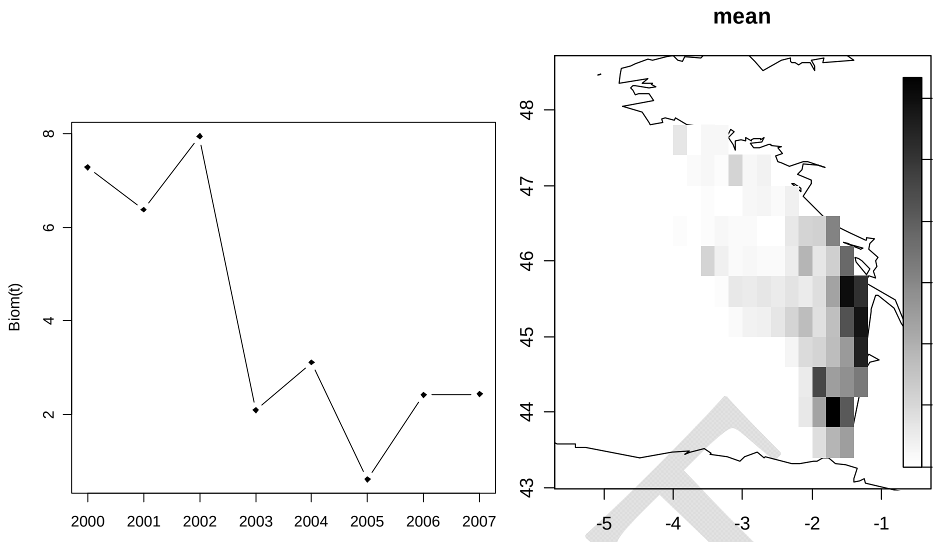

Figure 7.3.1: Time series of total biomass (left) and map of average percentage biomass (right) for the anchovy in Biscay (acoustic surveys 2000-2007).

The EOF decomposition of the residual percentage biomass matrix X explained 30% of total variance in the first principal scale, 50% in the first 2 and 70% in the first 3. Subsequently, only the first 2 principal scales were considered. The maps of the 2 first EOFs (Figure 7.3.2) show the areas where highest and lowest residual variability takes place above or below the mean. EOF 1 characterises the opposition in the variability in the high biomass sites, Gironde and Landes estuaries, where one site has high residuals above the mean the other has them below the mean. EOF 2 characterises the dispersion outside the high biomass sites, Gironde and Landes, when the other areas show residuals above the mean, these sites have them below the mean. The time series of the amplitudes (Figure 7.3.2) shows a clear trend for the EOF2 but not for EOF1.

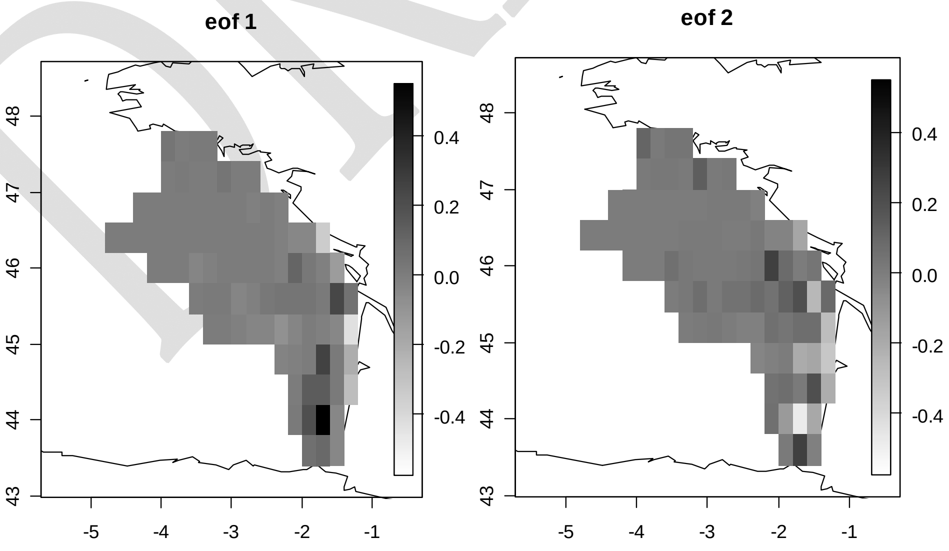

Figure 7.3.2: The two first empirical orthogonal functions (EOFs) characterising the time-invariant principal spatial scales in the centred densities.

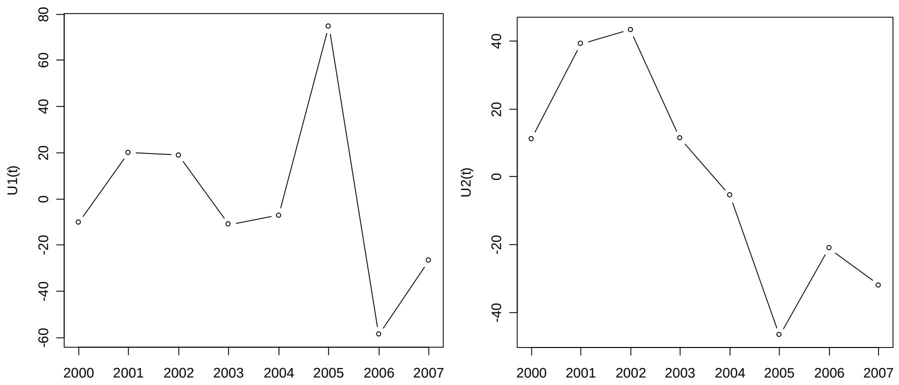

Figure 7.3.3: Time series of the amplitudes $u_m(t)$ for the two first EOF.

In the factorial plane of the amplitudes (Figure 7.3.4 left) the position of the yearly maps showed a clear separation between the maps containing low biomass (2005-2007) and the other maps. Considering the years 2000 and 2001 as reference, the distance in the factorial amplitude plane of each year to the reference can serve as an index to monitor changes in the global spatial distribution (Figure 7.3.4 right). The amplitudes were scaled to the eigenvalues (i.e., the variance supported by the corresponding EOFs).

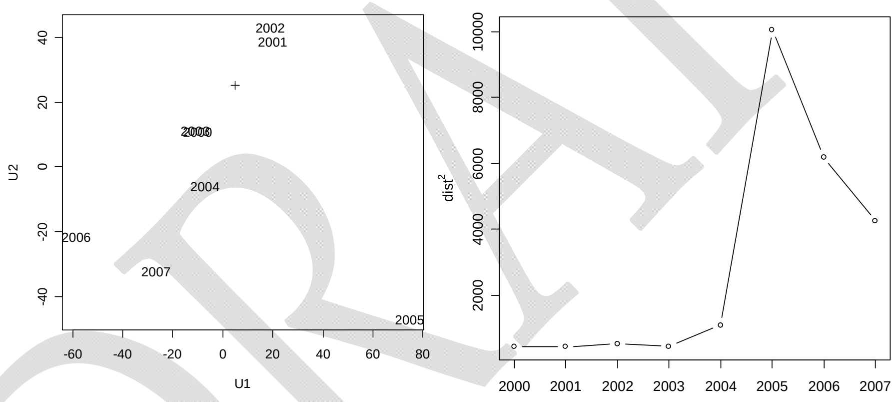

Figure 7.3.4: Differences in annual percentage biomass maps based on differences in the amplitudes of the EOF. Left: position of the annual maps in the first factorial plane of the amplitudes (the point is the position of the reference). Right: distance to the reference to serve as a monitoring index of changes in maps.

The amplitudes U1 and U2 were then related to the total biomass in each year (Figure 7.3.5). Both amplitudes show a relationship with biomass implying that biomass might be a conditioning factor for the spatial distribution. To illustrate this, a low and a high biomass scenario were designed with corresponding amplitudes (Table 7.3.1) and the maps for each scenario were predicted (Figure 7.3.6). The low biomass scenario had a predicted spatial distribution concentrated on the high biomass areas only (Gironde and Landes) while the high biomass scenario leads to a more dispersed distribution. Whatever the scenario, the Gironde area (4 squares) concentrated 20% approx. of the biomass, while the predicted percentage biomass for the Landes area depended on the scenario.

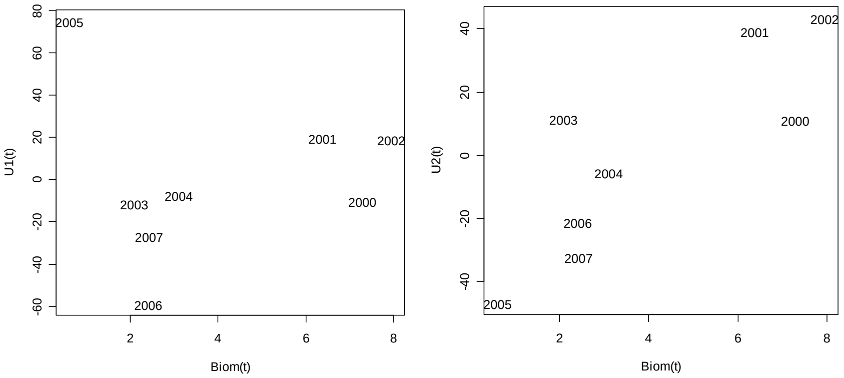

Figure 7.3.5: Relationship between the amplitudes U1 and U2 of the two first EOFs and total biomass Biom in each year.

Table 7.3.1. Prediction scenarios for two biomass levels. Amplitude values used based on their relationships with the biomass (Fig. 5.3.5) and predicted percentage biomass in the Gironde and Landes areas. Gironde : (45.6N, -1.5W), (45.6N, -1.3W), (45.2N, -1.5W), (45.2N, -1.3W); Landes : (44.4N, -1.9W), (44.4N, -1.7W), (44N, -1.9W), (44N, -1.7W).

	AMPLITUDE U1	AMPLITUDE U2	GIRONDE	LANDES
High biomass (B=8)	+ 15	+ 40	19 %	15 %c
Low biomass (B=1)	+ 42	- 30	21 %	21 %

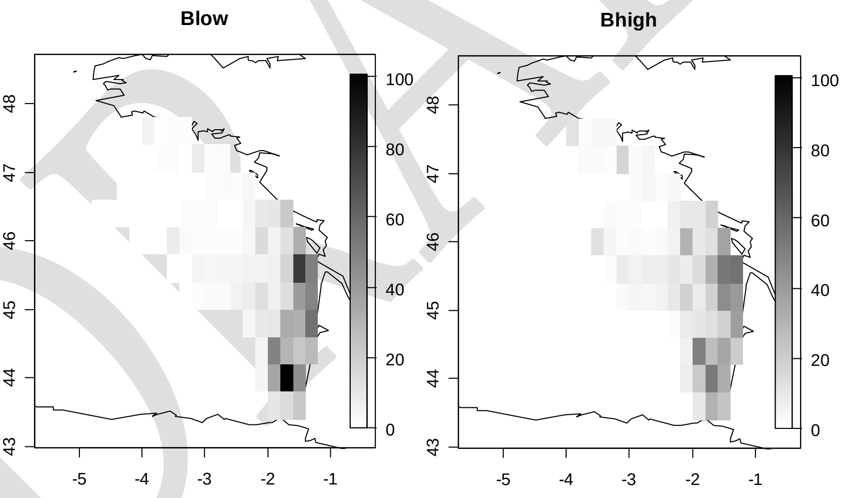

Figure 7.3.6: Predicted spatial distributions using EOF1 and EOF2 for high and low biomass scenarios (see Table 7.3.1), based on the relationship of the amplitudes U1 and U2 with biomass (Figure 7.3.5).

Conclusion

The EOF method extracts the principal time-invariant scales in a time series of maps and estimates time-varying amplitudes for each scale. It also allows us to predict spatial distributions for different scenarios when amplitudes can be related to explanatory variable such as biomass or some environmental parameters. The prediction is done by appropriately combining the principal scales with the scenario-based amplitudes. The EOF decomposition also has the potential to facilitate monitoring changes in spatial distributions based on the time series of the amplitudes (Figure 7.3.4b).

7.4 References

Fraser, H. M., Greenstreet, S. P. R., Fryer, R. J., and Piet, G. J. 2008. Mapping spatial variation in demersal fish species diversity and composition in the North Sea: accounting for species- and size-related catchability in survey trawls – ICES Journal of Marine Science, in press.

ICES. 2007a. Report of the working group on fish ecology (WGFE) ICES CM 2007/LRC:03 211pp.

ICES. 2007b. Report of Workshop on Testing the Entrainment Hypothesis (WKTEST). ICES CM 2007/LRC:10.

Kulka, D. W., L. Hendrickson, N. Bez, R. Schlitzer, J. Black, and M. R. Simpson. 2005. Report of the Workshop on Mapping and Geostatistical Methods for Fisheries Stock Assessment. *NAFO Sci. Coun Studies* 39, 1-43.

Mello, L.G.S., and Rose, G.A. 2005. Using geostatistics to quantify seasonal distribution and aggregation patterns of fishes: an example of Atlantic cod (*Gadus morhua*). *Canadian Journal of Fisheries and Aquatic Sciences*, *62*:659–670.

Preisendorfer, R. 1988. Principal Component Analysis in Meteorology and Oceanography. Elsevier, Amsterdam.

Rivoirard, J., Simmonds, J., Foote, K.G., Fernandes, P., and Bez, N. 2000. Geostatistics for Estimating Fish Abundance. Blackwell Science, Oxford. 206 pp.

Annex 1: List of participants

NAME	ADDRESS	PHONE/FAX	EMAIL
Ken Haste Andersen	The National Institute of Aquatic Resources, Department of Sea Fisheries, Charlottenlund Slot, Jægersborg Alle 1, 2920 Charlottenlund, Denmark	Tel: +45 33963375	kha@aqua.dtu.dk
Tom Blasdale	JNCC, Dunnet House, 7 Thistle Place, Aberdeen, AB10 1UZ, United Kingdom	Tel: +1224 655708 Fax: +1224 621488	tom.blasdale@jncc.gov.uk
Nick Dulvy	Simon Fraser University, Department of Biological Sciences, 8888 University Drive, Burnaby, BC V5A 1S6, Canada	Tel: +1 778 782 4124 Fax: +1 778 782 3496	nick_dulvy@sfu.ca
Daniel Duplisea (*Chair*)	Department of Fisheries and Oceans, Institut Maurice-Lamontagne, 850 Route de la Mer, Mont-Joli, QC G5H 3Z4 , Canada	Tel: +1 418 775 0881 Fax: +1 418 775 0740	duplisead@dfo-mpo.gc.ca
Jim Ellis	CEFAS, Lowestoft Laboratory, Pakefield Road, Lowestoft, Suffolk, NR33 0HT, United Kingdom	Tel: +44(0)1502 524300 Fax: +44(0)1502 513865	jim.ellis@cefas.co.uk
Michael Frisk	Stony Brook University, Stony Brook, NY 11794-5000, USA	Tel: +1 631 632 3750	mfrisk@notes.cc.sunysb.edu
Jakob Gjøsæter	Institute of Marine Research Flødevigen, 4817 His, Norway		jakob.gjoesaeter@imr.no
Simon P.R. Greenstreet	FRS, Marine Laboratory, 365 Victoria Road, Aberdeen, AB11 9DB, United Kingdom	Tel: +44(0)1224 295417 Fax: +44(0)1224 295511	s.greenstreet@marlab.ac.uk

NAME	ADDRESS	PHONE/FAX	EMAIL
Emma Guirey	FRS, Marine Laboratory, 365 Victoria Road, Aberdeen, AB11 9DB, United Kingdom	Tel: +44(0) 1224 295439	e.j.guirey@marlab.ac.uk
Ralf van Hal	IMARES, Haringkade 1, P.O.Box 68, NL-1970 AB Ijmuiden, Netherlands	Tel: +31 255 564694	ralf.vanhal@wur.nl
Remment ter Hofstede	IMARES, Haringkade 1, P.O.Box 68, NL-1970 AB Ijmuiden, Netherlands	Tel: +31 255 564686 Fax: +31 255 564644	remment.terhofstede@wur.nl
Dave Kulka	Department of Fisheries and Oceans, Northwest Atlantic Fisheries Centre, P.O. Box 5667, St John's, NF A1C 5X1, Canada	Tel: +1 709 772 2064 Fax: +1 709 772 5469	kulkad@dfo-mpo.gc.ca
Pierre Petitgas	IFREMER, rue de l'Ile d'Yeu, B.P. 21105, F-44311 Nantes Cedex 03, France	Tel: + 33 240374163 Fax: +33 240374075	pierre.petitgas@ifremer.fr
Anne Sell	Institute for Fisheries Research, Palmaille 9, D-22767 Hamburg, Germany	Tel: +49 40 38905 246 Fax: +49 40 38905 263	anne.sell@vti.bund.de
Alberto Serrano	Instituto Español de Oceanografía, Centro Oceanográfico de Santander, Promotorio San Martin, E-39080 Santander, Spain	Tel: +34 942291060	aserrano@st.ieo.es
Verena Trenkel	IFREMER, rue de l'Ile d'Yeu, B.P. 21105, F-44311 Nantes Cedex 03, France	Tel: + 33 240374475 Fax: + 33 240374157	verena.trenkel@ifremer.fr

Annex 2: List of working papers

Petitgas, P. Principal scales in fish spatial distributions and their use for scenario-based prediction, 8pp.

Serrano, A., Rodríguez-Cabello, C., Sánchez, F., Olaso, I., Velasco, F., Punzón, P. Effects of trawl exclusion in a set of indicators in the Cantabrian Sea inner shelf (southern Bay of Biscay), 60 pp.

Annex 3: Agenda

Work days begin 9:00 at ICES HQ

Lunch (12:30 – 13:30)

Day ends ~19:00

Monday, 3 March 2008:

Introduction to the group and its work

Terms of Reference for 2008

Personal introductions

Presentations

Tuesday, 4 March 2008:

Group and individual work

Plenary consultation on TOR (a) analysis

Wednesday, 5 March 2007:

Group and individual work

Plenary presentation of work and text

Thursday, 6 March 2007:

Group and individual work

Plenary presentation of work and text

Nominations and voting for new chair recommendation

Friday, 7 March 2007:

Group and individual work

Plenary presentation of work and text

Plenary on suggested TORs for 2009

Compilation of report

Annex 4: Proposed Terms of Reference for the next meeting

The **Working Group on Fish Ecology** [WGFE] (Chair: Dave Kulka, Canada – chair elect) will meet in as yet undecided location during an as yet undecided date.

WGFE will report in 2009 to the attention of the Living Resources Committee.

1) Explore the utility of a variety of modelling approaches for projecting, decline and recovery of fish community metrics to target indicators of state (e.g. EcoQOs) in response to fishing pressure and environmental variability

a) Explore the role of uncertainty (e.g. model, process, estimation, observation uncertainty) in these models in the context of probability of achieving targets

b) Evaluate the potential for and implications of fitting these models to data

c) Explore and update the list of modelling approaches from WGFE 2008, including qualitative modelling

d) Apply models to the North Sea EcoQO

2) Evaluate metrics to characterise, monitor and detect changes in the structure, function and productivity of fish communities

3) Further develop and explore mapping and other methods for comparing and summarising fish and fish community distributions in relation to environment and habitat

4) Under the umbrella of abundance-occupancy relationships and essential fish habitat:

a) Examine abundance-occupancy relationships within species, and groups of species in different ecosystems in relation to habitat, environment and anthropogenic impacts

b) Compare and contrast abundance-occupancy relationships and life history characteristics for species common to NW and NE Atlantic (e.g. cod, haddock, spurdog, thorny skate and herring)

5) Characterise the fish assemblages associated with offshore habitats (reefs, sandbanks, gas seeps) identified in the EU Habitats Directive.

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:	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:	EcoQO TORs result from request from OSPAR to ICES. Work comparing NW and NE atlantic species has bearing on NAFO, and NAFO data may be accessed to complete the proposed work on fish distributions in the North Atlantic.

Annex 5: Recommendations

RECOMMENDATION	ACTION
WGFE unanimously recommends Dave Kulka (DFO, Canada) as the next chair of WGFE for the next three meeting cycle (2009-2012).	
WGFE suggests that moving the meeting date to the autumn would be better timing for participants as there would be fewer conflicts with other working group demands than in the winter/spring. WGFE recommends that its next meeting be held in autumn 2009.	Discuss with WGECO chair if that group can get by without a full WGFE report in the spring of 2009

Annex 6: WGECO Technical Minutes

[To be inserted when available.}