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REPORT OF THE WORKING GROUP ON FISH ECOLOGY (WGFE)

13–17 MARCH 2006

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International Council for the Exploration of the Sea
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Contents

Executive summary

- The proportion of large fish (community abundance ratio of large fish to all fish) is a promising yet still untested (via simulation) ecosystem quality indicator (EcoQO). Management advice based on this EcoQO may preserve many desirable qualities of fish communities (e.g. trophic balance) but not necessarily all of them; therefore, the desired community qualities (management objectives) should be clearly stated in simulation testing to determine the utility of the proportion of large fish for preserving them.
- A clear community response to community exploitation rate is not obvious from empirical analyses. It may be also that exploitation rates vary with a community's potential to support fisheries.
- Simulation is perhaps the only tool currently available to adequately test the sensitivity, responsiveness and specificity of most community indicators.
- Geostatistical methods were explored for their ability to complement mapping methods in the examination of abundance-occupancy relationships (Section 3.4)
- Although WGFE feels that essential fish habitat issue are very important and a key work area for the group, it is still not clear how to define essential fish habitat in a context useful for management. General functional habitat categories have been described but it is not clear what kind of specific advice science should currently offer on protecting habitats fitting in one or more of these categories.
- WGFE recognises a need to liaise with other working groups to build upon activities in the areas of habitat mapping, with respect to both maps of abiotic habitat parameters and biotic components (e.g. WGMHM). These broadscale maps are fundamental to relating the distribution of fish to the distribution, structure and function of sea floor habitats, and for identifying important fish habitats.
- Because most WGFE work considers fish communities at the scale of entire seas and not coastal fish communities, it is not well equipped to deal issues related specifically to nearshore fish communities namely issues of the Water Framework Directive. Though WGFE has sometimes been successful at including relevant specialists from this domain into its meetings, it has proven difficult to maintain the expertise as most of the other work of WGFE is of limited interest to estuarine ecologists and their respective institutions. Therefore WGFE recommends that the Diadromous Fish Committee and/or Living Resources Committee give consideration to forming a group specifically to examine estuarine ecosystems and transitional waters. WGFE could then liaise with this group on questions related to EcoQOs.

Work done in 2006 against the 2006 Terms of Reference:

A. Ecological Quality Objectives (EcoQOs) for fish communities are required by OSPAR, and analyses on various size-based metrics have been undertaken by WGFE. A reporting guideline for surveys and data filtering for EcoQO indicator studies was developed to aid comparison of indicator studies (2.1). WGFE examined questions related to what is a large fish and the utility of a large fish indicator to meet probable management objective (2.2). Methods are outlined for constructing Multispecies F and community exploitation rates and an analysis of the latter with size spectrum indicators was conducted (2.3). Work on evaluating indicators in simulation environments was not carried out in the working group in 2006 but a clear plan exist for doing such and parameterisation of the OSMOSE model for doing this is well underway (2.4). Methods were explored for detecting rare species using survey data 2.5)

WGFE explored the possibility for primary publication and/or cooperative research reports. Group participants felt that several pieces of work from the present meeting could be adapted for publication in the primary literature and subgroups of workers with do this on their own schedule. It should also be noted that primary publications have already arisen out of work presented in WGFE reports most notably on community indicator studies. The ICES/SCOR ecosystem indicator volume of the ICES Journal of Marine Science (Vol 62 – 2005) contains several of these articles (Blanchard, Duplisea, and Dulvy).

B. Abundance-occupancy relationships were explored using mapping studies and geostatistical techniques on data ranging from the Barents Sea to the Iberian Peninsula (3).

C. WGFE did not work on the ToR related to gastric evacuation models and their impacts on mortality estimates derived from MSVPA. This work has now been taken up by the Study Group on Multispecies Assessment in the North Sea (SGMSNS) and the original participants in WGFE who worked on this topic now participate in SGMSNS.

D. Several upcoming nature conservation issues relevant to ICES were highlighted (7.2). The use of indicators in the context of the Water Framework Directive could not be addressed as we could not find appropriate experts willing to attend this year's meeting of WGFE.

E. Essential fish habitat issues were explored using mapping studies of the spatial distribution of juvenile stages of various North Sea species, suggesting that these may be considered nursery areas. Further analyses linked habitat to species via specific benthic prey of particular species and the potential use of ICES FishMap to uncover specificity of some species with particular areas (5)

E(i). Two length-based catchability studies were conducted to determine absolute biomass of North Sea demersal species and on a species basis for the French Thalassa survey (6).

F. IBTS data were well utilised in several sections of the report (e.g. 3, 5) to show spatial and temporal changes in fish distribution. The components of IBTS database was summarised (4). Useful methods were developed for the quantification of rare species and these were applied to rare elasmobranch species (2.5).

G. A roadmap for strategically focussing WGFE future work was developed (9). This was purposely not prescriptive in order to leave open the possibilities for incorporating any unanticipated pressing questions and developing techniques related to conservation of marine fish and aiding an ecosystem approach to fisheries.

1 Introduction

1.1 Terms of reference

The Working Group on Fish Ecology [WGFE] (Chair: A. Daniel Duplisea*, Canada) will meet at ICES Headquarters, from 13–17 March 2006 to:

- a) with regard to the development of EcoQOs for fish communities:
  - i) establish standardised protocols for filtering survey data to ensure that subsequent statistical analyses are comparable across a range of scales;
  - ii) liaise with other ICES Working Groups to collate a temporal series of fishing mortality rates for the main species of the assemblages to provide estimates of multispecies F at appropriate spatial scales;
  - iii) define what a 'large fish' is;
  - iv) evaluate how a suite of indicators change in relation to estimated trends in multispecies F;

v) use simulation tools to evaluate the sensitivity of various EcoQO indicators to multispecies F;

vi) undertake further studies for developing appropriate EcoQOs for threatened and declining marine fishes; Examine potential for publication of via CRR or Peer reviewed publications;

b) undertake further studies on the abundance-occupancy relationships in marine fishes, with special reference to fisheries and ecosystem management issues, and the underlying mechanisms that affect such relationships;

c) continue studies on food rations and prey composition of North Sea fishes by:

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

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

d) address any upcoming nature conservation issues for marine fishes including their value as indicators in the context of the Water Framework Directive;

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

f) obtain better estimates of relative catchabilities (commercial or RV) of marine fishes, on a size-specific basis when appropriate in collaboration with FTFB;

g) liaise with IBTS to continue studies on the broadscale spatial and temporal patterns in selected fish species and communities along the European continental shelf of the eastern North Atlantic (e.g. the area covered by parts of ICES divisions VI–IX). Cross cut with ACFM groups and WGRED, SGRESP. Liaise with WGEF on identification and quantification of rare shark species.

h) develop a road map for strategically focussing on future work of the group.

WGFE will report by 30 April 2006 for the attention of the Living Resources, the Resource Management, the Diadromous Fish Committees, as well as ACE.

1.2 Participants

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

Julia Blanchard (CEFAS, UK)
Tom Blasdale (part-time) (JNCC, UK)
Niels Daan (RIVO, Netherlands)
Nick Dulvy (CEFAS, UK)
Daniel Duplisea (DFO, Canada)
Jim Ellis (CEFAS, UK)
Helen Fraser (FRS, UK)
Ronald Fricke (SMN, Germany)
Concepción González Iglesias (IEO, Spain)
Edda Johanessen (IMR, Norway)
Dave Kulka (DFO, Canada)
Anne Sell (BFA-FI, Germany)
Yunne Shin (IRD, France)
Remment ter Hofstede (RIVO, Netherlands)
Verena Trenkel (IFREMER, France)

1.3 Background

The Working Group on Fish Ecology first met in 2003 (ICES, 2003). The rationale behind the formation of the group was to support ICES on issues of fish community metrics and to provide advice on threatened marine fishes. OSPAR and HELCOM had requested advice in these areas from ICES, and ICES had been unable to respond. Until 2002, fish community issues were considered by WGECO, but as the demands on WGECO increased the establishment of WGFE enabled a more focussed consideration of fish community issues. WGFE met again in 2004 and 2005, and continued ecological studies, including the development and testing of Ecological Quality Objectives (EcoQOs) for fish communities, abundance – occupancy relationships, and the relative catchability of fishes in different survey gears, evaluation of decline criteria used by various conservation organisations (ICES, 2004; ICES, 2005). WGFE has addressed issues on non-commercial fish species, including species of conservation importance, fish communities and assemblages, and other aspects of fish ecology (e.g. feeding habits and prey rations, habitat requirements), so that ICES can provide advice in these areas in relation to ecosystem, biodiversity and nature conservation issues.

1.4 References

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

2 Development of Ecosystem quality objectives (EcoQO) for fish communities

2.1 Reporting protocols for surveys and data use

One of the main problems with comparing fish community indicator studies is the reporting of details on survey design and data treatment. Without explicit documentation of what has been included and excluded in the analysis, it is difficult to determine if studies are comparable and therefore if the indicators are reflecting the same general properties of the communities in each study.

Though we advocate that various surveys follow as common a methodology as possible in order to sample the fish community with some degree of certainty, the purpose here is not necessarily a call for consistency between surveys but that workers report the details of their surveys and analyses in any fish community indicator study. Accordingly, we outline a list of questions that should be asked and *then reported* in the methods sections of any community indicator report (Table 2.1-1).

Table 2.1-1: Interrogative reporting guidelines for surveys and data filtering in studies of fish community indicators.

SURVEY DESIGN

- What is the survey design (e.g. fixed station, stratified random) and were there changes, when?
- What gear is used and have there been changes, when?
- What defines a standard haul and were there changes, when?
- What is the mesh size of the most relevant gear sections (e.g. codend)?
- Have filling-in methods been used to correct for missed stations, what?
- Are there hauls, stations, areas, years that have not been used and why?
  Are there particular species or groups of species not well caught (e.g. pelagics)?
  Does the survey cover the majority of the area of the populations sampled and used in analysis?

SPECIES FILTERING

- What is the accepted species list and comparative abundance (e.g. average % biomass over time series)?
- Which species have been excluded and why?
  Have any corrections been applied to species or species groups (e.g. catchability or taxonomic issues). If so what?
- Are there particular species that have dominated the catch, when, where?

SIZE FILTERING

- Which sizes and body types are well caught and not well caught by the survey?
- Which size cut-offs have been used, smallest and largest?

2.2 Proportion of large fish indicator

OSPAR considers that a complete system of ecological quality objectives has the ability to help to provide a practical, scientifically based and consistent method to implement the ecosystem approach to the management of human activities affecting the marine environment.

An Ecological Quality Objective (EcoQO) is the desired level of an ecological quality, preferably set in relation to a reference level. There will be a one-to-one relationship between ecological quality elements and ecological quality objectives. The desired level of ecological quality will be set in relation to a metric that can be objectively verified. One of the key issues concerned fish communities and the proposed element included, "Changes in the proportion of large fish and hence the average weight and average maximum length of fish community".

This indicator has been considered in previous WGFE reports and the previous approach has been to calculate the index for a variety of fish assemblages. A summary of findings is that there are ranges of ways in which this index can be measured and different measures may capture or represent varying ecosystem attributes and thus may inform and be used to monitor progress toward a variety of ecological objectives. The choice of indicator depends on the ecological objective and therefore indicator selection cannot be considered without any objective.

Here we seek to synthesise this knowledge; (1) by identifying a range of ecological objectives which could potentially be informed or monitored by a large fish index (which might include the proportion of large fishes), (2) by summarising the different measures of large fishes (3) mapping large fish measures against the appropriate objective and (4) by evaluating each measure against indicator selection criteria (Rice 2000).

The approach taken was to compare a range of indicators and a range of plausible ecosystem objectives. For the sake of brevity it was necessary to make comparisons of broad classes of indicator and ecosystem objectives and there is clearly scope to undertake a more exhaustive analysis. We sought to identify as many as possible indicators that may represent a range of definitions and attributes of large fish. This approach was taken to facilitate among indicator comparisons in terms of their attributes and their relevance to the range of ecosystem objectives. This may be a more productive exercise as and when detailed hierarchical ecosystem and operational objectives are defined and when ecosystem simulation tools become available to evaluate critical properties such as sensitivity, responsiveness and specificity.

2.2.1 Ecological objectives and a large fish index

2.2.1.1 Size, trophic structure and predatory function: large fish as large predators

Food web processes in marine ecosystems are strongly related to size. The principal primary producers are small unicellular algae, and these support size-structured food chains, in which most predators are larger than their prey (Pope *et al.*, 1994). Trophic level is therefore expected to increase with increasing size. Recent studies using nitrogen stable isotope ($\delta^{15}N$) as an index of trophic level have shown that $\delta^{15}N$ increases with the size of fish (Badalamenti *et al.*, 2002; Jennings *et al.*, 2002a, b), and of marine organisms in general (Fry and Quinones, 1994; France *et al.*, 1998). These results are consistent with the view that predator-prey relationships lead to powerful size-based trophic structuring. This may be seen at both across species and also within species. The lifespan of individual fish, because body mass may increase by five or more orders of magnitude (Cushing, 1975), and a species may begin life as prey, only to become the main predator on those species that it suffered from within its first year of life (Boyle and Boletzky, 1996; Köster and Möllmann, 2000). We will therefore try to specify what a "large fish" is as a predator, at both the species and the assemblage levels. We assume that a measure of the number or proportion of large fish within a system reflects changes in the trophic structure of fish communities.

As species mostly interact through predation, the existence of top-down control, which means the regulation of lower food-web components by one or several upper-level predators, should be critical in the functioning of marine ecosystems. Predation is estimated to be the major source of mortality for marine exploited species, even when compared to mortality caused by fishing. An analysis of six commercially exploited marine ecosystems (Benguela Current, Georges Bank, Balsfjord, East Bering Sea, North Sea and Barents Sea) suggested that predation represents between two and 35 times fishing mortality (Bax, 1991). It is therefore to be expected that the removal of predators (through fishing) will reduce the natural mortality of smaller fish otherwise caused by these predators and thereby additionally increase the proportion of smaller (prey) fish. Top-down control is diffuse in marine fish communities and may operate through multiple weak trophic interactions because of opportunistic size-based predation. This has been proved theoretically to favour stability, i.e. to dampen natural fluctuations of forage species (McCann, 2000; Shin and Cury, 2001).

In summary, large fish are important to monitor because they may have top-down effects on the rest of the trophic pyramid. Furthermore, they usually have higher longevity and longer turnover times and so integrate across medium and long-term impacts of fishing, combined with environmental effects such as climate change.

2.2.1.2 Assemblage reproductive capacity

Large and small fish are relative concepts that entail setting of arbitrary criteria. What should be considered large may vary with gear used, but also spatially and temporally. However, one may also look at the large-fish concept in a species-specific context. A 100 cm cod may be considered large, but is still small compared to whale shark that is already larger just after birth. Similarly, a Norway pout of 25 cm is a really large pout! For any individual species, one might set criteria that distinguish small, medium and large specimens. Rather than setting arbitrary criteria, these might also be linked to biological features such as young of the year, sub-adults and mature fish. Whatever criteria are used, variations in the proportion of different groups are likely to reflect population dynamic processes, particularly variations in total (natural + fishing) mortality. Consequently, such species-specific large-fish criteria might be used to develop criteria by which for instance the reproductive potential of the species constituting an assemblage sampled by a trawl survey might be judged. Because this clearly would have some bearing on the health of that assemblage, this type of approach would seem an attractive line to follow in trying to identify adverse effects of fishing on entire fish communities and in setting management objectives for restoration.

2.2.1.3 Conservation of threatened and declining species

A subset of wider biodiversity includes the conservation or management of threatened and declining species. Note that this is already dealt with within EcoQO framework in Ecological Quality Issues of “threatened and declining species”. This issue is best dealt with on a species-by-species basis, e.g. using World Conservation Union (IUCN) threat and decline index (Dulvy *et al*., in press; WGFE 2005), rather than using an aggregate assemblage level index such a large fish index. When considering threat or decline each species needs to be addressed separately considering its individual biology and needs.

While the decline or absence of large adult individuals would be of interest, a more direct indicator of threatened and declining species may more usefully represent this. It was noted that threatened and declining fish species may benefit from measures to protect large fishes. For example, the maximum size of specimens often depends on fishing pressure (example Porbeagle *Lamna nasus*, which matures at an individual age of 14 years and 1.2 m length with a very low reproduction rate, and in the past used to reach an age of 46 years and a maximum length of 3.5 m, but today at most 25 years and 1.8 m length). The absence of large adults in a population of threatened and/or declining species may be due to threats and may lead to an accelerated decline of the species.

2.2.1.4 Wider biodiversity

Biodiversity in its broadest sense is the quantity, variety and distribution of genes, populations, species, habitats and ecosystems. So there may be interest in defining and setting ecological objectives for some aspects of biodiversity, such as seabird and mammal populations. One aspect of biodiversity already covered with the OSPAR EcoQO framework is Ecological Quality Issues of “threatened and declining species”. The issue is then one of whether an index of large fishes captures any other attributes of biodiversity in addition to threatened and declining species. The group did not consider that a large fish index could capture any additional biodiversity attributes, especially where the biodiversity attribute is not size-dependent. Although it was noted that richness indices alone do not capture size structure. The use of richness indices would not be able to distinguish between an unexploited assemblage with full size / age structure or an exploited assemblage with the same richness but truncated size / age structures.

2.2.1.5 Charismatic species

An index of large fish might be able to capture the status of large charismatic species which are newsworthy and thus of societal interest. An ecosystem objective may be to increase or maximise the quantity, variety and distribution of charismatic species. Charismatic species include both large and small-bodied species. The large-bodied species might include: sturgeons, tunas, salmon, sharks and rays. The smaller-bodied species might include: European eel (because of interest in its enigmatic migration and lifestyle), seahorses and pipefishes (*Syngynathidae*), and shads (*Alosidae*). There are two reasons why an index of large fish may not be appropriate for evaluating the status of charismatic species with respect to an ecological objective: first, any ecological objective for charismatic species is more usefully framed in terms of individual species rather than as an assemblage or community attribute. Second, a number of charismatic species are small and would not be represented by any large fish index. Instead charismatic species may be better represented within ecological objective for threatened and declining species.

2.2.2 Methods of calculating a large fish indicator

There are a number of issues to consider when developing an index of large fishes appropriate to trophic and predatory objectives: (1) the application of an absolute or alternatively a proportional indicator, and (2) the method of calculating a proportion. A large fish indicator can be expressed in absolute terms or relative to smaller species or size classes and the choice of which is used depends on the ecosystem objective. Broadly the change in proportional indicators can be due to either changes in large fish abundance OR changes in the abundance of smaller fishes. Thus, the same size index value can be achieved EITHER by focussing management on small OR large size classes. Therefore it is more difficult to link proportional indices to appropriate management responses without unpacking the index into the absolute values of large and small fishes. In addition to this issue the absolute abundance of large (predatory) species or individuals may be more relevant to an objective pertaining to predatory control in the system, rather than a proportional index. There are two ways of calculating proportional indicators and the choice of which to use may depend on the objective. In each case, the proportions of size classes are first calculated on a species level then averaged across species. This latter step can be done in two ways, (1) by taking a weighted average, based on the abundance or biomass fraction of each species in the total assemblage. Method (2) would apply a non-weighted average in order to give each species the same impact and prevent very abundant species from dominating the index.

In further considering large fish indicators, we will not include methodology intended for the conservation of threatened and declining species, wider biodiversity and "charismatic species". We consider these issues much better addressed by species-based assessment of abundance and distribution, and direct conservation measures for the species and their habitats.

In Table 2.2-1 a range of different large fish indicators are outlined, along with definitions of what a large fish is, whether it can be considered or applied at a species-by-species basis or for the whole assemblages. All of these indicators assume that species and/or size disaggregated research survey catch rates are available. In addition we note additional data requirements. Below we have briefly summarised additional details on each method.

Table 2.2-1: Large fish indicators, definition of large fish, scale of application (species and or assemblage) and additional data requirements or recommendations.

INDICATOR	DEFINITION OF "LARGE" FISH	SPECIES- OR ASSEMBLAGE-BASED	ADDITIONAL INFORMATION REQUIRED
Demersal / pelagic ratio	Assumes demersal fish abundance or biomass represents large fishes	assemblage	Explicit list of demersal and pelagic species, does not require size information
Proportion of large fish and/or (numbers of large fish and small fish)	Use percentiles to statistically define large fish	species/assemblage	Length data for all species included in analysis. No detailed biological information needed
""	Use arbitrary cut-offs, e.g. 20, 30, 40 cm to statistically define large fish	species/ assemblage	Length data for all species included in analysis. No detailed biological information needed
""	Use biologically relevant cut-off (e.g. length-at-maturity)	species/ assemblage	Length at maturity data for all species included in analysis
Proportion of piscivorous fish	Piscivorous species	assemblage	List of the predominantly piscivorous species within the assemblage
""	Piscivorous individuals	species	Body length of ontogenetic switch to piscivory for the species considered
""	Piscivorous individuals of all species	assemblage	For each species in the assemblage, calculation of number/proportion of piscivorous individuals – thereafter average the proportion or sum numbers across species
Abundance/biomass of large keystone species	Assumes large species are more likely to include those with disproportionately large interaction strengths	species	Some defensible method for identifying keystone species

2.2.2.1 Demersal to pelagic fish ratio

This method assumes that demersal fishes are larger-bodied and fed at higher trophic levels than smaller bodied pelagic fishes. Note that the nature of this ratio means that the index may have low specificity to fishing effects.

2.2.2.2 Proportion of large fish and/or [numbers of large fish and small fish]

This approach suffers from the problem that there are a variety of methods of defining size thresholds. There is no right or wrong threshold and as yet we have little defensible method for selecting an appropriate threshold. Size thresholds can be defined statistically, arbitrarily or biologically. Two of the methods (statistical and biological thresholds) appear slightly more objective, defensible and potentially comparable across systems.

Statistical thresholds have used percentiles, such as 60, 85 and 90%, or the upper quartile. The percentiles are calculated based across the whole time series. The higher the quartile used to define “large” fish results in fewer individuals and thus lower signal to noise ratio. The lower percentile used the higher the likelihood that the index will include juvenile individuals and may thus be biased by recruitment variation. (Thus the only way to choose the appropriate percentiles requires scrutiny of the data, which may then reduce the objectivity of the approach, and wider comparability across systems).

The biological threshold approach would use some method of discriminating important life history stages, such as the length at which individuals of each species mature. Length at maturity can be calculated using relatively sophisticated methods such as histological examination to develop maturity ogives or simple methods based on observed life history invariant or dimensionless ratios (Beverton 1967; Charnov 1993). Across species fish typically mature at two thirds the maximum size. Maximum size can be derived from species catalogues or databases and maximum size should not be taken from that observed in surveys – which are likely to be considerably lower than the historic maximum size. The later approach is more applicable across a wide range of target and non-target species in the assemblage.

The final approach uses an arbitrary size value to define large fish, this is not defensible and consequently we do not see value in this approach.

2.2.2.3 Proportion of piscivorous fish

This indicator assumes that many large fish exist at high trophic levels (Jennings *et al*. 2002a) and thus an index of large fish may capture the degree of top-down predatory control in a system (Dulvy *et al*., 2004). A key problem for the calculation of this index is the measurement of the degree of piscivory within and among species. The index can be calculated in three ways: (1) by species for an assemblage level indicator, (2) by individual for a species level indicator and (3) by individual for an assemblage level indicator. Piscivorous species can be defined on the basis of stomach contents or behavioural observations, and is typically available from faunal reports, mass balance modelling (e.g. ECOPATH) or species databases (e.g. FishBase). If the majority of the diet of adult consists of fish then the species will be considered as piscivorous. Many individuals within species begin life feeding on plankton and consume larger higher trophic level prey as they grow and gape size increases throughout life (Karpouzi and Stergiou 2003; Trenkel *et al*., 2003). Defining piscivory within species is much more data intensive and requires some form of stomach contents and /or stable isotope analysis by size class.

2.2.2.4 Abundance/biomass of large keystone species

This indicator assumes large fish may also include those with disproportionately strong interaction strengths (keystone species) and may control trophic cascades and primary producer community structure and function. That a keystone species indicator was considered does not reflect a view that there may be keystone species within the OSPAR / ICES area, moreover that this may well be a property relevant to large species somewhere. The key difficulty is that, under a strict definition of keystoneness, the indicator requires the direct measurement of interaction strength across species and size classes and preferably across time. One crude shortcut would be simply to infer keystoneness of individual species from time series of ecosystem change (e.g. Carscadden *at al*., 2001, Frank *et al*., 2005).

2.2.2.5 Assemblage reproductive capacity

For most commercial species there are good data on age and size-at-maturity that can be used to split survey catches in a mature, adult component and a sub-adult component. However, for the majority of the non-commercial species such information is lacking and therefore proxies cannot be avoided. The only information available for all species is the maximum size recorded. Two pieces of information are required a definition of large and small fish expressed as a proportion of L_{max} or L_{∞}.

One approach is to use dimensionless ratios or life history variants to estimate thresholds of maturation for the range of species of interest. Beverton (1963) showed that on average most species mature at approximately 60% of their L_{∞}. Reported L_{max} is typically 10% less than L_{∞} (Froese and Binohlan 2000), which would suggest that 50% might be a suitable criterion. For many of the non-commercial species the lower end of the length compositions in survey catches will be affected by the size selectivity of the gear. In addition, recruitment variations may strongly affect the proportions of large vs. small fish. Therefore, it would seem sensible to set a lower limit of 30% of the L_{max} to separate sub-adults from juveniles and to concentrate on the ratio of fish above 50% of their L_{max} and those that are within 30 and 50% of their L_{max}.

Although the length compositions might be standardized in terms of their L_{max}, it would not seem appropriate to sum the numbers caught in each category over all species, because it that case the overall index would simply reflect the patterns in the most dominant species (e.g. in the North Sea herring and sprat) and would tell very little about the fish assemblage in general. Therefore, it would seem more appropriate to calculate ratios for individual species and average those (or counting the percentage of species not conforming to a preset condition).

To be sensible, such an analysis should be restricted to those species: (1) that are sufficiently abundant to come up with a reasonable estimate of the large/medium ratio and (2) that both feed and reproduce within the survey area.

2.2.2.6 Reference points

For estimating directions of change, time series analyses of existing data may be applied using a variety of different criteria. However, it would of course be extremely helpful if reference levels could be specified independently. There seems to be an option here, because the ratio of mature vs. adults is somehow linked to the spawning stock biomass criterion developed for commercial species in providing TAC advice (B_{lim}). A reference B_{lim} could be set by comparing the SSB trend with the indicator trend in the ratio of adult to sub-adult fish in the assessment. Because the ratio of adult versus sub-adult fish should be a function of total mortality, it seems quite possible that reference levels may have to reflect differences in natural mortality among species.

The development of this indicator is at an early stage and considerably more work is required before the utility of such an approach can be evaluated.

2.2.3 Mapping large fish measures against the appropriate objective

The broad vision of sustainable development of the sea could be supported by high-level ecological objectives of the desired state. Indicators, with appropriate target and / or limit reference points, are required to track the progress toward or away from the ecological objectives.

Here we outline some broad categories of ecosystem state objectives and map the large fish indicators against each. We scored the degree to which each indicator is relevant to the state of each ecosystem objective, from high to low, according to the relevance of each large fish indicator. The best indicators are those that have high relevance to the objective (Jennings 2005; Rice and Rochet 2004). These scores are arbitrary, based on group discussion and thus they are easy to criticise, however it was not clear to the group that there are other less contentious alternative scoring approaches. The scores can also be viewed as an average of the individual scores of the variety of different methods of calculation for each broad category of indicator. This exercise is illustrative and could be undertaken with a different and or more detailed set of ecosystem or operation objectives. The ecosystem state objectives included the maintenance or restoration of (1) trophic structure, (2) predatory function, (3) reproductive capacity, (4) threatened and declining species, (5) wider biodiversity and (6) charismatic species. As noted above, the latter three objectives will not be considered further here, because there are other indices that are more promising than an index of large fish.

The main finding is that indicators of large fish such as proportion of large fish and the components (absolute numbers of small and large fish) have moderate (medium) relevance to a wide range of ecosystem objectives (Table 2.2-2). The proportion of large fish and the numbers of large and small fish was of only medium relevance to four ecosystem objectives: maintenance and restoration of trophic structure, predatory function, reproductive capacity and threatened and declining species. The large fish indicator was most relevant for the size structure ecosystem objective. For all of these ecosystem objectives other indicators exist that may be more directly linked than the proportion of large fish indicator:

ECOSYSTEM OBJECTIVE TO MAINTAIN OR RESTORE….	INDICATOR
Trophic structure	Demersal/pelagic ratio
Predatory function	Proportion/absolute abundance of piscivorous fish
Reproductive capacity	Proportion/absolute abundance of mature fish
Threatened and declining species	Threat index (Dulvy *et al.* in press; ICES WGFE 2005)

These scores may shift depending on critical detail of the ecosystem and objectives of interest. For example an indicator of keystone species may have higher relevance to ecosystem objectives to maintain or restore charismatic species or threatened and declining species if the charismatic or threatened species exhibited some keystone properties. It is also implicit that we are only considering predatory keystone species explaining the high relevance to an ecosystem objective to maintain and restore predatory function. We recognise that keystone species may exist at other trophic levels, and thus the mapping to ecosystem objectives would need revisiting.

Table 2.2-2: Large fish indicators and their relevance for a range of potential ecosystem objectives.

	POTENTIAL ECOSYSTEM OBJECTIVE: TO MAINTAIN OR RESTORE						
Indicator	size structure	trophic structure	predatory function	reproductive capacity	threatened and declining species	wider biodiversity	charismatic species
Demersal to pelagic fish ratio	Medium	High	Medium	Low	Low	Low	Low
Proportion of large fish	High	Medium	Medium	Medium	Medium	Low	Low
Numbers of large and small fish	High	Medium	Medium	Medium	Medium	Medium	Low
Proportion of piscivorous fish	Low	Medium	High	Low	Low	Low	Low
Abundance/biomass of large keystone species	Low	Medium	High	Low	Medium	Medium	Low
Assemblage reproductive capacity	Medium	Medium	Medium	High	Low	Low	Low

2.2.4 Evaluation of indicators against selection criteria

The group used its expertise to evaluate the usefulness of the large fish indicators, according to the selection criteria that were proposed by the SCOR/IOC Working Group 119 (Rice and Rochet 2005). The main conclusions of this exercise are that (i) that it was not possible to make an *a priori* assessment of the sensitivity, responsiveness and specificity properties of all these indicators, which we consider essential, and (ii) the selection criteria considered here were not very discriminating although demersal/pelagic ratio and abundance of keystone species ranked lower than the other indicators (Table 2.2-3).

Recent work has suggested that it may be erroneous to attribute *a priori* reference directions of change to ecosystem indicators as these seem to be strongly case dependent (Travers *et al*. 2006). Both ecosystem functioning and fishing schemes will change the direction of change of ecosystem indicators in a non-linear way). The Working Group then recommends undertaking both model simulations to evaluate the sensitivity of indicators, and tree decision analyses which combine a set of complementary ecosystem indicators.

The only discrimination that we can make *a priori* is between proportion- and abundance-based indicators according to their specificity to fishing. All ratio indicators (e.g. proportion of large fish, proportion of mature fish) will not only reflect the abundance of large fish but will also be influenced by the abundance of small fish (particularly by strong recruitment) or small species of a community, especially because small fish are in general more abundant than larger fish. In addition, small fish have a fast turnover rate and have less resistance to their environment so their abundance is more likely to have short term responses to variations of hydroclimatic and food conditions, whereas large fish of a population or large species of a community are more likely to respond more specifically to fishing effects. Therefore, proportion indicators may be *a priori* less specific to fishing than the absolute numbers of large fish. Proportion indicators provide more synthetic information on the balance between different functions in the population or ecosystem (demersal versus pelagic fish, SSB versus recruitment), so that both types of indicators are complementary for avoiding misleading interpretation of the trends.

One selection criterion which could be considered when evaluating the usefulness of the indicators is to consider the comparability across ecosystems. According to this criterion, the proportion of large fish, whatever the size cut-off chosen, has the advantage to be dimensionless compared to the absolute numbers of large fish. There may therefore be cases in which a combination of both, the relative and the absolute numbers would best be used in combination to allow both, intra- and inter-system comparisons.

Table 2.2-3: Evaluation of large fish indicators using selection criteria.

				SELECTION CRITERIA				
Indicator	Concreteness	Theoretical basis	Public awareness	Measurement	Historical data	Sensitivity	Responsiven-ess	Specificity
Demersal/pelagic ratio	High	Medium	Low	Medium	High	?	?	?
Proportion/# of large fish – percentile	High	High	High	High	High	?	?	?
Proportion/# of large fish – arbitrary	High	High	High	High	High	?	?	?
Proportion/# of piscivorous fish	Medium	Medium	Medium	Medium	Medium	?	?	?
Proportion/# of mature fish	Medium	High	High	Medium	Medium	?	?	?
Abundance / biomass of large keystone species	Low	Medium	Medium	Medium	Medium	?	?	?

2.2.5 Concluding remarks

The approach we have taken is easy to criticise and imperfect however a key value of the exercise was to at least make explicit that there may not be one indicator that meets all selection criteria or necessarily have relevance or specificity to single ecosystem objectives.

The proportion of large fish indicator is intuitively useful and has many desirable properties and can be used to monitor a broad array of ecosystem objectives. Some critical issues remain that must be resolved before if this indicator can be considered for operational use including: reference points and directions, sensitivity, responsiveness and specificity. While intuitively having more large fish is better there is little basis for setting limit and target reference point for such an indicator. However, it is likely that reference directions can be defined. The issue of setting appropriate size cut-offs is currently unsatisfactory and the problem has persisted throughout the long period of evaluation of this indicator. "There is no theory that could predict what kind of average weight or average maximum length might be obtained in a specific survey for a specific reduction in exploitation rate of the fish community, let alone what kind of values might be expected in a non-exploited system. The only relevant information is the empirical relationship between any metric and available estimates of community exploitation during the period a survey has been carried out systematically. Even if the correlation is statistically significant, the relationship may reflect delayed responses of the fish community, because community metrics integrate effects over several years of change in exploitation superimposed on annual (random) variations in recruitment to all species in the assemblage sampled in the survey gear. For these reasons, the predictive value of any empirical relationship is very limited, while extrapolations outside the observed range of values are not warranted. Thus any sensible reference level should be within the observed range. Given that none of the available surveys extends into periods when communities can be considered as unexploited, the reference level could only indicate the state of an exploited ecosystem and therefore, should be used as a limit reference level." (ICES, 2003).

This problem may be eased as the ecosystem and operational objectives are defined. The use of ecosystem simulation and management evaluation frameworks have the potential to evaluate the sensitivity, responsiveness and specificity of the various possible methods of setting size cut-offs above which fish are considered large (Fulton *et al*., 2005).

The moderate relevance of the proportion of large fish indicator to a wide range of potential ecosystem objectives would suggest, at a first glance, that it is a useful indicator. Indicators tightly linked and specific to a particular ecosystem objective may have greater value because they will be easier to operationalize in any management framework. There are other large fish indicators that have high relevance to each of those same objectives (Table 2.2-1). So while the proportion of large fishing index may have broad general value this should not preclude a search for and development of indicators more tightly linked to ecosystem objectives.

Overall, WGFE feels that (i) the management objectives have to be clearly set so that they can be tightly linked to appropriate "large fish"-derived indicators (ii) once the objectives set, there is a strong need to assess the sensitivity, responsiveness and specificity of those indicators. Unless these steps are not achieved, it is not as yet neither appropriate to implement a proportion of large fish index as part of an EcoQO, nor to define a global North Sea reference level for management. This statement applies to most of the ecosystem indicators that were proposed by the SCOR/IOC WG 199 (2000-2004, www.ecosystemindicators.org).

2.3 Community exploitation and indicator response (Multispecies F)

Under ToR a), three different items refer to multispecies F:

(ii) liaise with other ICES Working Groups to collate a temporal series of fishing mortality rates for the main species of the assemblages to provide estimates of multispecies F at

appropriate spatial scales; (iv) evaluate how a suite of indicators change in relation to estimated trends in multispecies F; (v) use simulation tools to evaluate the sensitivity of various EcoQO indicators to multispecies F.

These items are interrelated and this chapter addresses the various aspects.

Fish communities are directly influenced by all fleets removing part of the biomasses of the constituent species, and indirectly by subsequent responses of individual species to changes in interactions. Thus, all fisheries combined determine the exploitation rate of the community. However, one fishery may affect the community much more than another, for instance because of the number of species in the catch or because the target species occupies a key position in the foodweb. Thus, it is not straightforward to come up with a suitable measure of the exploitation rate at the community level that may be correlated with indicators of change.

There are essentially two ways in which a trend in community exploitation rate may be derived:

<u>Averaging single-species F-values.</u> Exploitation is targeted on the relatively small number of species that are assessed on a regular basis. Therefore, exploitation of the entire fish community must be somehow be related to the trends in F observed in commercial species. However, it is not directly obvious how a suitable common trend may be derived, because mixed fisheries may exert F on different species simultaneously and thus there is a danger of double counting the same effort. Also, in the averaging process, F on an abundant species may be considered to contribute more to community exploitation rate than F on a less abundant species, suggesting that weighting by biomass might be appropriate.

<u>Removal rates (removals/standing stock biomass).</u> Another measure of community exploitation that might be considered is the rate at which fish are removed from the system by the fisheries relative to the exploitable or total biomass. Removals may be estimated from landings statistics and discard estimates, but absolute biomass estimates (for what they are worth) are only available for the assessed commercial species. Therefore, this approach very much depends on the availability of survey estimates of biomass of the various components of the assemblage considered based on area trawled corrected for catchability.

This Section addresses these two approaches in two specific case studies: the trend in multispecies F in the North Sea based on MSVPA (Daan *et al.*, 2005) and one based on removal rates in the Eastern Scotian Shelf using landings statistics and research vessel surveys. We then make a preliminary comparison between the trends derived from applying the two methods for the North Sea, using a limited set of estimates for the demersal fish community.

Finally, the last section describes how simulations may be used to evaluate the sensitivity of various indicators to multispecies F (MSF).

2.3.1 MSF North Sea based on MSVPA

2.3.1.1 Data and methods

The North Sea MSVPA (ICES, 2005a) provides a coherent set of average fishing mortalities (taking into account interspecific predation) for the fully exploited age groups of the 10 main commercial species over the period 1963–2003. These species assessed can be divided in 4 groups: roundfish (cod, haddock, whiting, saithe), flatfish (plaice, sole), pelagics (herring) and industrial (Norway pout, sandeel, sprat). Although there is some overlap between roundfish and flatfish fisheries, these four groups are largely exploited in different fisheries. Thus, as a first step it would seem appropriate to estimate a MSF for each group separately.

Assessments provide population estimates with high precision but with low accuracy in absolute terms, because of uncertainties in catch statistics, poor estimates of discards and unverifiable assumptions of various sources of natural mortality. Therefore, F estimates are considered to provide a reasonable reflection of the relative trends over time rather than absolute trends. In the context of indicators of change in the fish community, we are primarily concerned in trends in MSF and therefore F estimates for individual species were first standardized by division by the long-term mean. Then the mean MSF by group was obtained by simple averaging (type a). As an alternative, a weighted mean MSF by group (type b) was derived using the average SSB over the entire period as a weighting factor. The average SSB was chosen rather than the annual value, otherwise a stock with a low SSB in a particular period owing to overexploitation would get a lower weight than during a period when it was exploited less. It might have been preferable to take the average exploitable biomass as a weighting factor, but this is not readily available from the MSVPA output. The total stock biomass appears to be less suitable because it includes large amounts of 0-group fish that are not subject to exploitation.

At the community scale we would not expect to see responses to the specific exploitation rate in a particular year, but rather to the trend perceived over the preceding period. Thus, some smoothing seemed appropriate. To come up with an overall exploitation rate for the entire community, we used the annual average of the standardized indices, both for type (a) and (b) MSF, because selecting weighting factors for fisheries exploiting completely different components of the fish community would seem completely arbitrary.

2.3.1.2 Results

Figure 2.3-1 provides (a) the unweighted and (b) the SSB-weighted MSF for each of the four groups based on the most recent MSVPA estimates of F by species (note that the plot for pelagics is exactly the same because only one species is represented. Surprisingly, the weighting does not have a major effect on any of these group estimates. Also the trend in overall mean does not differ among the two types (Figure 2.3-2). Overall, there seems little to gain in pursuing the biomass weighted approach any further, because the differences are small.

The pattern observed in the overall MSF suggests that community exploitation rate approximately doubled between 1963 and 1985 and declined by approximately one third subsequently. In principle, this should provide enough of a signal to elicit an indicator response.

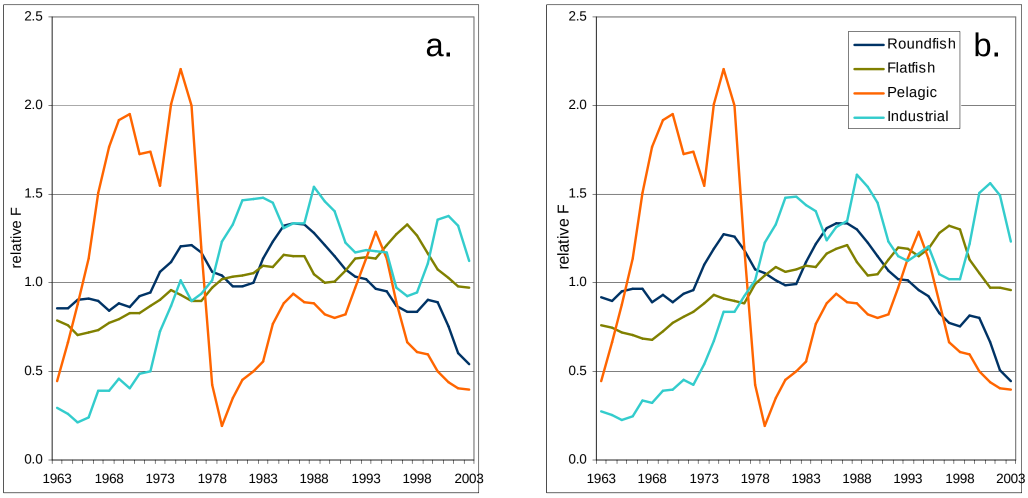

Figure 2.3-1: (a) Unweighted and (b) SSB-weighted standardized indices of MSF (3-year moving averages) for 4 major groups of North Sea fish species based on MSVPA (ICES, 2005).

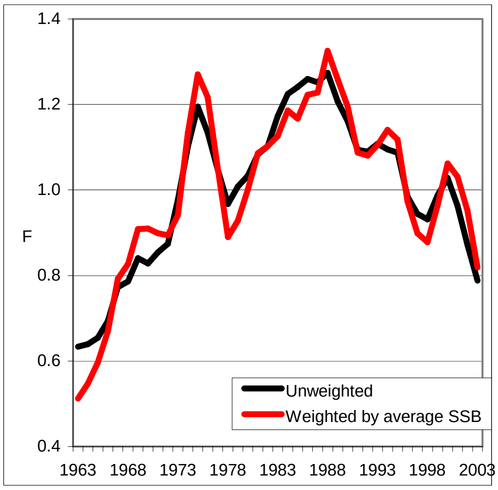

Figure 2.3-2: Comparison of overall MSF for the entire North Sea fish community based on unweighted and SSB-weighted indices for specific groups.

2.3.2 Community removal rates and size spectrum indicators: a case study from the eastern Scotian Shelf

In this case study, we have taken a time series of indicator values for the eastern and western Scotian Shelf (Duplisea and Castonguay 2006), and compared with a time series of multispecies exploitation over the same time period. The size indicators were based on a stratified random survey design with a western IIA otter trawl and 19mm mesh codend liner. Species caught by the survey are mostly demersal fish but some pelagics, notably herring and mackerel are caught in the survey but with lower catchability than the demersals. Indices were calculated for fish sizes between 15 and 150 cm. Details of the survey design and calculation of indices have been published elsewhere (Duplisea and Castonguay 2006). Total survey biomass was used as the measure of community biomass for calculation of the community exploitation rate.

Landings for calculating exploitation rate were taken from the NAFO fishstat database (www.nafo.ca). All reported demersal landings were included for area 4VSW for the eastern Scotian Shelf and 4X for the western Scotian Shelf.

Relative exploitation rate was calculated as the total landings/total survey biomass. Even though pelagics (primarily herring and mackerel) were used in the calculation of indices, they on average (1970–1995) comprised only 6.8% of the total survey biomass and only 5.5% if an outlier (1987) was removed.

Direct comparisons within year of the indicator value and the multispecies exploitation are likely to reflect mostly the removals of the fish within the year rather than subsequent ecological effects. Ecological effects of removals (e.g. compensation, predatory release) are more likely to occur at lagged time scales. Evaluation of community indicators implicitly contain the assumption that we are not just looking at primary effects of removals but more importantly the secondary effects. Therefore, in addition to looking at the year-on-year comparison of relative community exploitation rate and indicator value, we also considered lags of up to 10 years in a correlation analysis of exploitation rate vs. indicator value (the indicator was considered the lagged response).

2.3.2.1 Results and discussion

On the Eastern Scotian Shelf there is a trend in indicator response with relative exploitation rate (RER) but it tends to be positive (Figure 2.3-3), contrary to what we normally expect. For example, the size spectrum slope shows a positive (though non-linear) relationship with increasing RER. If causal, this suggests that increasing RER creates a shallowing of slope and therefore relatively more large fish in the system. Other studies have shown that size spectrum slope usually steepens with increasing exploitation rate both for empirical data from real systems (Blanchard *et al*., 2005) and also simulated systems (ICES, 2005b).

We tested for correlation between RER and size spectrum indicators such as slope (Table 2.3-1). The correlation was positive for lags of up to 10 years between indicator responses after an RER. We therefore cannot conclude that indicators respond at all to RER on the Eastern Scotian Shelf in the expected direction.

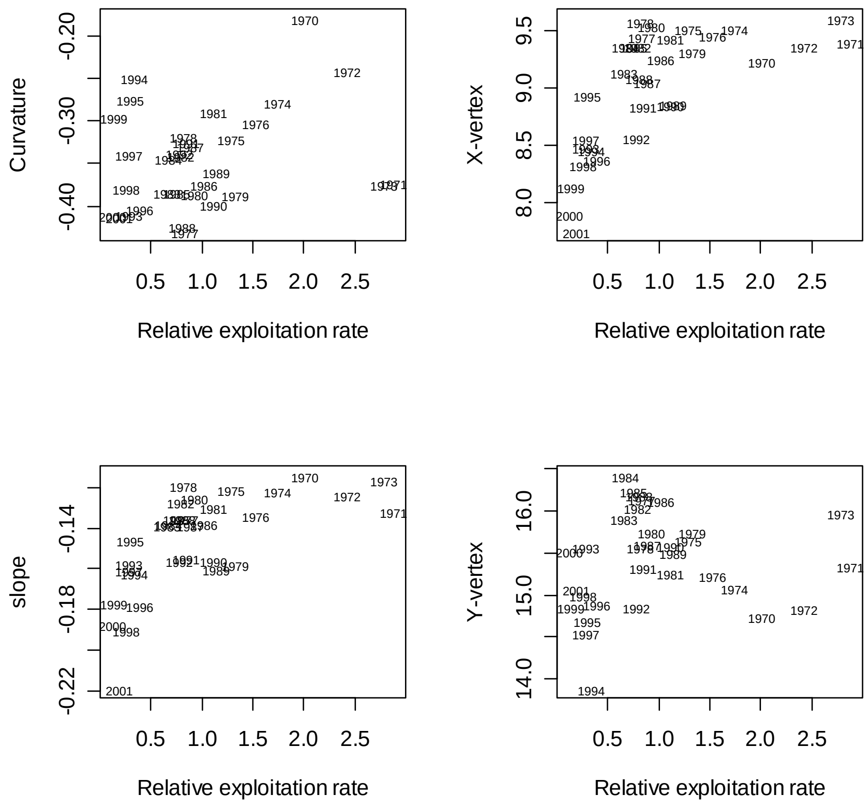

Figure 2.3-3: Values of indicators versus relative exploitation rate for the Eastern Scotian Shelf (NAFO zone 4VSW). Curvature is the curvature of a quadratic fitted to biomass size spectra, X-vertex is the body size at the fitted parabola vertex, Y-vertex is the biomass at the vertex of a fitted parabola and slope is the straight line slope of numbers vs. weight.

Table 2.3-1: Lagged correlation between relative exploitation and size spectrum indicator for the Eastern Scotian Shelf (4VSW). The lag is in the response of indicator to the exploitation rate.

Lag (years)	Curvature	Slope	X-vertex	Y-vertex
0	0.28	0.64	0.66	0.12
1	0.31	0.71	0.68	0.18
2	0.28	0.63	0.67	0.20
3	0.10	0.62	0.67	0.25
4	0.25	0.62	0.71	0.18
5	-0.04	0.56	0.70	0.29
6	-0.03	0.60	0.70	0.27
7	0.19	0.53	0.67	0.24
8	-0.01	0.55	0.66	0.41
9	0.07	0.53	0.61	0.45
10	0.00	0.53	0.62	0.62

Though this indicator response is the opposite of that expected two hypotheses might explain the pattern: (1) the exploitation rate in the system is a fishery response to increasing numbers of large fish which is explained by a shallowing of the length-frequency spectrum slope. That is, fisheries increased their effort and subsequently exploitation rate when there were valuable fish to catch. Therefore one could exchange the X and Y variables to reflect this causal relationship. (2) The Eastern Scotian Shelf is one of the best documented collapsed fish communities in the world (e.g. Frank *et al*. 2005). This collapse occurred in the late 1980s. Figure 2.3-3 shows that until about the late 1980s there was little if any relationship between RER and size spectrum slope, but after that the exploitation rate decreased (as fishing moratoria were imposed) and the indicators of fish communities showed rapid declines in the numbers of large fish. One might even consider that the present Eastern Scotian Shelf fish community was a functionally different community before and after the collapse period. As a result, the time around 1990 represents a transition break point between periods rather that a continuous change over the entire period.

Hypothesis 1 is quite plausible when we consider that fishing effort was not well regulated in the early years of the survey and therefore foreign and domestic fishing fleets had the latitude to increase or decrease effort according to the opportunities they saw for extracting wealth from the demersal fish community. Hypothesis 2 is also plausible if the collapse led to a new functional community on the Eastern Scotian Shelf. Of course there is no reason that both could not be true but one must be aware of the collapse breakpoint in interpretations of indicators.

Hypothesis 1 suggests that a more appropriate system for looking at this sort of comparison would be a more robust system that has not displayed such drastic changes in fish community structure. The North Sea is a likely candidate for such as study.

2.3.3 Comparison of two the two exploitation indices for the North Sea

Based on the international catch statistics for demersal species (Figure 2.3-4) and the estimates of the demersal-fish biomass (>10 cm), a removal rate was calculated for the years 1998/2004, the trend in which is compared with the MSVPA MSF for demersal fish only in Figure 2.3-4. This figure is not totally consistent, because the removal rate estimate includes Norway pout and the MSVPA estimate does not. Nevertheless, the conclusion for this limited range of years can only be that the two estimates from two different sources do not yield the same signal. However, it should be noted that the removal rate is based on some unverifiable assumptions about the grouping and are heavily contingent on the assumption that the biomass estimates for cod and plaice from the assessment are accurate. Moreover, in the absence of a consistent series of discard data, the removals include only landings statistics. We suggest that the MSVPA series provides a more consistent picture of the trends in exploitation rate of the different components of the fish community.

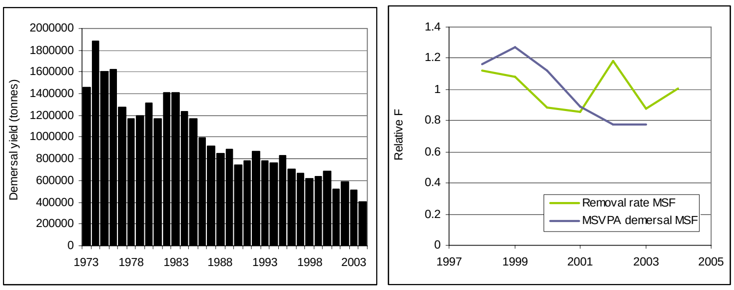

Figure 2.3-4: (a) Landings statistics for demersal fish; (b) comparison of the estimated MSVPA MSF for demersal fish (roundfish + flatfish) and the removal rate MSF for demersal fish (includes Norway pout).

2.3.4 Simulation studies

Empirical examination of how community exploitation indices, such as multispecies F, relate to different community indicators enables us to examine the changes that have occurred in the community after the exploitation has occurred. It is very difficult to disentangle the effect that exploitation has had on the community indicator (and vice-versa) especially when other factors are affecting both of the indices. Therefore assuming direct causality from such approaches can be misleading. Simulation modelling is a complimentary approach that can be used to test the effects of assumed processes on one another, and can be used to investigate the effects of changing particular parameters when others are controlled. To address issues relating to the response, specificity and sensitivity of community indicators to changes in multispecies F and other measures of exploitation, simulation modelling of the North Sea will be carried out using the tool Osmose. An overview of the objectives, model structure and initial parameterisation is given in Section 2.4.

2.4 Simulation environments

2.4.1 Introduction

While trying to evaluate of the usefulness of ecosystem indicators (see for example section on “what is a large fish?”), it is difficult, *a priori,* to score the sensitivity, responsiveness and specificity of ecosystem indicators to fishing pressure because these features are case specific and result from complex multiple and nonlinear species interactions. It is also a difficult task *a posteriori* because we are not always able to provide reliable estimates of multispecies fishing mortality, and even if we are able to, the dynamics of fishing mortality is multidimensional and its direct effects are combined with indirect trophic effects and climate variability.

In this context, model simulations can help us to better understand the response of ecosystem indicators to various levels of fishing, owing to the fact that the forcing factors are controlled and that direct “observation” of simulated data is possible. One of the candidate multispecies models for undertaking such sensitivity analyses is the Osmose model. Because this model explicitly considers size and species dimensions, it can be used to study of the properties of a large subset of the ecosystem indicators such as those proposed by the SCOR/IOC Working Group 119 (size-based and species-based indicators).

We carried out the initial steps required for parameterisation of Osmose for the North Sea ecosystem for the purpose of addressing terms of reference being undertaken as a part of WGFE and in particular to carry out the following:

- Sensitivity tests of ecosystem indicators (against variations of single species and multispecies F). In particular, we will tackle the methodological issues raised in the section “what is a large fish?”
- Specificity tests of ecosystem indicators (relative and combined effects of fishing and productivity of the system)
- Tests of different spatio-temporal management strategies and the responsiveness of indicators

The model Osmose will also be used to study the effect of fishing small prey species (i.e. sandeel) on larger predator species compared to the direct effect of fishing larger species (which relates to work already being carried out in the EU project BECAUSE).

Finally, a goal of this work will be to carry out a model comparison exercise. Models that could be compared potentially include the North Sea 1991 Ecopath model (G. Daskalov and S. Mackinson, CEFAS, unpublished), the size-based model of Pope *et al.* (in press), a coupled dynamic size spectra model (J.L. Blanchard, CEFAS, unpublished), and North Sea MSVPA (ICES, 2005) and SMS (Lewy and Vinther, 2004). Model comparison work will also be relevant for the ICES Study Group on Multispecies Assessments (meeting proposed for late 2007). We think that there are potential interests in all these existing models because they focus on different aspects of the functioning of marine ecosystems. But a common difficulty remains the validation of simulation output. In this context, cross-comparing the results produced by independent models may help to consolidate, refute or improve “what if?” scenarios of ecosystem effects of fishing. If the results show similar trends, then more confidence in them may be gained. On the contrary, if the results are divergent, the cross-comparison can still be informative, because it allows for the identification of a range of possible trajectories for the system dynamics.

2.4.2 Overview of the structure and hypotheses of the Osmose model

A detailed description of the Osmose model is provided in Shin and Cury (2001; 2004). Osmose is a multispecies model based on the central hypothesis that fish predation is an opportunistic process depending on size suitability and spatial co-occurrence between a predator and its prey. This size-structured and spatial model proposes an individual-based formulation of the key processes of the fish life cycle: predation, growth, reproduction and mortality (by predation, starvation, and fishing). The unit of interaction is the fish school, defined in the model as being a group of fish having the same size, the same spatial coordinates, requiring similar food and belonging to the same species. Using object-oriented terminology, a fish group is represented by a class belonging to the class "cohort" who in turn belongs to the class "species". This hierarchical structure allows key variables (biomass for example) to be tracked at different levels of aggregation. From each class, which is a kind of mould characterized by attributes (*e.g.* biological parameters) and functions (e.g. growth, predation), a number of objects are created that are part of the simulated system.

Only the dynamics of fish species are explicitly modelled. The other trophic compartments of the ecosystem are implicitly taken into account through various model parameters. For example, phytoplankton, zooplankton and invertebrate prey of fish are represented through the total carrying capacity of all the non-piscivorous fish modelled. Predation of fish by top predators such as mammals and birds is taken into account by an additional natural mortality term. Step by step, different processes that affect fish species dynamics are modelled. Fish schools move in a two-dimensional grid, which is represented by a set of square cells, with closed boundaries. Their dynamics and interactions are modelled through some processes that are briefly described in the order in which they are implemented within a time step.

2.4.2.1 Carrying capacity constraint

In OSMOSE, the carrying capacity corresponds to the upper limit of the viable biomass of all non-piscivorous fish in the system. At the beginning of each iteration, and in each cell of the grid, the total biomass of non-piscivorous fish is compared to the system carrying capacity. If it exceeds the carrying capacity, then the abundance of non-piscivorous fish schools is reduced to the implemented level of the carrying capacity. For the sake of simplicity, the reduction in biomass operates uniformly across the non-piscivorous schools. It means that in each cell, each group of non-piscivorous fish undergoes the same density-dependent mortality due to the lack of food.

2.4.2.2 Spatial distribution

Depending on species, age or size, fish are placed at the beginning of each time step in their mean spatial distributional areas. These areas will be provided as input to the model for each semester or quarter depending on a compromise between the availability of the information and the migration patterns of the species modelled. Then, within the time step, local movements occur as described below.

2.4.2.3 Foraging and predation

This phase only concerns the piscivorous fish in the system. The order in which fish schools act is randomly set at each time step. School displacement is directed by the search for the adjacent cell with the highest biomass of potential prey. Once it has moved (or stayed in its cell), each fish school proceeds to the feeding phase, thereby causing an explicit predation mortality for each school preyed upon. It should be noted that two criteria form the basis of the predation process: an individual can potentially feed on any species provided that (i) there is spatio-temporal co-occurrence (to be considered accessible, prey fish schools have to be located in the same cell as predators) and (ii) the predator/prey size ratio is not less than a minimal threshold (input parameter of the model). Therefore, two species can simultaneously be predator and prey of each other and predation opportunism takes into account the possibility of cannibalism. These trophic patterns are consistent with the observations of very diversified and time-varying diets of fish in marine ecosystems.

Finally, when all fish schools have achieved feeding activity, a predation efficiency coefficient is calculated for each fish school. This coefficient is determined as the ratio between the food biomass ingested by a school and the food biomass required to fulfil its vital functions (input parameter of the model).

2.4.2.4 Growth

Mean annual growth rates in length are calculated from the von Bertalanffy model (1938). This mean rate is readjusted to take into account the amount of food consumed by a fish school during a time step. A critical threshold ξ_{crit} can be determined for predation efficiency beyond which it is considered that the food ration is dedicated to fish growth. An approximation is to consider that if a school predation efficiency $\xi \geq \xi_{crit}$, then growth rate in length varies linearly with ξ such that (i) for $\xi = \xi_{crit}$, the rate is null, otherwise (ii) for $\xi = (\xi_{max} + \xi_{crit})/2$, with $\xi_{max} = 1$, growth rate equals the mean growth rate calculated by the von Bertalanffy model.

2.4.2.5 Other sources of mortality

Starvation mortality affects fish schools when the food ration is too low for fish maintenance requirements. Beverton and Holt (1957) advocate the possible existence of starvation mortality for adult stages of fish in a linear model linking natural mortality rates to fish density. By considering that for each species, nutritional resources are limited, this linear model is applied under the hypothesis that the greater the density of fish, the less the food ration per fish will be. Hence, starvation mortality rates of fish schools are linearly expressed in relation to predation efficiency when $\xi \leq \xi_{crit}$.

Fishing mortality rates are applied to the different species, following the classical survival equation structured by age:

$$N_{s,t+1} = N_{s,t} \; e^{-Fs}$$

where $N_{s,t}$ is the abundance of species s at time t, and F_s the fishing mortality rate applied to species s.

2.4.2.6 Reproduction

Let ϕ_s be the relative fecundity of species s, SB_s its spawning biomass, $B_{s,a}$ the biomass of age-class a, a_{Ms} its age at maturity, and A_s its longevity. Assuming that the sex-ratio equals 1:1 for all species, and noting $N_{s,0,t+1}$ to be the number of eggs spawned by species s at the end of the time step t, the following equation is applied:

$$N_{s,0,t+1} = \phi_s \, SB_{s,t} \quad \text{with} \quad SB_{s,t} = \frac{1}{2} \sum_{a=a_{M_s}}^{A_s} B_{s,a,t}$$

The tuning of the model is done using a genetic algorithm. The objective is to find a solution vector for age0 mortalities for the species of the model, knowing that the explicit mortality applied to this age class is actually underestimated in the model. Indeed, we apply the same starvation (limiting carrying capacity) and additional predation (due to other predators not explicitly represented in the model) mortalities than for other age classes. In addition, we do not take into account the huge loss of eggs due to exportation from the system, sinking and non-fertilization. The model is tuned towards observed species biomass for a given time period.

2.4.3 Parameterisation of osmose for the North Sea

2.4.3.1 Species to include

The species for the model will be selected on the basis of their numerical importance in the North Sea, in terms of their abundance/biomass, commercial interest and relevance as predators/prey in the community. The most recent run of MSVPA was used for guidance on what assessed species should be included in the model. The 10 species in MSVPA that are main predators include: cod, whiting, saithe, haddock, herring, sprat, Norway pout, sandeel, plaice and sole (see the Report of Study Group on MultiSpecies assessment in the North Sea (SGMSNS), ICES, 2005). Additional non-assessed species included in MSVPA as “other predators” are: grey gurnards, grey seals, North Sea mackerel, *Raja radiata*, sea birds, horse mackerel, and western mackerel. Since OSMOSE is a fish community centered simulation tool our species will only include fish explicitly, and predation by other predators is accounted for implicitly in the model.

Relative abundance estimates from the Quarter 3 English Groundfish Survey were ranked by species, and expressed in terms of the proportion of total relative abundance of all species caught. This was calculated for the reference year, 1991, and for the 1991–2003 period, to contrast whether ranked species composition taken from one year of the survey data would give the same general picture as over a longer time period. Certain species are under represented by these data (e.g. sandeel) due to low catchability of the gear. Total biomass estimates used in the North Sea 1991 Ecopath Model, that were originally derived from MSVPA estimates and catchability corrected IBTS data (based on Sparholt, 1990) were also examined. The most abundant species in the North Sea during 1991 was Dab, representing 25% of the total relative abundance of the EGFS survey data and 25% of the total biomass in the North Sea 1991 Ecopath model. Grey gurnard has recently been considered to be an important emerging predator in the North Sea (Floeter *et al*. 2005) and is also among the most abundant species. Adding dab and grey gurnard to our subset of 10 MSVPA species in the North Sea accounts for 86% of the total relative survey abundance during 1991 and 88% over 1991-2003. The list of proposed species to be included in this model is therefore comprised of: cod, whiting, saithe, haddock, herring, sprat, Norway pout, sandeel, plaice, and sole, dab, and grey gurnard.

Figure 2.4-1 shows the percentages of total relative abundance for the survey data pooled over the 1991–2003 period. The most abundant 18 species accounted for 99% of the total relative abundance in the survey data. The only species of those proposed above that is not within the most abundant 18 species is Sole, which is ranked 35 out of the total observed 116 species. Additional species that may be added to the model species are Horse Mackerel, Lesser Weever and Long Rough Dab, due to their importance in terms of survey abundance and interactions with the other model species, but this needs further consideration.

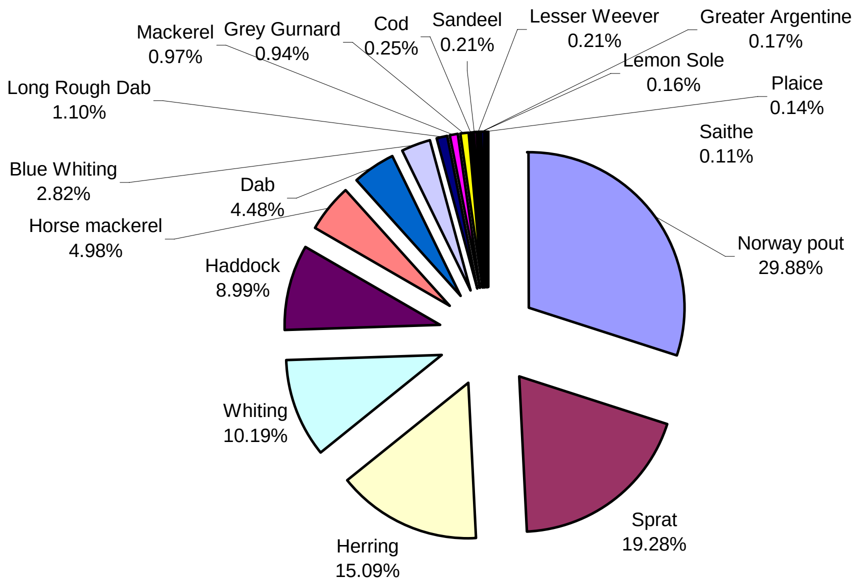

Figure 2.4-1: Proportions of total relative abundance for the most abundant species accounting for 99% of the total relative abundance. Based on Quarter 3 EGFS data pooled for the entire 1991–2003 period.

2.4.3.2 Biological parameters for each species

Several biological parameters are required for each species, in order to specify the species/size/age-based functions of the model. These include parameters required for growth, reproduction, mortality. Mean biomass of each species is required along with the range of estimates (min and max) for model calibration. Estimates for biomass were obtained from the North Sea 1991 Ecopath model, which were originally based on MSVPA estimations for assessed species and from IBTS for the others following the method proposed by Sparholt (1990). For the twelve of the selected species Table 2.4-1 shows the parameters required by the Osmose model and that have been obtained so far from the literature. As collection of all parameters was not possible during the Working Group meeting, the missing values will be added in due course.

Table 2.4-1: Life history parameters required for the Osmose model for twelve of the North Sea species. There are three categories of parameters: growth, reproduction and survival parameters. Linf, K and t0 are the parameters of the von Bertalanffy growth model, a and b are parameters from length-weight relationships, φ is the annual relative fecundity (number of eggs spawned per gramme of mature female), a_{mat} is the age at maturity, a_{max} the maximal age or longevity, a_{rec} the age at recruitment, M_{add} the additional annual natural mortality (other than that due to predation by fish and starvation).

		Growth					Reproduction		Survival		
Species		Linf	K	t0	a	b	φ	a_{mat}	a_{max}	a_{rec}	M_{add}
		(cm–1)	(year–1)	(year)			(eggs g–1)	(year)	(year)	(year)	(year-1)
Cod	*Gadus morhua*	123	0.23		0.00653	3.097	492.33	3.8	25	2	
Haddock	*Melanogrammus aeglefinus*	68	0.9		0.00558	3.133	480.49	2.5	20	2	
Herring	*Clupea harengus*	29.2	0.45		0.00603	3.090	247.20	1.84	10	2	
Norway pout	*Trisopterus esmarkii*	23	0.52		0.00518	3.117	720.30	2.3	4	1	
Plaice	*Pleuronectes platessa*	54	0.11		0.0215	2.790	342.95	2.5	50	2	
Saithe	*Pollachius virens*	177.1	0.07		0.01	2.962	1992.86	4.68	25	3	
Sandeel	*Ammodytes spp.*	21.8	0.89		0.001243	3.320	296.57	1.5		1	
Sole	*Solea vulgaris*	36.4	0.198		0.0036	3.313	590.72	2.5	40	2	
Sprat	*Sprattus sprattus*				0.002112	3.475	250.00	2	5	1	
Whiting	*Merlangius merlangius*	42	0.32		0.00518	3.117	1382.46	1.5	20	2	
Dab	*Limanda limanda*	27.75	0.27	0.21	0.0074	3.113					
Grey Gurnard	*Eutrigla gurnardus*	36.2	0.81	0.15	0.0062	3.100			9		

2.4.3.3 Spatial extent

The spatial rectangular grid has been defined to encompass the spatial extent of the IBTS survey stations (Figure 2.4-2) and is comprised of 21 latitudinal x 17 longitudinal grid cells. The grid cells in the model will be based on ICES statistical rectangles, which in the southern North Sea represent an area of approximately 30 by 30 miles.

2.4.3.4 The reference period for calibration

A reference time period is required for calibration of the model, and usually should represent a period during which the community and environment were stable. Although in previous Osmose studies decadal periods have been used (Shin *et al*. 2004, Travers *et al*. 2006), we have chosen to calibrate the model for the same reference period as that adopted for the most recent North Sea Ecopath model (Daskalov and Mackinson 2004), to facilitate comparison between the two models. Therefore the model parameters will be derived for 1991.

2.4.3.5 Average spatial distribution over the time period by species and size class

The average spatial distribution of each species (and age class) is required for the model, particularly for representing the processes of seasonal displacement of populations and ontogenetic changes in spatial distribution. In cases where there is no seasonal or life-stage changes in distribution for species, usually one average distribution can be used for the species. The first steps towards examining spatial distribution averaged over the 1991-2001 period for each species (by age class) is being estimated using the mapping tool CEFAS iSEA (http://www.ices.dk/marineworld/fishmap/ices/advanced.asp), developed for ICES FishMap. Figure 2.4-2 shows an example for one species by age - class, Cod, during the 4th Quarter. The data are from North Sea IBTS averaged over the 1991–2001 period. Further exploration of the spatial distribution variation across quarters is required in order to determine whether "seasonal" set-up will be used in the model and by age classes. Maps for 1991 will be contrasted with the decadal period, to determine whether a longer-term characterisation of the distributional limits is preferable to the spatial distribution for species in the reference year alone. Since distribution in one year will be strongly related to abundance, it may be more desirable to take and average across the longer time period. OSMOSE does not model distribution-abundance relationships, therefore the extent of the spatial distribution will not contract with changes in abundance. At the beginning of each time-step fish are placed in their mean spatial distributional areas and then within the time-step local movements can take place whereby the probability of moving depends on the biomass of potential prey in adjacent cells (Shin and Cury 2001, 2004).

2.4.3.6 Next steps

During this meeting, we just started to parameterise the model Osmose to the North Sea, and several biological and ecological parameters still need to be documented. The following time frame (Table 2.4-2) is adopted for the model to be usable before next meeting, and for addressing specific issues on the usefulness of EcoQO indicators.

Table 2.4-2: Schedule of tasks and time frame for evaluating ecosystem indicators in the simulation environment provided by the OSMOSE model.

TASK	TIME FRAME
Decide the final set of species to be modelled and complete the table of life history parameters, species biomass and F	end of March 2006
Document the spatial distribution for each species and age class and decide on temporal discretisation (quarter or semester according to species migration or distribution patterns)	mid-May 2006
Model calibration using genetic algorithms	end June 2006
Sensitivity tests of ecosystem indicators	end September 2006
Sensitivity test of the proportion of large fish	end September 2006
Specificity tests of ecosystem indicators (relative and combined effects of fishing and productivity of the system)	end October 2006
Testing different spatio-temporal management strategies and the responsiveness of indicators	end November 2006
Identify scenarios to be tested as part of model comparison exercise and coordinate activity with other relevant working groups	end December 2006

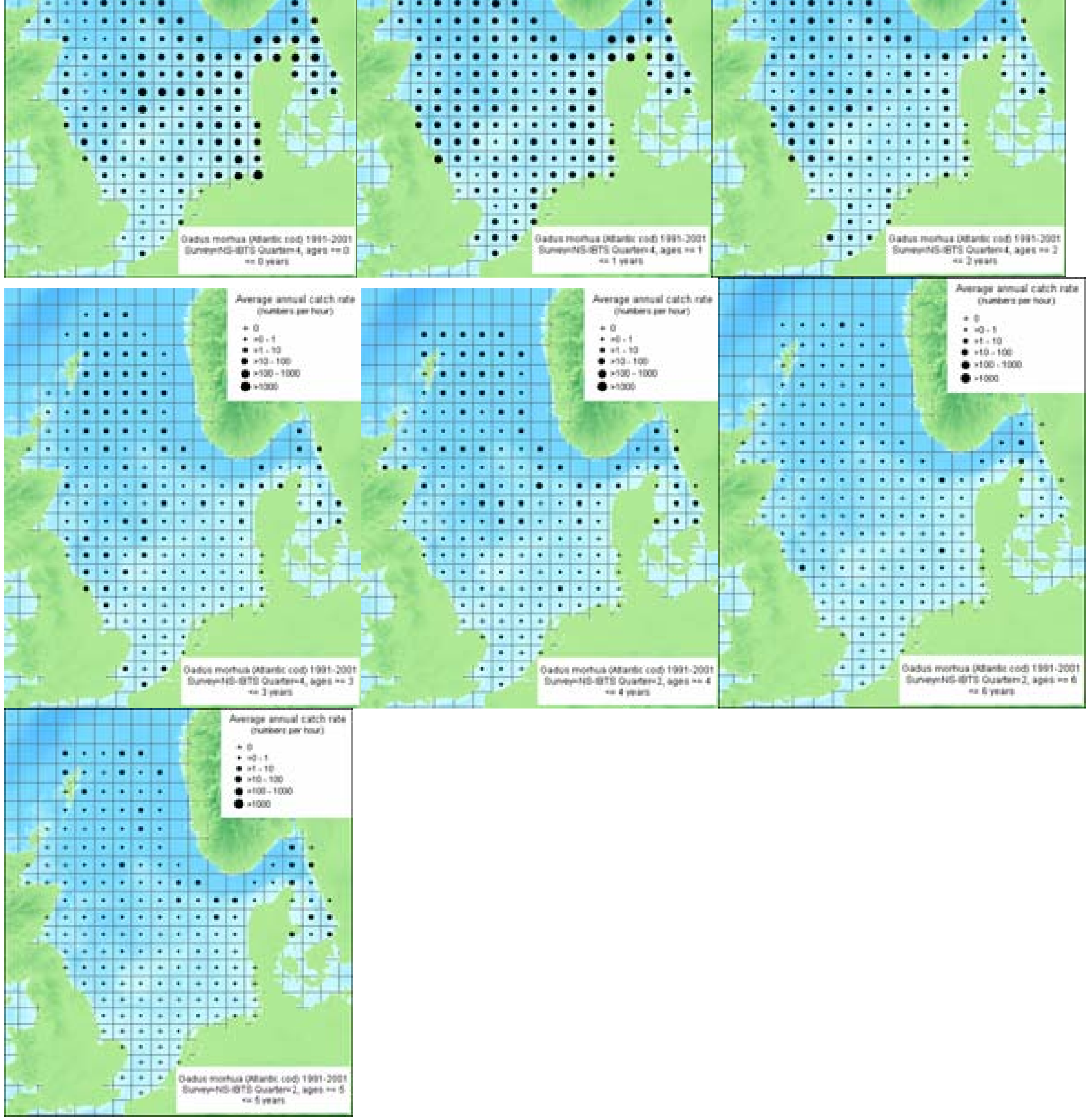

Figure 2.4-2: Example maps from ICES-FishMap showing the distribution of Atlantic Cod (*Gadus morhua*) by age class in Quarter 4 from IBTS data. The spatial extent of the model will encompass all of the stations sampled by IBTS.

2.5 Evaluating the status of rarer elasmobranch species

2.5.1 Background

During 2005, WGEF began to report directly to ACFM, so that ICES could provide advice to the EC on the status of elasmobranch stocks. To date, WGEF has concentrated on those species that are of high commercial importance and for which there are appropriate data, such as landings data, biological information and fishery-independent survey data (e.g. spurdog *Squalus acanthias* and thornback ray *Raja clavata*). There are, however, several demersal elasmobranchs on the continental shelf of the ICES area for which such fundamental data are lacking, and even surveys do not catch in sufficient quantities to allow trends in relative abundance to be examined. Hence, WGFE was requested to examine such issues and "*Liase with WGEF on identification and quantification of rare shark species*".

2.5.2 Introduction

Though over 140 chondrichthyan species are known from European seas (ICES, 2004), for the purposes of the present report, WGFE restricted analyses to nine of the more infrequent demersal elasmobranchs occurring on the continental shelf of the ICES area (Table 2.5.1).

Table 2.5-1: List of demersal elasmobranchs occurring on the continental shelf of the ICES area that were considered.

Common name	Latin name
Angel shark	*Squatina squatina*
Electric ray	*Torpedo nobiliana*
Common skate	*Dipturus batis*
Long-nose skate	Dipturus oxyrinchus
Sandy ray	*Leucoraja circularis*
Shagreen ray	*Leucoraja fullonica*
Undulate ray	*Raja undulata*
White skate	*Rostroraja alba*
Common stingray	*Dasyatis pastinaca*

2.5.3 Potential approaches

2.5.3.1 Life-history approaches

When survey data are absent, it may be possible to assess the intrinsic vulnerability of a species based on its life-history and appropriate comparison to better-known species.

Life history parameters can be estimated using life history invariants (Beverton dimensionless ratios) starting from a measure of maximum body size. The minimum data required is an estimate of maximum length, derived from surveys or regional faunal guides. The preferred starting point is to estimate von Bertalanffy growth parameters K and L_∞. From these all other key life history parameters (fecundity, natural mortality, maximum age, age and length at maturity) can be estimated. A recipe of how this can be done is shown in Dulvy *et al*. (2004), which summarises the original literature (Charnov 1993; Jensen 1997; Froese and Binholan 2000; Frisk *et al*., 2001). Be aware that fecundity may not be related to intrinsic rate of population increase (Sadovy 2001; Reynolds *et al*., 2005), and may not be informative unless phylogenetically appropriate comparisons can be made e.g. skate vs. skate might be informative but skate (*Rajidae*) vs. smoothhound (*Triakidae*) comparisons may not be valid.

The simplest starting point is to compare the maximum body size of the species of interest with the life history, demography and range sizes of other similar species, e.g. Dulvy and Reynolds (2002). The basic premise is that species with relatively large body size will also have slow life history (low K/M, high Lmax/L_∞) and low intrinsic rate of population increase (Frisk *et al*. 2001, 2005). Species with small geographic ranges may also be particularly vulnerable. If a range of species is to be considered then they can simply be ranked in terms of life history parameters (Stobutzki *et al*., 2001).

If some simple fishery information are available and the assessor is prepared to make a few leaps of faith then some basic density-independent demographic analyses can be undertaken, including Fjeopardy, Rebound potential and stage or age-based matrix models. If length or age of first capture is known then Fjeopardy ($F\varphi$) can be calculated and explored (WGFE 2005, Section 5.4, pp 87–93). Fjeopardy is the fishing mortality required to drive the population down to a chosen proportion of the virgin conditions (Pope *et al*. 2000). Population is defined in terms of spawning stock biomass produced per recruit, SSB/R. The previous proposed jeopardy levels considered range form 5 – 50% of virgin SSB/R.

One approach to estimating vulnerability to decline is to calculate the intrinsic rate of population increase, r at a standardized population size. Smith *et al*. (1999) achieved this by calculating a rebound potential r_{2m}, the growth rate at twice the natural mortality level. This equates to the point of maximum sustainable yield (MSY), which is assumed to be at half the virgin population size (Au and Smith 1997; Smith *et al*. 1999). The five life-history parameters required are age at maturity, maximum reproductive age (~maximum age), adult instantaneous natural mortality, average number of female offspring per female, and survival to age at maturity (pre-adult survival). The approach uses a modified version of the Euler-Lotka equation and is detailed in Smith *et al*. (1999) and Dulvy *et al*. (2004).

Fecundity and survival estimates by age or stage (juvenile, sub-adult and adult) can be used to build simple age or stage based demographic models (Crouse *et al*. 1993; Benton and Grant 1999; Heppell *et al*., 1999, 2000; Cortés 2002; Frisk *et al*., 2005). This approach can be used to calculate the population growth rate, generation time and also determine the most vulnerable age or stage in the life cycle, thereby allowing targeted management. These approaches are simple to implement using a freely available MS Excel add-in called “Poptools” (http://www.cse.csiro.au/poptools/matrices.htm).

2.5.3.2 Survey data

Some of the more uncommon elasmobranchs on the continental shelf are still found in some surveys, though the power of the surveys to determine trends in relative abundance may be low. Simple survey-based indices (such as frequency of occurrence) may be appropriate for some of the more frequently occurring species, though for those species that only occur very infrequently other methods, such as the probability of extinction may be more appropriate.

2.5.3.3 IUCN approach

The IUCN-Shark Specialist Group (SSG) held a meeting at Peterborough (UK) in February 2006, and IUCN-style assessments, where quantified, estimated or inferred declines in the abundance or extent of a population can lead to a broad categorisation of threat status. The IUCN approach was summarised in ICES (2004). Much of debate over the validity of these methods has been resolved by recent analyses showing that the decline rate thresholds are consistent with ICES stock assessments (Dulvy *et al*., 2005).

2.5.3.4 Probability of extinction

For some rare species, it may be possible to estimate the probability that local scale extinction has occurred using a time series of incidental observations, such as occurrence in surveys (e.g.

Solow 1993a, b; Burgman *et al*., 1995; Reed 1996; Grogan and Boreman 1998). Some studies have suggested that the probability of local 'extinction' (p) follows a stationary Poisson process:

$$p = 1 - \left(\frac{t_c}{t}\right)^n$$

where: n = the number of time intervals in which the species was observed

t = the total number of intervals sampled

Tc = the number of intervals up to the time the species was last observed

Alternatively, the number of individuals observed can be taken into account, with the probability that a species is locally extinct (p) becoming:

$$p = 1 - \left(\frac{t_c}{t}\right)^k$$

where: k = the total number of individuals observed

tc = the number of time intervals over which the species was observed

t = the total number of intervals sampled.

Probability of observing a species at low densities

The Poisson distribution assumption made above will not be appropriate in the case of individuals not being randomly distributed in space, for example if the population contracts when overall abundance decreases or if the species requires particular habitat types which are not taken into account in the sampling protocol. For these cases the negative binomial distribution provides a flexible way of describing the spatial distribution as it has two parameters in contrast to one parameter for the Poisson distribution. The mean and variance of the negative binomial distribution are:

$E[N]=\mu$ $V[N]= \mu + \mu^2 / k$,where k is called the overdispersion parameter. For small k, the species is strongly clustered in space while for large k it is randomly distributed, i.e. the negative binomial becomes a Poisson distribution. Both μ and k can be estimated from observed count of individuals per haul. Note the raw numbers per haul need to be used, and not catch rates raised to, for example, numbers per hour.

For given values of mean numbers, μ, and overdispersion factor, k, the probability to observe a zero haul (for the same gear and haul duration as used to collect the initial data) can be calculated as

$$p(0|\mu,k) = \frac{1}{1+\left(\frac{k}{\mu}\right)^{\mu}},$$

and thus the probability to observe at least one individual is 1-p(0| μ, k). Now the probability of observing at least one individual during a whole annual survey, i.e. *x* number of hauls, can be estimated using a binomial distribution. The underlying assumption is that the hauls are independent of each other, i.e. each haul is a Bernoulli random draw with the same probability of observing an individual. Thus the probability of observing at least one individual during the whole survey is $p_1=1- p(0|\ \mu, k)^x$.

We have estimated the probability of observing a non-zero haul and the probability of observing at least one individual during a whole survey for the different elsmobranch species in particular regions using consistent survey data series. The estimations were carried out for three assumptions for underlying mean number of individuals, minimum, maximum and average observed values, as well as a range of overdispersion parameters k (between minimum and maximum estimated for time series). The results allow an exploration of how likely a given number of surveys is to pick up an individual of a given species, given it is present at a certain density and has a certain type of spatial aggregation encapsulated in k.

2.5.4 Analyses of infrequent demersal elasmobranchs

Data on the catches of nine relatively uncommon demersal elasmobranchs were examined from those survey data available during the meet. The surveys analysed are summarised in Table 2.5-2.

Table 2.5-2: Summary details of surveys examined (See IBTS (year) for further information on the designs of the various GOV trawls.

Source	Region	Gear	Years	Hauls
IMR	Barents Sea trawl survey	Campelen trawl	1981-2002	6198
RIVO	Dutch Beam Trawl Survey	8m beam trawl	1985-2003	2075
RIVO	Dutch Demersal Fish survey	6m beam trawl	1970-2003	12991
IBTS	North Sea and Skagerrak	GOV	1968-2005	22762
IFR/IBTS	Bay of Biscay, Celtic Sea	GOV	1987-2004	1762
IFR	Eastern Channel Groundfish Survey	GOV	1988-2004	1447
FRS/IBTS	North-west Scotland	GOV	1998-2004	519
Cefas	Eastern English Channel	4mBeam trawl	1992-2005	1280
Cefas	Irish Sea, Bristol Channel, western English Channel	4mBeam trawl	1990-2005	2349
Cefas/IBTS	Celtic Sea, Irish Sea	GOV	2003-2005	225
Cefas	Celtic Sea	PHHT	1982-2003	1299
Total				52907

Data analysis using the negative binomial is based on the number of individuals encountered and not raised catch rates, and the present analyses have all used the number of individuals reported during the surveys. It is recognised that, in terms of some of the longer-term surveys (e.g. IBTS) there has been a reduction in tow time, which will likely affect the probability that a rare species is caught. It should also be stressed that in many of the surveys there may be taxonomic issues, with some species of skate often misidentified, and so careful interpretation of survey data are required. Some surveys have missing data, especially in early years (e.g. these species were observed, but no information on numbers were collected). In terms of ensuring that data interpretation is appropriate, it is important that the spatial distribution of the species is examined in relation to the spatial distribution of the survey grid (see Figures 2.5-1 to 2.5-3).

The results below should be considered as exploratory analyses to highlight the utility of the method. More rigorous examination of the survey data, to ensure that the survey grids are appropriate for the species in question is required to provide more accurate results. The numbers of individuals in these surveys is relatively low (Table 2.5-3, Figures 2.5-4 to 2.5-10).

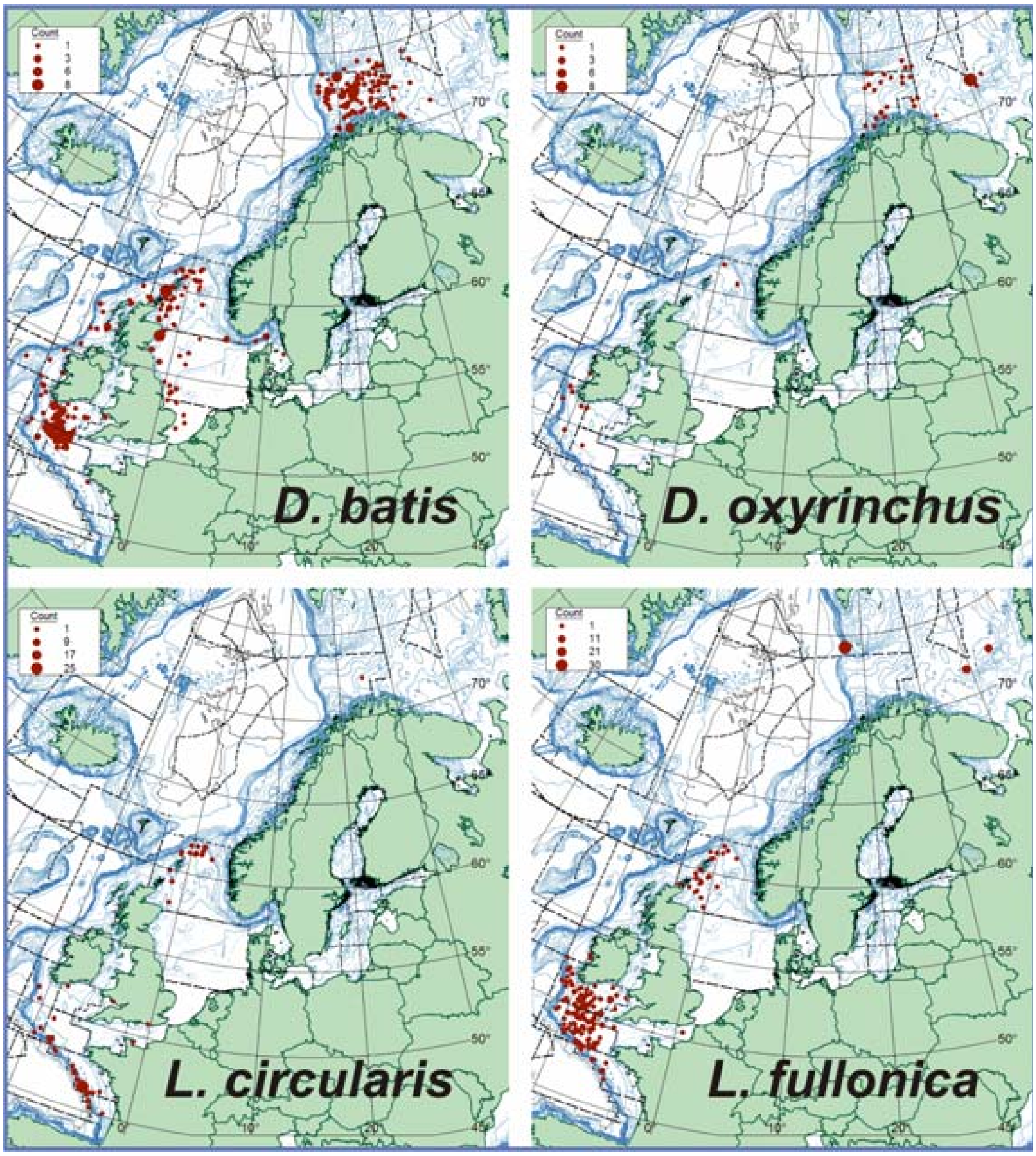

Figure 2.5-1: Occurrence of common skate, long-nose skate, sandy ray and shagreen ray in the North-east Atlantic (note: data were limited for NW Scotland and no data were available for Iceland and the Faeroes).

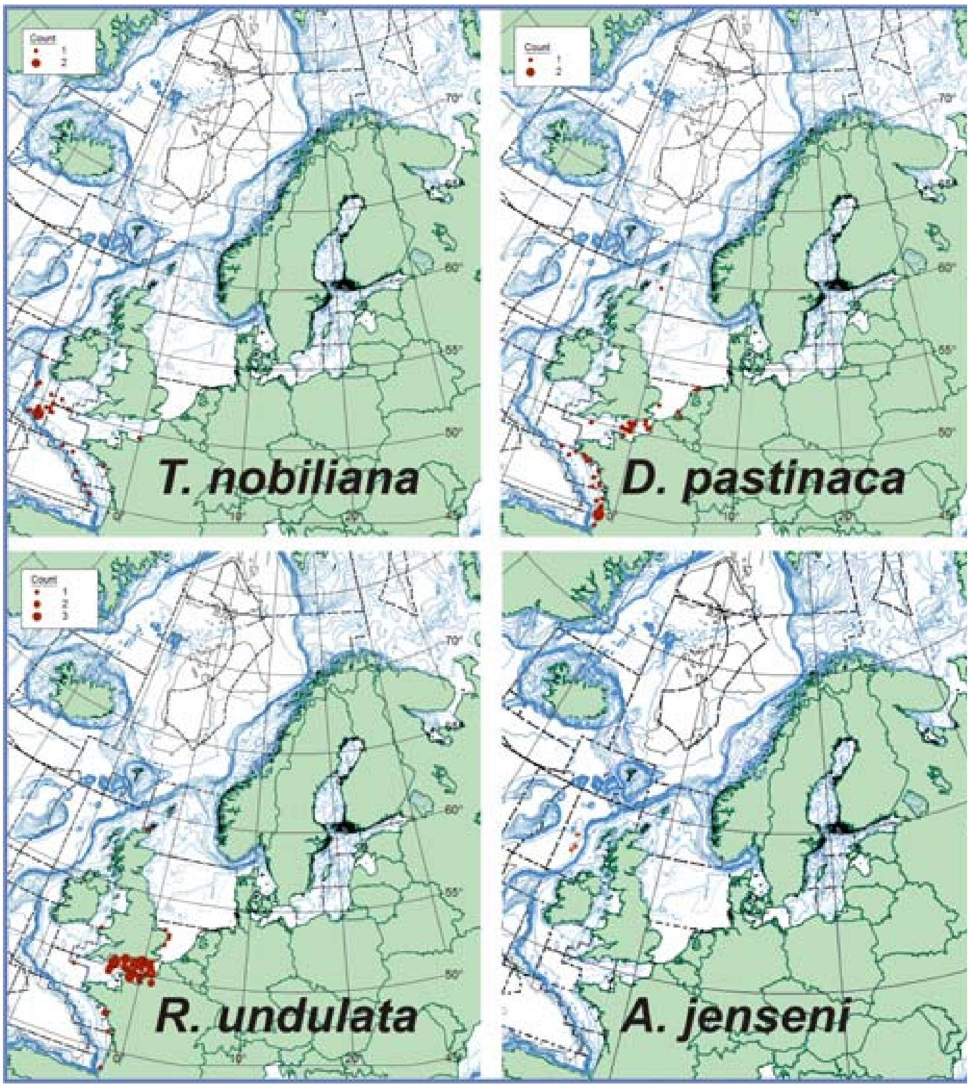

Figure 2.5-2: Occurrence of electric ray, stingray, undulate ray and Jensen's skate.

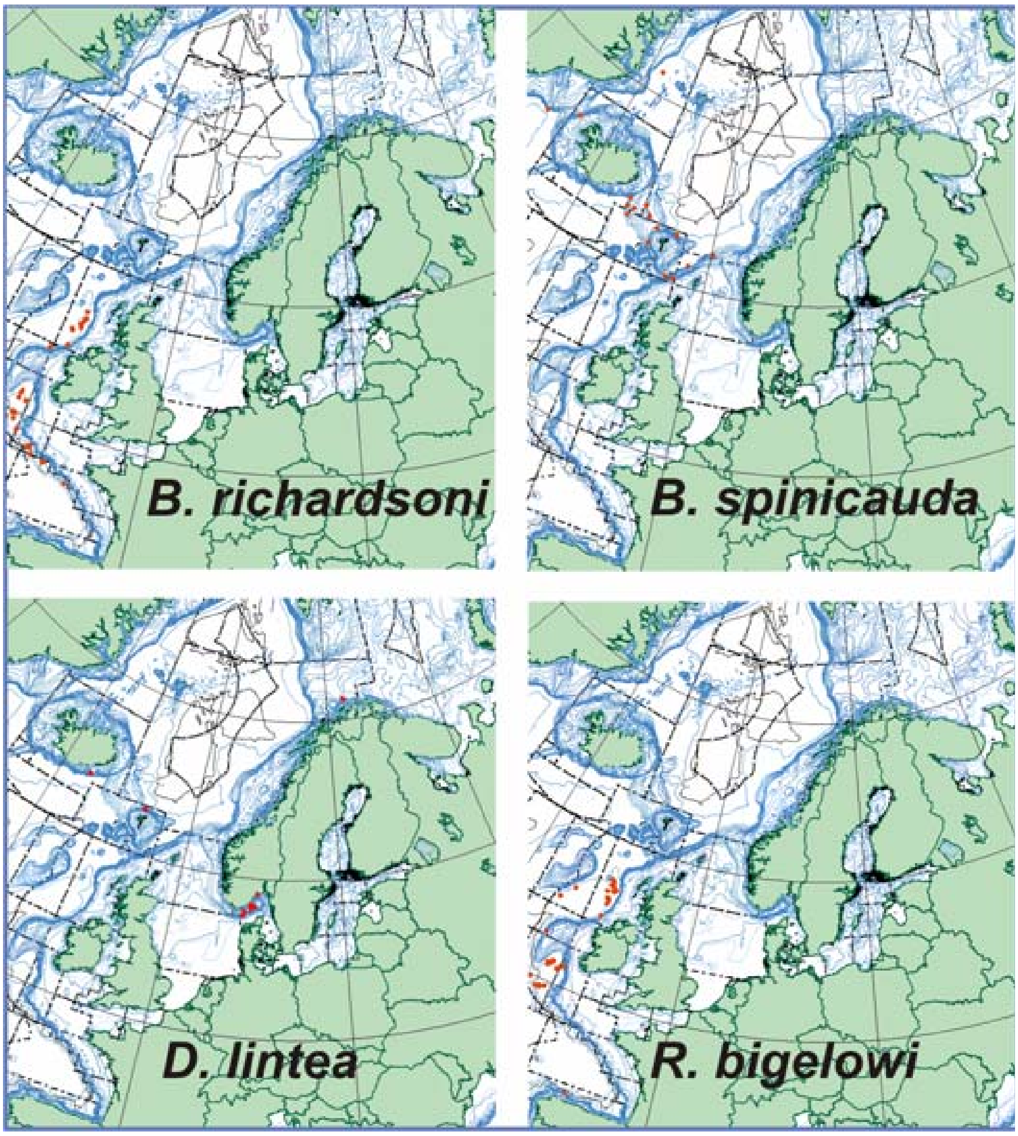

Figure 2.5-3: Occurrence of Richardson’s skate, spinytail ray, sailray and Bigelow’s ray.

2.5.4.1 White skate and angel shark

Neither angel shark *Squatina squatina* nor white skate *Rostroraja alba* were recorded in these survey data. Both these species have a northerly range limit around the British Isles, and their absence in more than 29 000 survey hauls examined that were south of 55°N hampers analysis. Both species were historically encountered in northern Europe (e.g. Couch, 1864; Day, 1884), and several studies have discussed their apparent disappearance (Quero and Cendrero, 1996; Rogers and Ellis, 2000). Analyses of survey data from more southerly areas, and examination of more historical data may be required, and surveys of areas of known occurrence would be useful in determining whether local refuges of these species remain.

2.5.4.2 Common skate and long-nose skate

Common skate were caught routinely in the Barents Sea, North Sea, Celtic Sea and NW Scotland (Figures 2.5-1, 2.5-4 and 2.5-5), with long-nose skate only caught regularly in the Barents Sea, with catches elsewhere more sporadic. Barents Sea surveys yielded <20 individuals per year (one exceptionally large catch was recorded in the 2003, though this record was considered erroneous), and on average were observed in <2% of tows. Common skate occurred less frequently in the North Sea (<1% of catches), though were more frequently encountered in the Celtic Sea (4-5% of hauls in Cefas and Ifremer surveys) and off North-western Scotland (ca. 15% of survey hauls over the period 1998-2004). Long-nose skate were observed infrequently in the Celtic Sea and North Sea, and the only survey analysed with routine catches of this species was the Barents Sea (Figures 2.5-1 and 2.5-6), where they were observed, on average, in <1% of hauls.

The occurrence of common skate in the Celtic sea quarter 1 UK survey varied at about 4%, while it was around 0.5% in the North Sea and decreased in the early 70s (Figure 2.5-11). The estimated k parameters for the negative binomial distribution were generally large in all three ecosystems, indicating that common skate seems to be rather randomly distributed within the survey areas (Figure 2.5-12). The probability of observing a non-zero haul was higher for the Celtic Sea survey compared to the North Sea (Figure 2.5-13) but given that more hauls are carried out in the North Sea (555 on average) compared to the Celtic Sea (61 on average), the estimated probabilities of observing at least one common ray in a given year are similar for the two areas (Figure 2.5-14), around 0.8 for mean observed densities.

In the Barents Sea, the occurrence of long-nose skate was very variable and increased slightly in the 1990s, but never above 3%. It spatial distribution seems to be random in most years. The probability of observing an individual in a standard haul is below 0.06, but due to large number of hauls of the survey (281 on average) the probability of observing a long-nose skate are high (0.8) unless the density is very low.

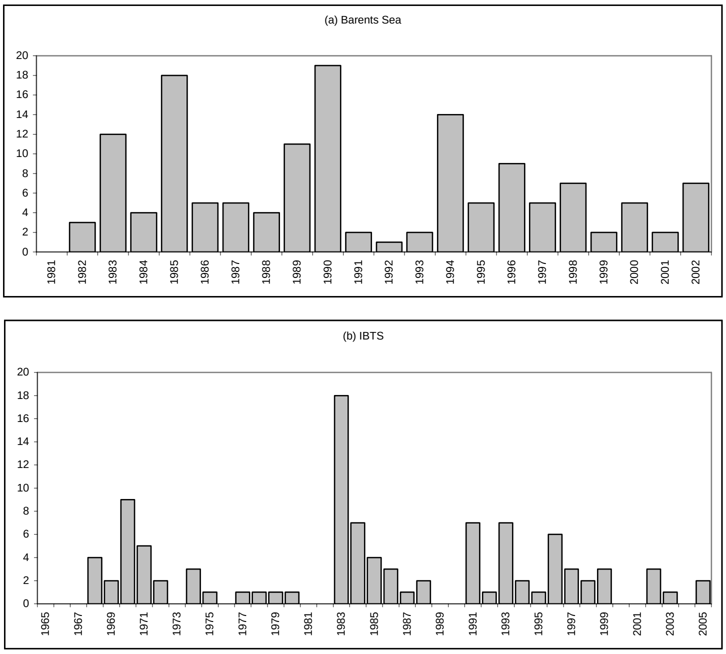

Figure 2.5-4: Numbers of common skate caught in Barents Sea and North Sea surveys.

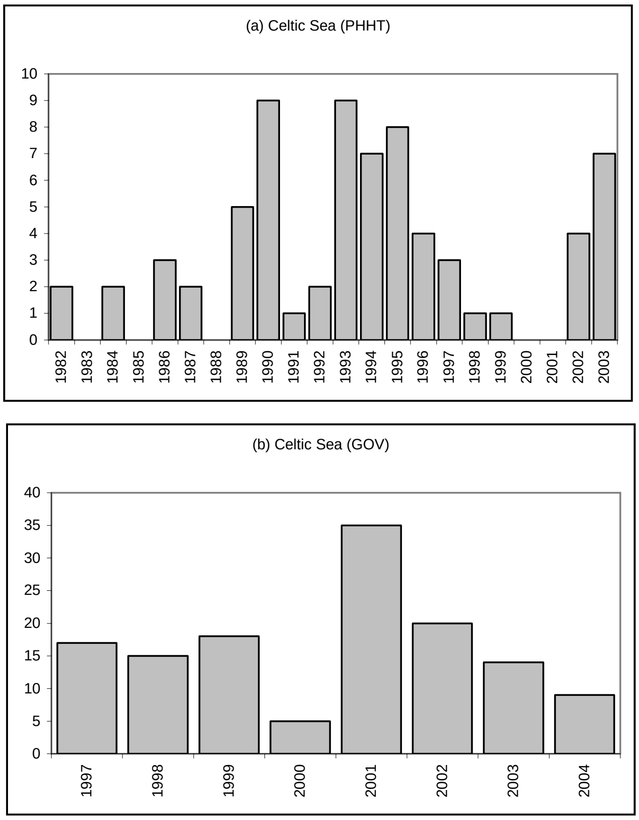

Figure 2.5-5: Numbers of common skate caught in (a) UK and (b) French surveys in the Celtic Sea.

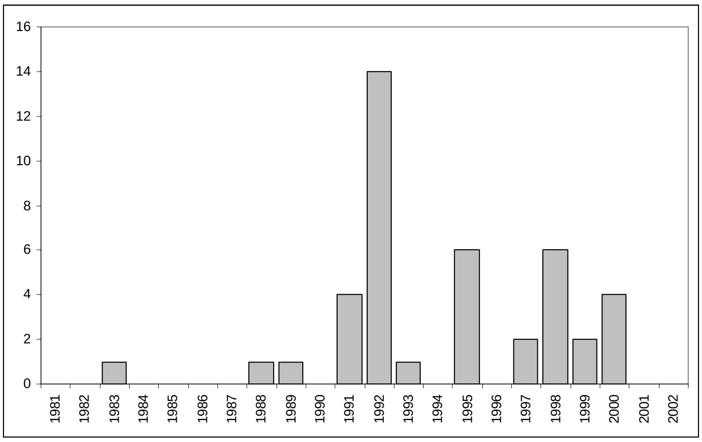

Figure 2.5-6: Numbers of long-nosed skate caught in the Barents Sea.

2.5.4.3 Shagreen ray and sandy ray

Shagreen ray and sandy ray were both caught routinely in the North Sea and Celtic Sea (Figures 2.5-4, 2.5-7 and 2.5-8), and they are also known to occur off North-west Scotland. Within the Celtic Sea, shagreen ray were, on average, observed in about 7% and 10% of hauls with GOV and PHHT respectively. Sandy ray was also most frequently recorded in the Celtic Sea, though only occurred in <3% of hauls. Both of these species was very rarely observed in the North Sea (<1% of hauls).

The occurrence of shagreen ray in the Celtic sea UK Q1 survey peaked in the early 1990s with over 20% and decreased thereafter to about 5% in recent years (Figure 2.5-11). The average occurrence in the French Q4 survey was generally higher, with about 15% in the most recent years. Spatial shagreen ray distributions were highly clustered in most years (k<1) with some exceptions (Figure 2.5-12). The probabilities of observing an individual were similar for the two surveys (0.05–0.4) at average densities. The overall detection probability was slightly higher for the French survey due to the higher observed historic densities (Figure 2.5-13).

In contrast to shagreen ray, the occurrence of sandy varied strongly between the two surveys in the Celtic Sea. While the UK survey generally does not observe any, the French survey has sandy ray occurrences between 1 and 3 percent. Sandy ray was strongly aggregated (k< 0.01), with the exception of the last two years in the French survey. The probability of observing a sandy ray was lowest for the UK surveys for average observed densities. As a consequence the probability of observing an individual during the UK was estimated as below 20% and about 90% for the French survey, again assuming average population densities.

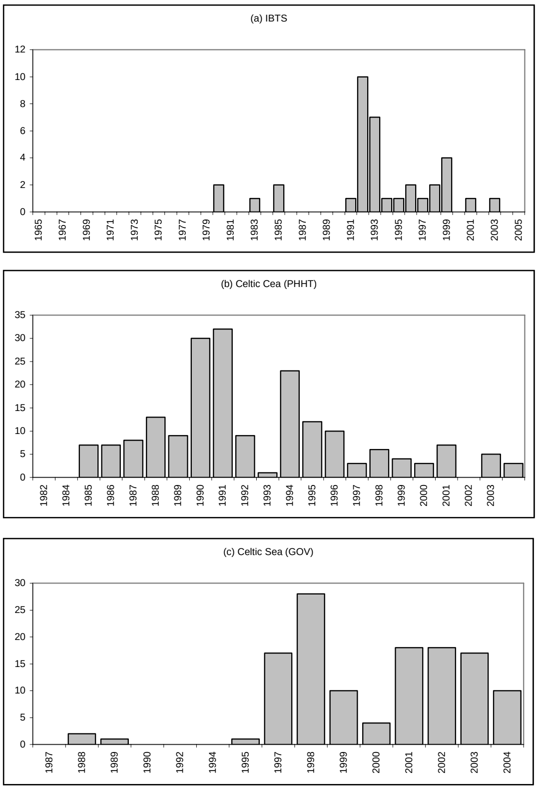

Figure 2.5-7: Numbers of shagreen ray (a) IBTS North Sea surveys and (b) UK and (c) French surveys in the Celtic Sea.

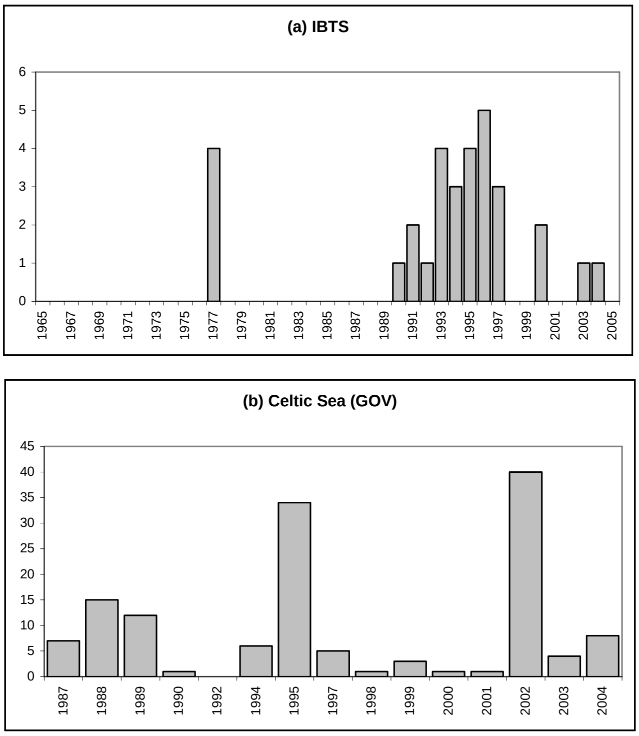

Figure 2.5-8: Numbers of sandy ray caught in (a) IBTS North Sea surveys and (b) French surveys in the Celtic Sea and Bay of Biscay.

2.5.4.4 Undulate ray

Undulate rays were only recorded regularly in the English Channel (Figures 2.5-2, 2.5-9). Though occasional vagrants have been recorded in the North Sea and Irish Sea, these may have been misidentified. They are routinely recorded in the eastern English Channel (VIId), and occur in about 4-5% of beam trawl and GOV trawls in this area, though tend to be caught in small numbers (<5 individuals).

Undulate ray occurrence was similar, around 4% in both the GOV and beam trawl surveys in the eastern English Channel (Figure 2.5-11). Catches of undulate rays were found to be randomly distributed in both surveys, (Figure 2.5-12). As a consequence, the probability of observing an undulate ray in a given haul was found to be around 0.04 in both surveys for average densities. The probability of catching at least one undulate ray at average population densities in the whole survey was at least 0.9 in both surveys (Figure 2.5-14).

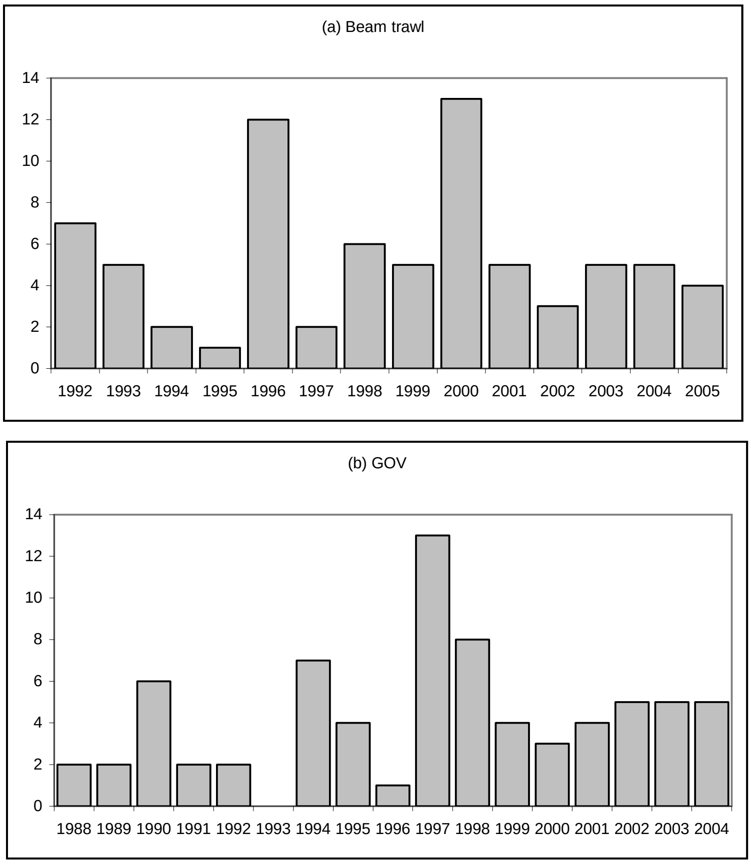

Figure 2.5-9: Numbers of undulate ray in the (a) beam trawl surveys and (b) GOV surveys of the eastern English Channel.

2.5.4.5 Electric ray and stingray

Electric ray and stingrays were encountered sporadically in various south-western surveys (Figure 2.5-2, 2.5-10), and stingrays are also known to occur in the southern North Sea. The only survey that reported them regularly was the IFR survey of the Celtic Sea and Bay of Biscay, and stingrays and electric rays occurred in 2% and <1% of survey hauls. All catch records of electric ray were of individual specimens.

The average occurrence of electric ray in the two Celtic Sea surveys (UK and French) was about one percent in both cases, but occurrence more variable for the UK survey (Figure 2.5-5 - 11). The species was randomly distributed in both surveys (Figure 2.5-12). The probability of observing an electric ray in any given haul was about one percent for the UK survey and seven percent for the French survey, assuming average observed densities (Figure 2.5-13). The overall probabilities for detecting the species in a given year were 0.5 for the UK and 0.6 for the French survey, again for average densities.

Stingrays were observed both in the English Channel small GOV survey and in the Bay of Biscay GOV survey. Average occurrences were similar in the two areas (2–4%) with no observations in the English Channel for the last two years (Figure 2.5-11). The type of spatial distribution was highly variable in both areas, ranging from random to clustered (Figure 2.5.12). The probability of observing a stingray in a haul was estimated to be slightly lower in the English Channel for average densities in each case (Figure 2.5-13). The overall probability of observing a stingray in the survey was around 0.8 for both areas (Figure 2.5-14).

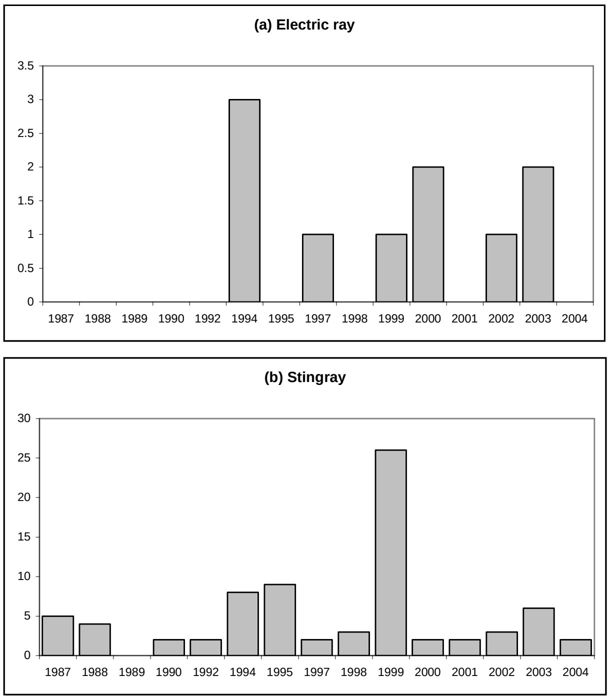

Figure 2.5-10: Numbers of (a) electric rays and (b) sting rays taken in GOV surveys of the Celtic Sea and Bay of Biscay.

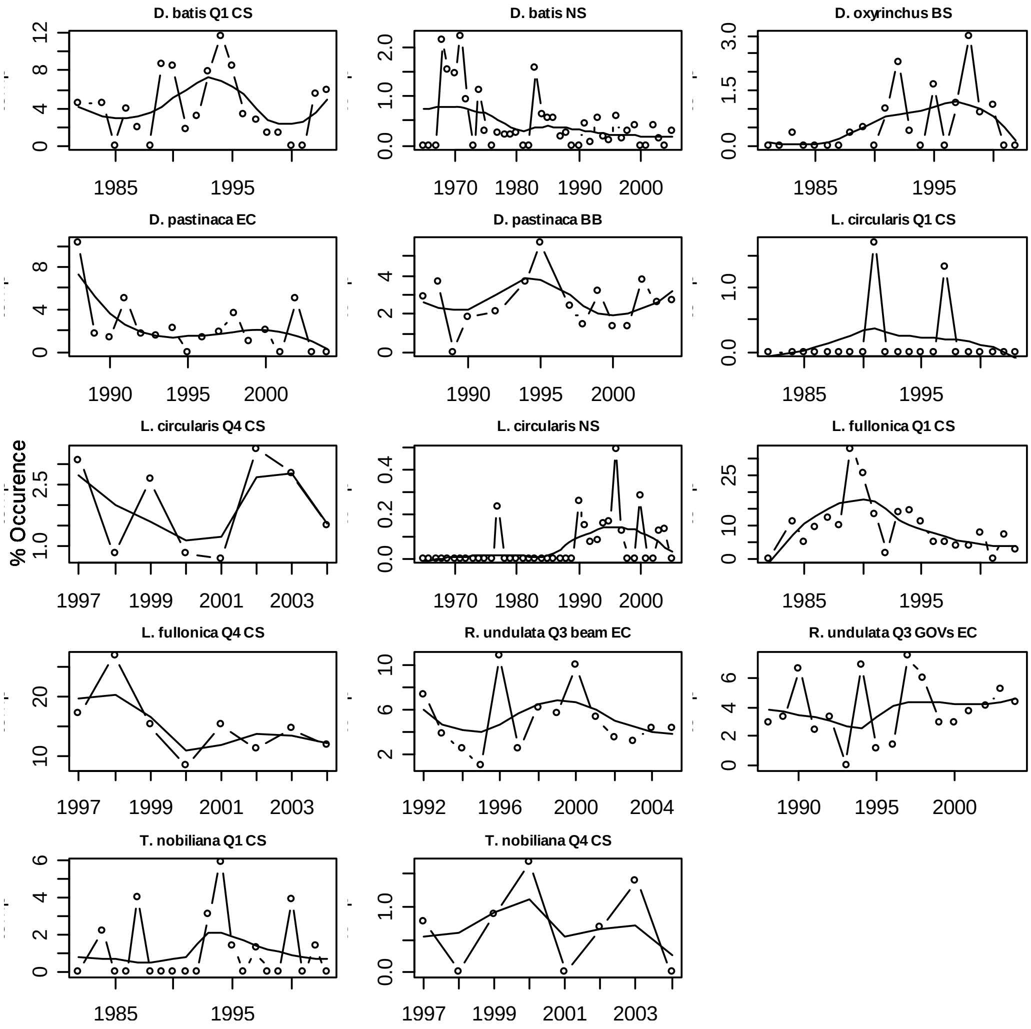

Figure 2.5-11: Occurrence (% of hauls present) of ray species in GOV and beam trawl surveys in different areas. EC English Channel, BB Bay of Biscay, CS Celtic Sea, NS North Sea. Q1 quarter 1, Q3 quarter 3, Q4 quarter 4. Lines indicate loess smoothes.

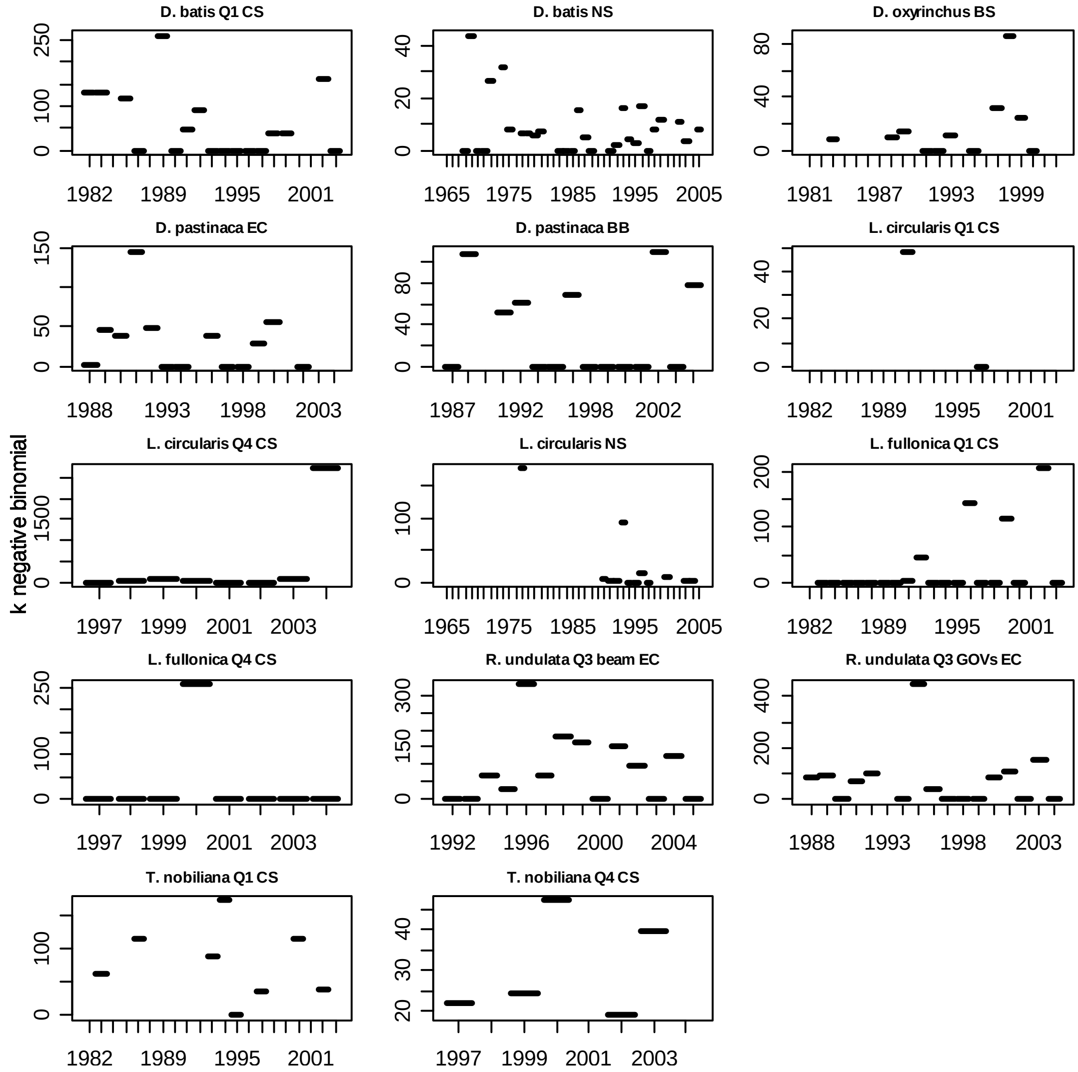

Figure 2.5-12: Estimated overdispersion parameter (k) of negative binomial distribution fitted to numbers per haul for ray species in GOV and beam trawl surveys in different areas. EC English Channel, BB Bay of Biscay, CS Celtic Sea, NS North Sea. Q1 quarter 1, Q3 quarter 3, Q4 quarter 4.

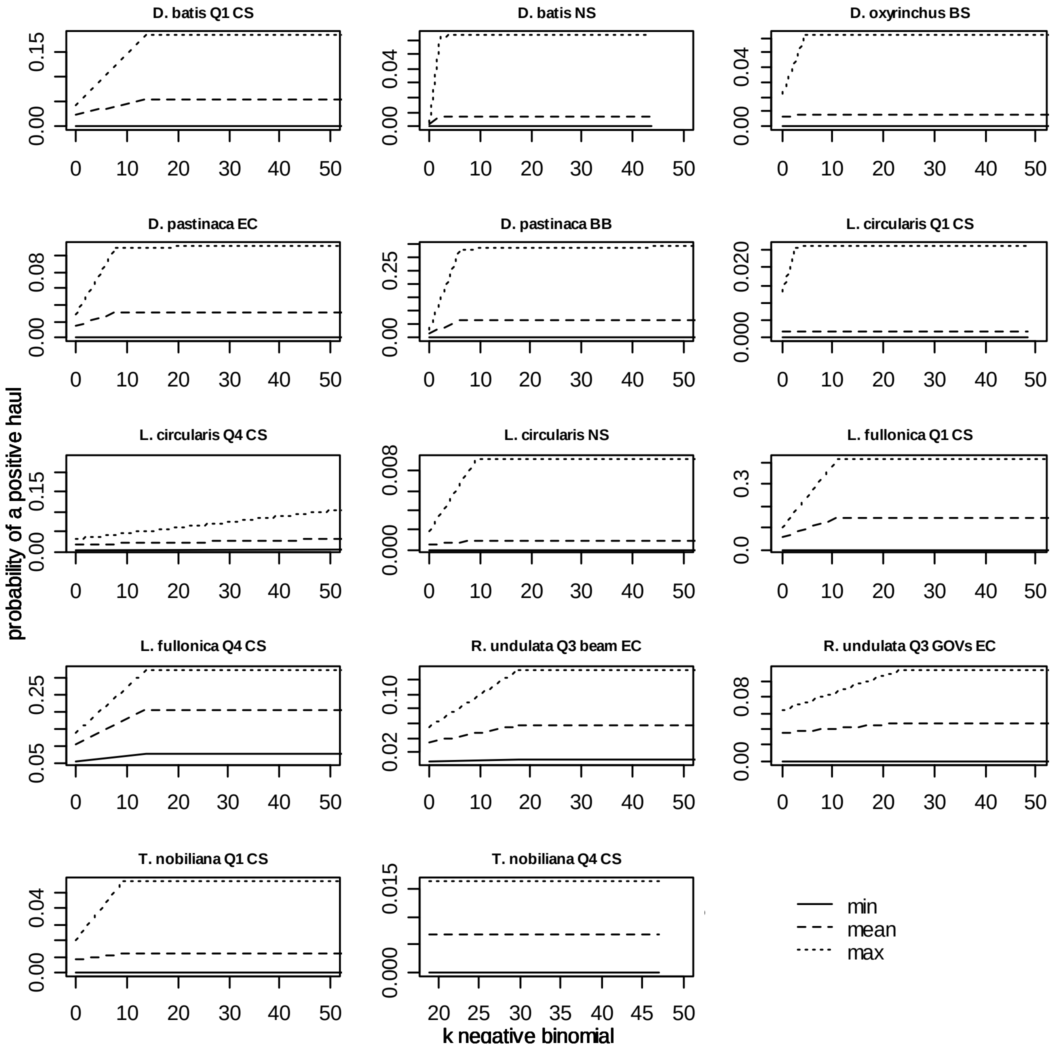

Figure 2.5-13: Probability of observing a zero haul for ray species in GOV and beam trawl surveys in different areas under different hypothesis of average numbers per haul (minimum, mean and maximum of time series) and type of spatial distribution (k parameter ranging from smallest to largest observed).

EC English Channel, BB Bay of Biscay, CS Celtic Sea, NS North Sea. Q1 quarter 1, Q3 quarter 3, Q4 quarter 4.

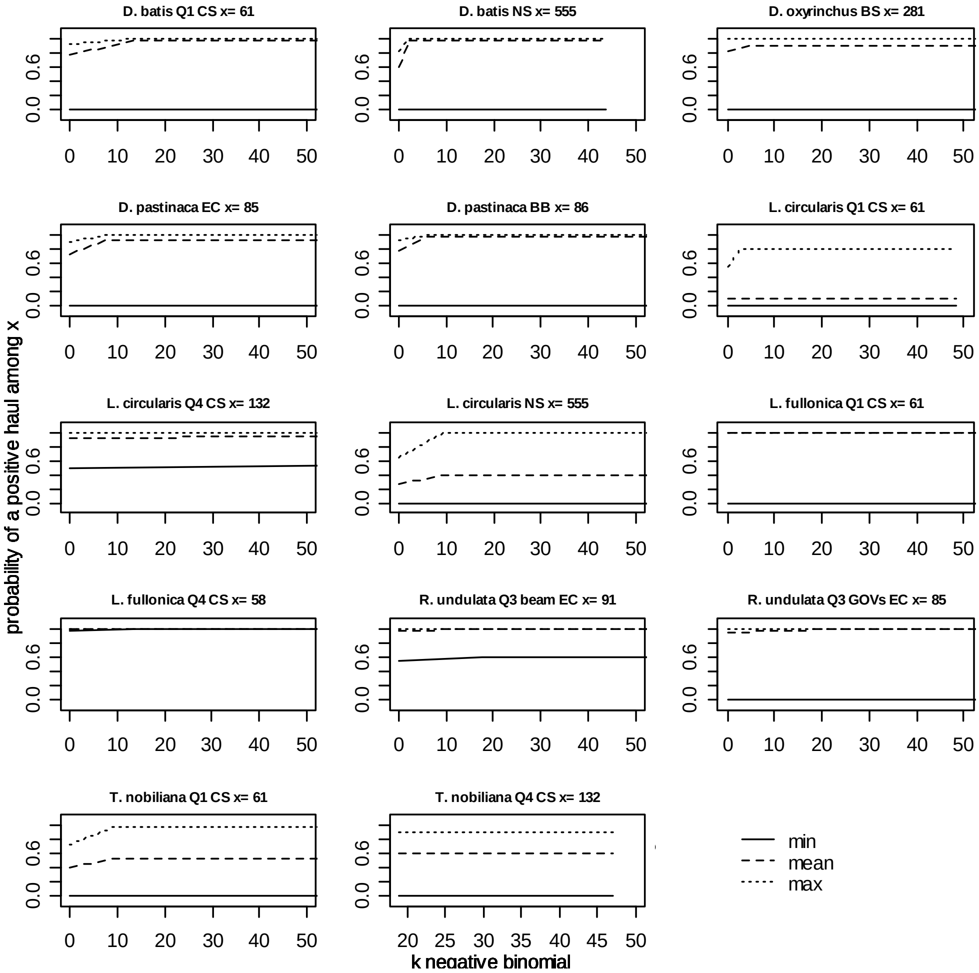

Figure 2.5-14: Probability of observing x number of zero hauls for ray species in GOV and beam trawl surveys in different areas under different hypothesis of average numbers per haul (minimum, mean and maximum of time series) and type of spatial distribution (k parameter ranging from smallest to largest observed). ***x*** **is taken as the average number of hauls of each time series. EC English Channel, BB Bay of Biscay, CS Celtic Sea, NS North Sea. Q1 quarter 1, Q3 quarter 3, Q4 quarter 4.**

2.5.5 Discussion

The percent occurrence of a species is a robust measure for following population trends, when catch rates are too low to allow temporal trends in abundance to be monitored. The proposed methods based on the negative binomial distribution allows the estimation of the probability of observing a given species at a particular population density in a survey taking into account the type of spatial distribution the species shows and the survey design (number of hauls). Based on this, the number of years can be calculated that a species has to be absent from the survey before one can be sure that the true density is really lower than, for example, the average historic density. Of course this method can also be used to calculate the probability of observing the individual if the number of hauls were increased or decreased and whether the underlying density or spatial distribution changed.

Whereas this method has applications to some of the more infrequent skates and demersal sharks on the continental shelf of the ICES area, there are extensive survey data for these areas. However, this method may not be applicable to assessing some of the infrequent deep-water and pelagic elasmobranchs, as spatially and temporally fishery-independent survey data are not available for these species.

Table 2.5-3: Number of positive hauls and total individuals caught.

	Source	Barents Sea	BTS	Channel beam trawl	Irish Sea, Celtic Sea	Celtic Sea	Irish Sea beam trawl	DFS	FRS	IBTS	IFR	IFR	Total
	ICES Div	I	IV	VIId, IVc	VIIa,e-h	VIIe-h	VIIa,e,f	IV	VIa	?	VIII, VIIf-j	VIId	
	Sets	6198	2075	1280	225	1299	2349	12991	519	22762	1762	1447	52907
Squatina squatina	+ve hauls	-	-	-	-	-	-	-	-	-	-	-	-
	∑ ind.	-	-	-	-	-	-	-	-	-	-	-	-
Torpedo nobiliana	+ve hauls	-	-	1	1	13	-	-	-	-	10	-	25
	∑ ind.	-	-	1	1	14	-	-	-	-	10	-	26
Dipturus batis	+ve hauls	106	-	-	4	50	-	-	75	74	59	-	368
	∑ ind.	293	-	-	6	70	-	-	99	103	133	-	704
Dipturis oxyrinchus	+ve hauls	30	-	-	-	6	-	-	-	2	2	-	40
	+ve Hauls	42	-	-	-	7	-	-	-	2	2	-	53
Leucoraja circularis	+ve hauls	1	-	-	-	2	1	-	-	20	45	2	71
	∑ ind.	1	-	-	-	3	2	-		31	138	2	177
Leucoraja fullonica	+ve hauls	4	-	-	-	117	1	-	2	29	78	1	232
	∑ ind.	76	-	-	-	192	1	-	2	36	126	1	434
Raja undulata	+ve hauls	-	-	65	-	1	22	-	-	2	6	57	153
	∑ ind.	-	-	75	-	1	30	-	-	2	7	73	188
Rostroraja alba	+ve hauls	-	-	-	-	-	-	-	-	-	-	-	-
	∑ ind.	-	-	-	-	-	-	-	-	-	-	-	-
Dasyatis pastinaca	+ve hauls	-	-	-	1	1	-	4	-	2	33	31	72
	∑ ind.	-	-	-	1	1	-	7	-	2	76	43	130

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3 Abundance-occupancy relationships

3.1 Background

During 2004, WGFE undertook preliminary case studies examining inter- and intraspecific abundance-occupancy (or abundance range-size) relationships, with North Sea cod used in the latter case study, and also reviewed some of the underlying macro-ecological theory (e.g. ICES, 2004a). WGFE examined further case studies the following year, illustrating the abundance-range size relationships for starry ray *Amblyraja radiata* in the North-western Atlantic, and inter-specific relationships in selected Barents Sea and North Sea fishes, with further analyses undertaken for Norway pout *Trisopterus esmarki* in the North Sea (ICES, 2005a).

The ToR for this year was to "undertake further studies on the abundance-occupancy relationships in marine fishes, with special reference to fisheries and ecosystem management issues, and the underlying mechanisms that affect such relationships".

3.2 Introduction

The effects of exploitation on macro-ecological patterns have received only limited attention (Fisher and Frank, 2004). Abundance-distribution relationships have been found over a broad range of species (Hanski *et al*., 1993; Gaston *et al.*, 1997, 1998; Holt *et al*., 1997; Gaston 1999; Gaston and Blackburn, 2000; Watkinson *et al*., 2003), and such relationships may be useful for highlighting species of concern.

In terms of fisheries, a consequence of positive intraspecific relationships in abundance-distribution is that catch rates will be proportionately higher for a given level of effort (Paloheimo and Dickie, 1964; Fisher and Frank, 2004). For example, the collapse of the northern cod stock located on the northeast Newfoundland and Labrador Shelf co-occurred with a hyper aggregation of the cod at low stock abundance. The affect was a rapid decrease in area occupied but with an increased CPUE at the centre of mass, in spite of a strong reduction in population abundance. These spatial changes co-occurred with the collapse of this stock (Rose and Kulka, 1999). Knowing how a stock responds spatially to changes in abundance can therefore be important to prevent stock collapse.

An improved knowledge of abundance-occupancy relationships are also important for the effective implementation of any marine spatial planning and other spatial management actions, such as marine protected areas (Jennings, 2000; Fisher and Frank, 2004). Additionally, such analyses may also have the capacity to highlight shifts in distribution that may be linked to environmental conditions (e.g. climate change).

As survey data are used in examining abundance-range size relationships, when undertaking such analyses it is essential to consider various issues: including density-dependent catchability, age-dependent catchability, contrasting spatial extent of the survey area in comparison to the stock area or species range, and changes in environmental conditions (e.g. Fretwell and Lucas, 1970; Smith *et al*., 1991; Marshall and Frank, 1994; Albert *et al*., 2001).

In recent years there have been an increased number of studies examining the abundance-range size relationship in a variety of taxa (e.g. Foggo *et al*., 2003; Freckleton *et al*., in press), including various fishes (e.g. Macpherson, 1989; Marshall and Frank, 1995; Swain and Morin, 1996; Brodie *et al*., 1998; Giske *et al*., 1998; Albert *et al*., 2001; Blanchard *et al*., 2005).

If spatial analyses are to be used in the assessment and management of marine fisheries, or for species of nature conservation interest, it is important to determine whether declines in extent/occurrence are due to either a contraction in range, hyper-aggregation to core habitat or a general reduction in abundance leading to a decreased catchability in surveys.

3.3 Summary of methods for examining abundance-range size

There are several methods with which to examine abundance-occupancy (or range size) relationships. The terms occupancy, distribution and range size reflect the geographic spread of the species/stock in question, often based on observational data. The use of the different terms reflects the type of data used, so that range size or distribution area are used when latitudinal or actual spatial units (km^2) are used, whereas the terms 'occupancy' and 'incidence' generally represent the proportion of an area, sometimes reported as a subset of grid units, where species are recorded as present or the frequency of occurrence in sampling programmes.

Some of the main studies incorporating abundance and spatial distribution in analyses of marine fish are summarised in Table 3.3-1.

Table 3.3-1: Overview of recent studies involving abundance-range size relationships.

AUTHORS	TOPIC	METHOD	AREA	RESULTS
Blanchard *et al.* (2005)	- density-spatial coverage relationship	survey data; predictions using ideal free distribution with habitat suitability based on temperature and weight; index of spatial coverage= proportion of rectangles containing >95% of relative abundance; temperature surfaces by splines	North Sea, cod ages 1 and 2;	cod spatial distribution seems to agree with IFD as in years of higher density they occupy a wider area; relationship between abundance and spatial coverage is curvilinear (flattens off)
Fisher and Frank (2004)	- abundance-distribution relationship	distribution indices 1) min # hauls with 90% abundance 2) prop. occurrence (prop. of hauls with species)	groundfish survey Scotian shelf and Bay of Fundy	significant relationship between ln-abundance and distribution index 1) for 16 out of 34 species
Garrison and Link (2000)	- fishing effects on spatial distribution and guild structure	matrix of index of spatial overlap Sij= # hauls with both predator i and predator j/hauls with pred i	Georges Bank groundfish survey 23 predator species by length category (according to ontogenetic shift in diet)	impact of exploitation on piscivores reducing spatial overlap (haddock and yellowtail flounder) as spatial distribution retracts (map)

Authors	Topic	Method	Area	Results
Garrison *et al.* (2000)	- predator-prey spatial overlap	Williamson spatial-overlap index based on random spatial distribution $$O_{ij}=\frac{m\sum_{z}^{m}N_{iz}N_{jz}}{\sum_{z}^{m}N_{iz}\sum_{z}^{m}N_{jz}}$$ permutation test	Georges Bank cod and haddock larvae with predators herring and mackerel	overlap depends on environmental conditions (salinity and temperature), e.g. for cod larvae and herring; always overlap haddock larvae and herring
Gaston (1996a,b)	- review of empirical studies of abundance-distribution relationship	contribution of different processes to abundance -distribution varies across spatial scales; mainly terrestrial animals and plants, mainly birds; Measures of distribution: extent of coverage; area of occupancy; number of sites (or samples)	locally abundant species are geographically more widespread	need to study underlying mechanisms: - resource usage (breadth, quantity) - metapopulation dynamics - aggregation patterns - vagrancy
Kulka, Miri and Simpson (2004)	- density related abundance-distribution changes	Survey data – GIS based calculation showing the changes in fish density with respect to abundance – direct measure of area occupied.	Grand Banks demersal survey	Demonstrated a change in the spatial structure of starry ray, from dispersed to highly aggregated distribution more vulnerable to exploitation.
Link *et al.* (2002)	- fishing effects on spatial distribution and guild structure	stomach content and survey data; matrix of spatial overlap=# hauls with both / #hauls with only pred or prey	Georges Bank; groundfish 1963-1997	as abundance of exploited species decreased, spatial overlap decreased
Mountain and Murawski (1992)	- variations in spatial distribution with respect to environment	weighted mean environmental index (temperature bottom or surface, depth latitude) Indw=Ind_i * Dens_i/Sum Dens_i i station inspect slope of Indw vs. mean Ind_i should be 1 if no relationship	bottom trawl survey USA coast; ~30 demersal and pelagic species	north shift or to deeper waters observed for several species
O'Driscoll *et al.* (2000)	- scale dependent spatial association	potential contact index= # neighbours potentially encountered by an individual within radius t $PC(t)= \pi t^2 E(d_t)$ $$E(d_t)=\frac{\sum_{i=1}^{n}d_i\,\bar{d}_{i,j}}{\sum_{i=1}^{n}d_i}$$ di density at location i $\bar{d}_{ij}$ average density within distance t from i; could be for other species, extra contact (more than expected under random distribution), XC(t)=PC(t) - PC(t) random, PC(t) random spatial distribution, plot XC(t) vs. t: use first peak as index of patch size	simulations and cod and capelin off Newfoundland	no change in XC between cod and capelin though cod abundance declined

AUTHORS	TOPIC	METHOD	AREA	RESULTS
Rose and Kulka (1999)	- spatial changes at the centre of mass in relation to abundance	Trawl and acoustic survey data used to define a density-abundance relationship	Northern cod on the northeast Newfoundland and Labrador Shelf	Hyper-aggregation of cod just prior to the collapse of the stock
Sundermeyer *et al.* (2005)	- relationship between fish distribution and environmental variables	CPUE data; weighted environment index $I = \ln(CPUE_i)\mathrm{var}_i / \sum_i \ln(cpue_i)$ i station/rectangle var= eg temperature, depth, any numerical environmental variable; plotted I against mean(var) which should lie on the diagonal if there is no preference; could be used as indicator directly	Georges Bank cod and haddock	cod and haddock at temperatures 5 °C in winter/spring
Swain and Sinclair (1994)	- abundance-distribution relationship	stock area = min area which contains 90 or 95% of density highest abundance area= 50% density	cod in Gulf of St Lawrence	stock area increased with increasing abundance, in contrast highest abundance area did not increase
Swain and Wade (1993)	- density dependent geographic distribution	index of occupied area A= sum of areas for which density> level level= 60th percentile of density distribution across whole time series density per area estimated by kriging; asymptotic relationship tested A = a - b (exp(-b3N)) N= population size	cod by age group in Gulf of St Lawrence	area-abundance relationship well explained by asymptotic model

3.4 Spatial pattern indices and fish population abundance

3.4.1 Introduction

The following work was conducted in the framework of the EU funded project FISBOAT (Fisheries Independent Survey Based Operational Assessment Tools). Different indices of population spatial occupation have been tested and used to capture the spatial patterns of fish resources and look for links between them and abundance (see Woillez *et al.*, 2005).

3.4.2 Spatial indices

In the context of populations with diffuse limits, spatial indicators have been built in such a way as to avoid the problem of the delineation of the area of presence, which may be variable among years. This is achieved by removing zero sample values (zero hauls) so that they have a null contribution to the indices (Bez and Rivoirard, 2001).

Centre of gravity and inertia

The spatial distribution of a population can be easily summarized by tools such as the centre of gravity and inertia (Bez, 1997). The centre of gravity (*CG*) is the mean location of the population, also the mean location of an individual taken at random in the field, and the inertia is the mean squared distance between such an individual and the centre of gravity.

Let x be a point in 2D (short for usual 2D notation (x,y)), $z(x)$ the local fish density, then the total abundance of the population is:

$$Q = \int z(x)dx$$

and the probability density function of the location x_I of a random individual I is:

$$\frac{z(x)}{Q}.$$

Then, the centre of gravity is:

$$CG = E(x_I) = \int x \frac{z(x)}{Q} dx = \frac{\int x.z(x)dx}{\int z(x)dx}$$

and the inertia is:

$$Inertia = Var(x_I) = \frac{\int (x - CG)^2 .z(x)dx}{\int z(x)dx}$$

In the case of an irregular sampling design, influence surfaces for each sample (haul) are used as weighting factors. Practically, from sample value z_i at locations x_i, with surface of influence s_i, we have:

$$CG = \sum_{i=1}^{N} x_i.s_i z_i / \sum_{i=1}^{N} s_i z_i$$

$$Inertia = \sum_{i=1}^{N} (x_i - CG)^2 .s_i z_i / \sum_{i=1}^{N} s_i z_i$$

Anisotropy

In 2D, the total inertia of a population can be decomposed along its two principal axes, orthogonal to each other, explaining respectively the maximum and the minimum of the overall inertia. The square root of the inertia along a given axis gives the standard deviation of the projection of the location of the population along this axis. Anisotropy exists when there is a difference in inertia between the directions. This is summarized by the square root ratio between maximum and minimum of inertia.

$$A = \sqrt{\frac{I\max}{I\min}}$$

Number of spatial patches

The spatial distribution of a fish population in a given area may not be homogeneous. Local aggregations of fish, i.e. spatial patches which are bigger than a fish school, may be present. To identify spatial patches, an algorithm has been written, based on a distance limit to attribute samples to patches. The algorithm starts from the highest value (maximum density) and considers each sample in decreasing order. The highest value forms the first patch. Then, the next sample value is attributed to the nearest patch, provided that the distance to its centre of gravity is smaller than the chosen distance limit. Otherwise, the sample defines a new patch. The indicator obtained is, for each survey, the number of patches whose total abundance corresponds to at least 10 % of overall abundance. The distance limit has been fixed at 100 nm in this study.

Positive area

An area of influence is attributed to each sample. The positive area is the sum of the surfaces of influence of positive sample values.

$$PA = \sum_i s_i 1_{z_i \geq 0}$$

Spreading Area

This index comes from the selectivity curves which have been developed in mining geostatistics to characterize probability distributions and their dispersion (Matheron, 1981). These curves are in particular useful to handle the effect of the support on which the variable is measured or defined. They have been used in fisheries to look at the aggregation of values when the underlying abundance changes (Petitgas, 1998).

Let Q be the total abundance (in number of individuals), $Q(T)$ the highest abundance found in area of size T (expressed in nm² or percentage of a total area). A selectivity or aggregation index can be defined as:

$$2\int_0^1 \left(\frac{Q(T)}{Q} - T \right) dT$$

when expressing *T* as a proportion of the total area. However this index is dependent on the selected total area and on zero values. In order to have a statistics for which the contribution of zero sample values is null, the selectivity curves have been reversed bottom up, thus leading to the spreading area. The spreading area is then proposed instead; it is equal to:

$$2\int_0^1 \frac{R(T)}{Q} dT$$

with *R(T) = Q-Q(T)* being the abundance remaining once that in the area *T* has been removed. Thus higher values of spreading area mean that the population covers a larger area.

Equivalent area

The transitive geostatistical approach (Matheron, 1970) can be used to describe the spatial distribution of a fish population when this includes a few large density values, and when delimitating a domain with homogeneous variations is difficult (Bez *et al.*, 1995, Bez *et al.*, 1997). The (transitive) covariogram, function of the distance between two locations, is a tool for description of the spatial structure and can be used for mapping:

$$g(h) = \int z(x) z(x+h) dx$$

Here, the equivalent area is defined as the integral range of the covariogram:

$$EA = \frac{\int g(h) dh}{g(0)} = \frac{Q^2}{g(0)}$$

As we can write:

$$EA = \frac{Q}{\int z(x) \frac{z(x)}{Q} dx}$$

The equivalent area represents the area that would be covered by the population, if all individuals had the same density, equal to the mean density.

Microstructure index

This index comes from the covariogram. It measures the relative decrease between distance h=0 and a distance h0 chosen to represent the mean lag between samples; here a fixed value of 10 nm was chosen for all analyses. It measures the relative importance of the structural components with scale smaller than h0 (including random noise):

$$MI = \frac{(g(0) - g(h0))}{g(0)}$$

Some of the presented indices are spatial statistics in the sense that their values would be changed by permuting densities between two locations. This is the case of the centre of gravity, inertia, anisotropy and the microstructure index. The abundance and the other indices (positive area, spreading area and equivalent area) depend only on the statistical distribution of values over the region.

For comparison between surveys, we have decided to fix the maximum area of presence which has been surveyed along the time series (polygon restriction); the surfaces of influence are computed within this delineated domain for all years. This domain limits the surface of influence of positive sample values when the spatial population is not closed by zero values. Using such weighted factors when computing spatial indices such as centre of gravity, inertia, and global index of collocation, reduces the effects of the variations in sampled area along a time series. So the centre of gravity of trawl hauls is fixed over the period.

Measure of rank correlation

Kendall’s Tau method (Conover, 1980) has been used to test for positive correlations (one sided test) between species abundance and spatial indices.

3.4.3 Data

Data were collected during 14 groundfish surveys carried out by IFREMER from October to December between 1987 and 2003 (EVHOE series with gaps in 1991, 1993 and 1996), on the eastern continental shelf of the Bay of Biscay (ICES, 1997; Poulard *et al*., 2003; Poulard and Blanchard, 2005). The sampling design is stratified according to latitude and depth. A 36/47 GOV trawl is used with a 20 mm mesh codend liner. Haul duration is 30 minutes at a towing speed of 4 knots. Fishing is mainly restricted to daylight hours. Catch weights and catch numbers are recorded for all species. The study area is situated between 48º30'N and 43º30'N and depth ranges from 15 to 600 m

Thirteen species *Arnoglossus imperialis* (ARNOIMP), *Arnoglossus laterna* (ARNOLAT), *Capros aper* (CAPOAPE), *Conger conger* (CONGCON), *Gadiculus argenteus argenteus* (GADIARG), *Lepidorhombus whiffiagonis* (LEPIWHI), *Leucoraja naevus* (LEUCNAE), *Microchirus variegatus* (MICUVAR), *Microstomus kitt* (MICTKIT), *Mullus surmuletus* (MULLSUR), *Scomber scombrus* (SCOMSCO), *Trisopterus minutus* (TRISMIN), *Zeus faber* (ZEUSFAB) and 6 age groups (0 to 5+) of hake (*Merluccius merluccius*) have been selected for this study.

3.4.4 Results

Centre of gravity and inertia indices are illustrated for different species in Figures 3.4-1,-2. Usually there were few changes in location through years, except for mackerel and hake > 45 cm. The abundance of several selected species exhibited increasing time trends (Figure 3.4-3). The relationship between abundance and 5 spatial indices have been explored (Figures 3.4-4–3.4-8) and the results of a Kendall’s tau test for positive correlation is given in Tables 3.4-1 to 3.4-4. Positive area index was positively correlated with abundance for 11 species out of 13 and all hake age groups or categories.

Spatial indices can be used to describe changes in the spatial occupation of a species during its life cycle. For instance, the low values of the spreading area index for young hake (Figure 3.4-9, age group 0) suggest that high densities of fish were concentrated in a small part of the total area of distribution, i.e. mainly between 80 and 120 m depth (Figure 3.4-10). From age 0 to 3, the spreading area index increased and the fish progressively scattered towards shallower waters at first (age 1 in Figure 3.4-10) and then over the entire depth range Figure 3.4-10, ages 2 and 3). From age 4 (sexually mature fish) onwards, hake was again more concentrated in the deepest parts of the study area.

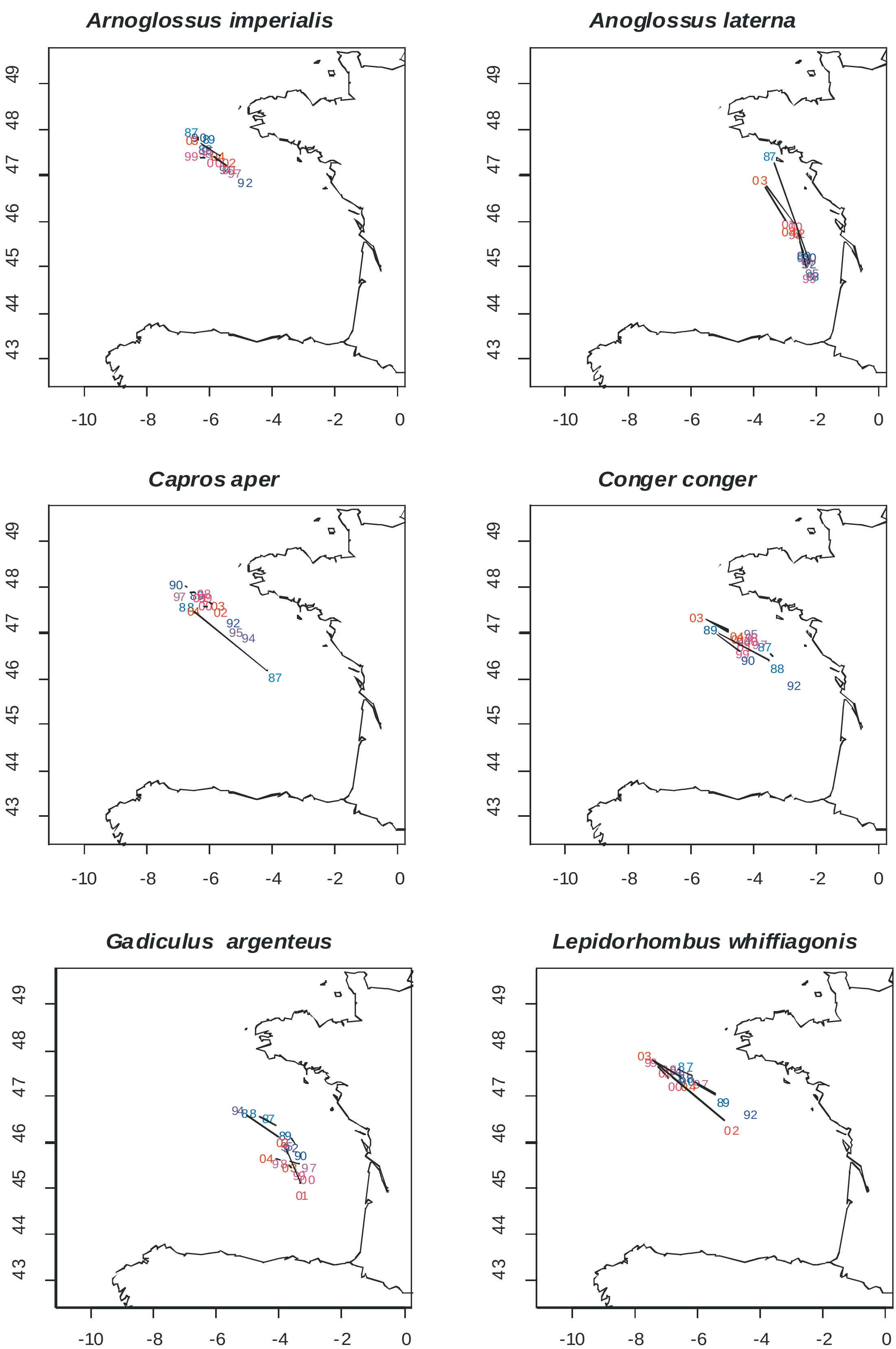


Figure 3.4-1: Variation of the location of the centre of gravity of different species densities during the autumn surveys carried out from 1987 to 2003.

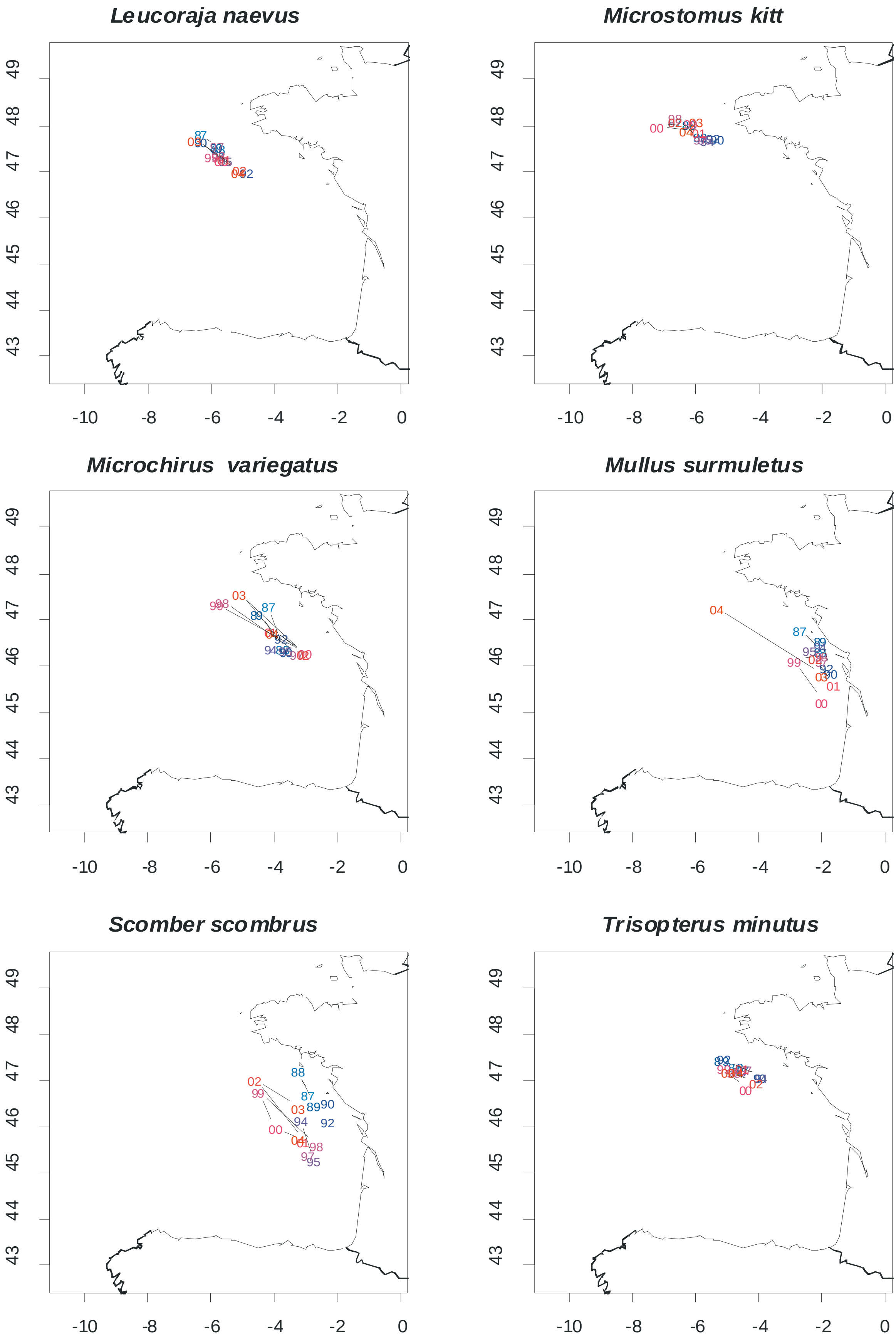


Figure 3.4-1 continued.

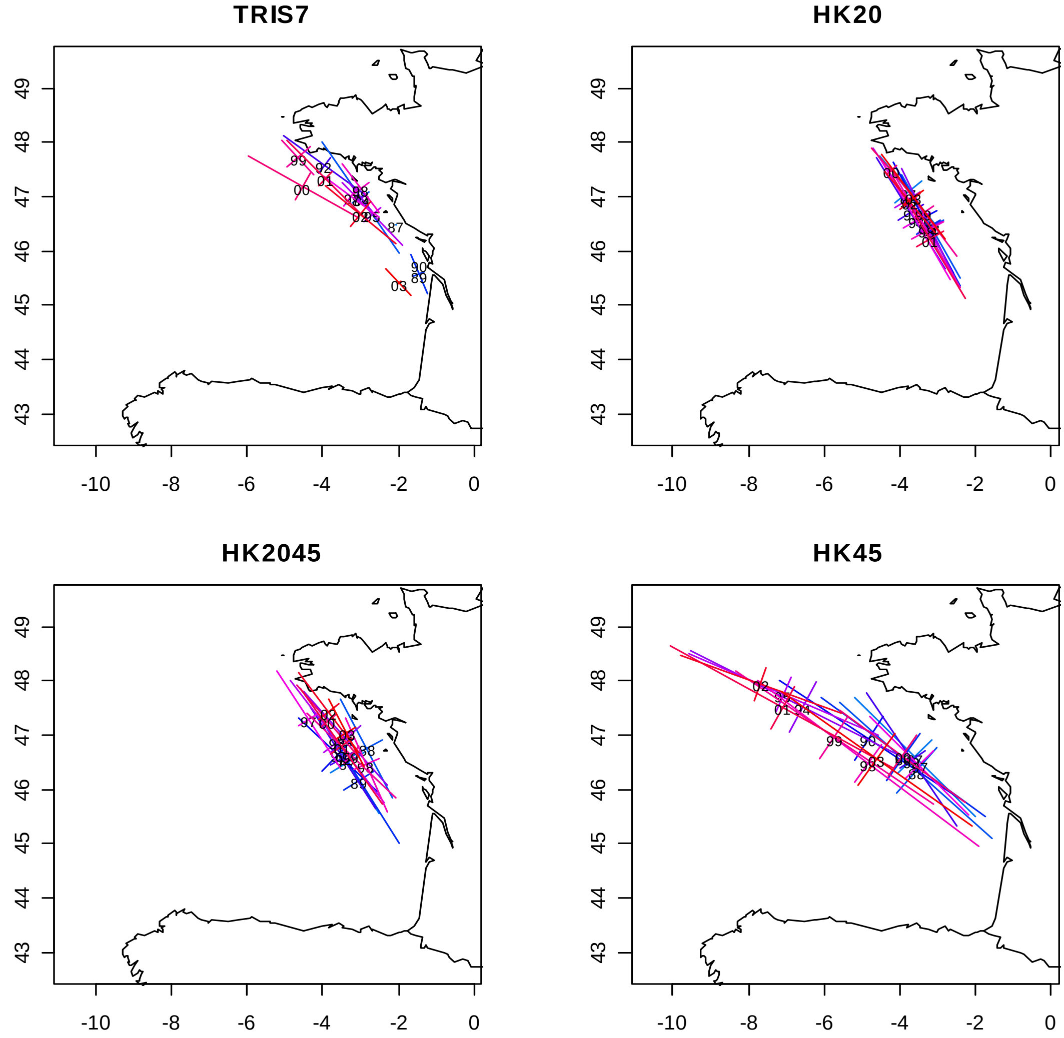


Figure 3.4-2: Centre of gravity and inertia of the location of *Trisopterus* spp smaller than 8 cm (TRIS7), *Merluccius merluccius* smaller than 20 cm (HK20), *Merluccius merluccius* longer than 19 and smaller than 45 cm (HK2045) and *Merluccius merluccius* longer than 44 cm (HK45) during the autumn surveys carried out from 1987 to 2003.

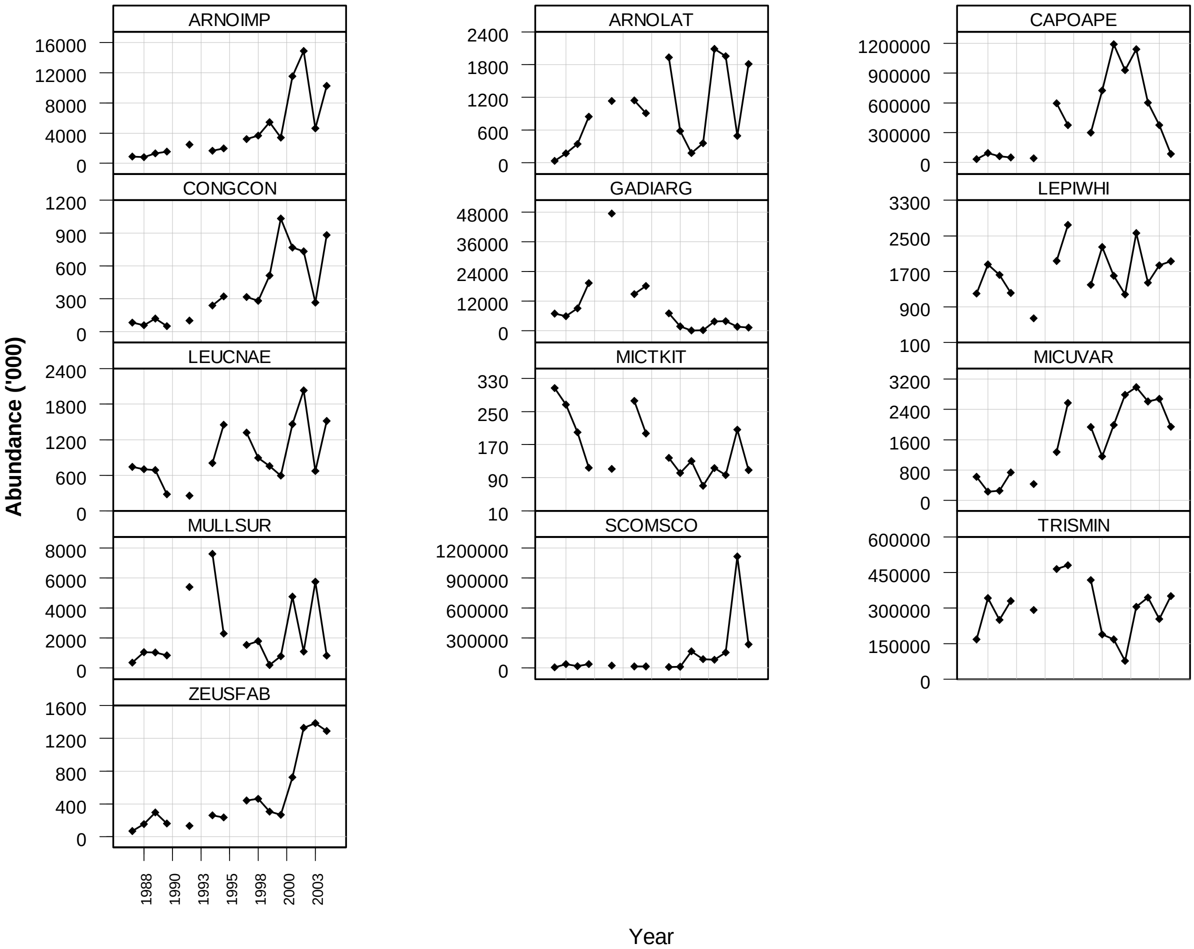


Figure 3.4-3: Changes in species abundance through time.

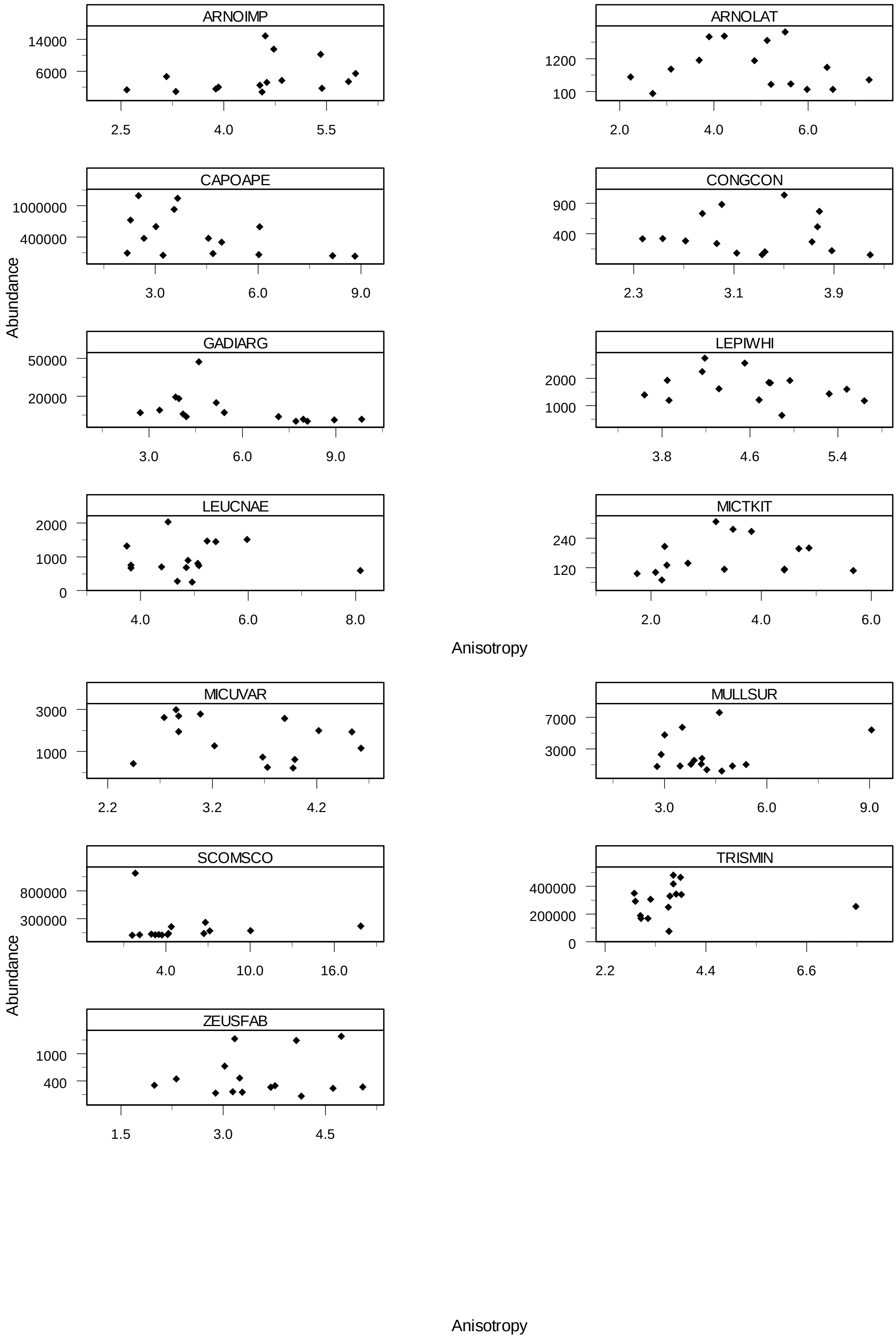


Figure 3.4-4: Scatter plot of species abundance versus anisotropy index.

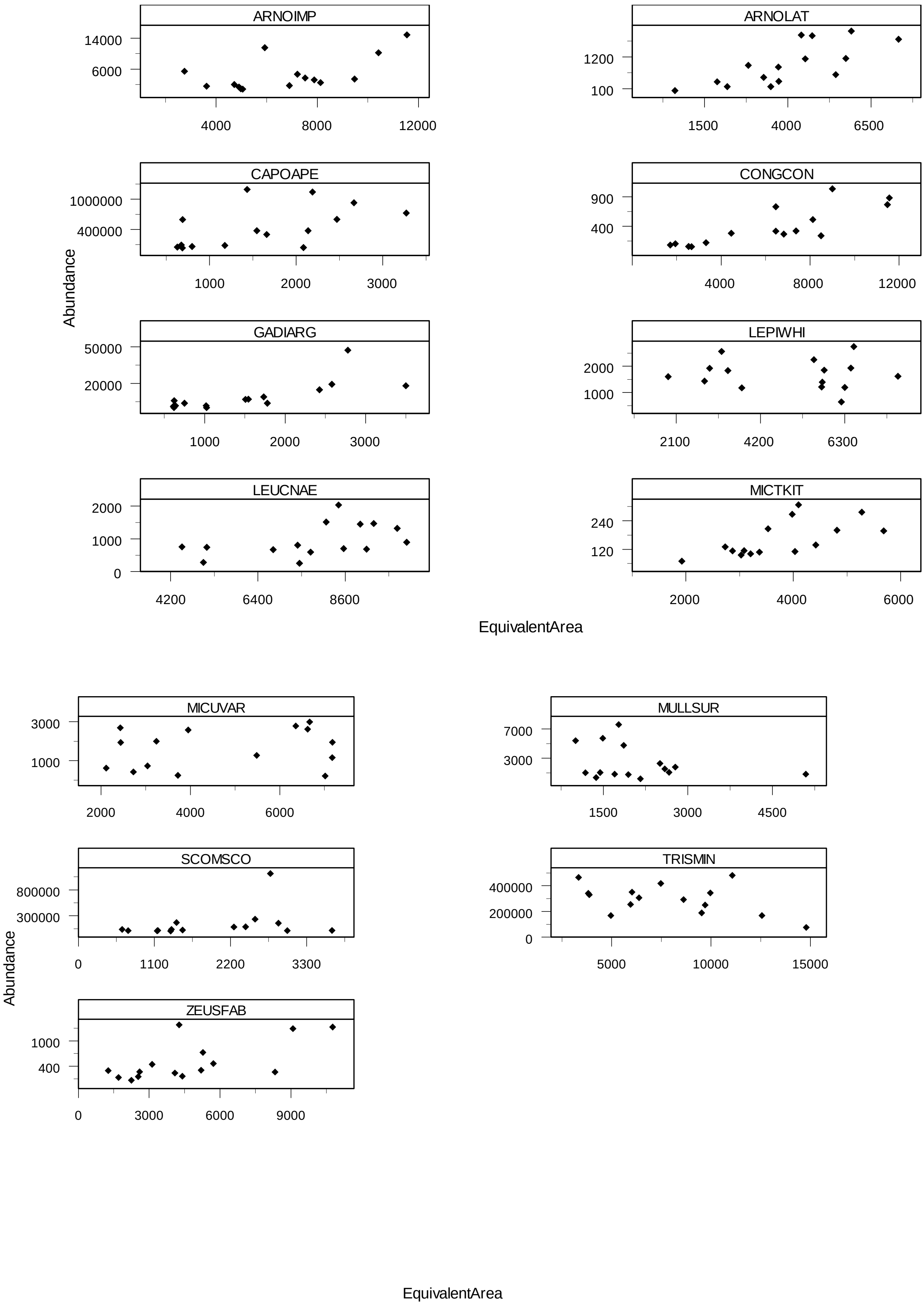


Figure 3.4-5: Scatter plot of species abundance versus equivalent area index.

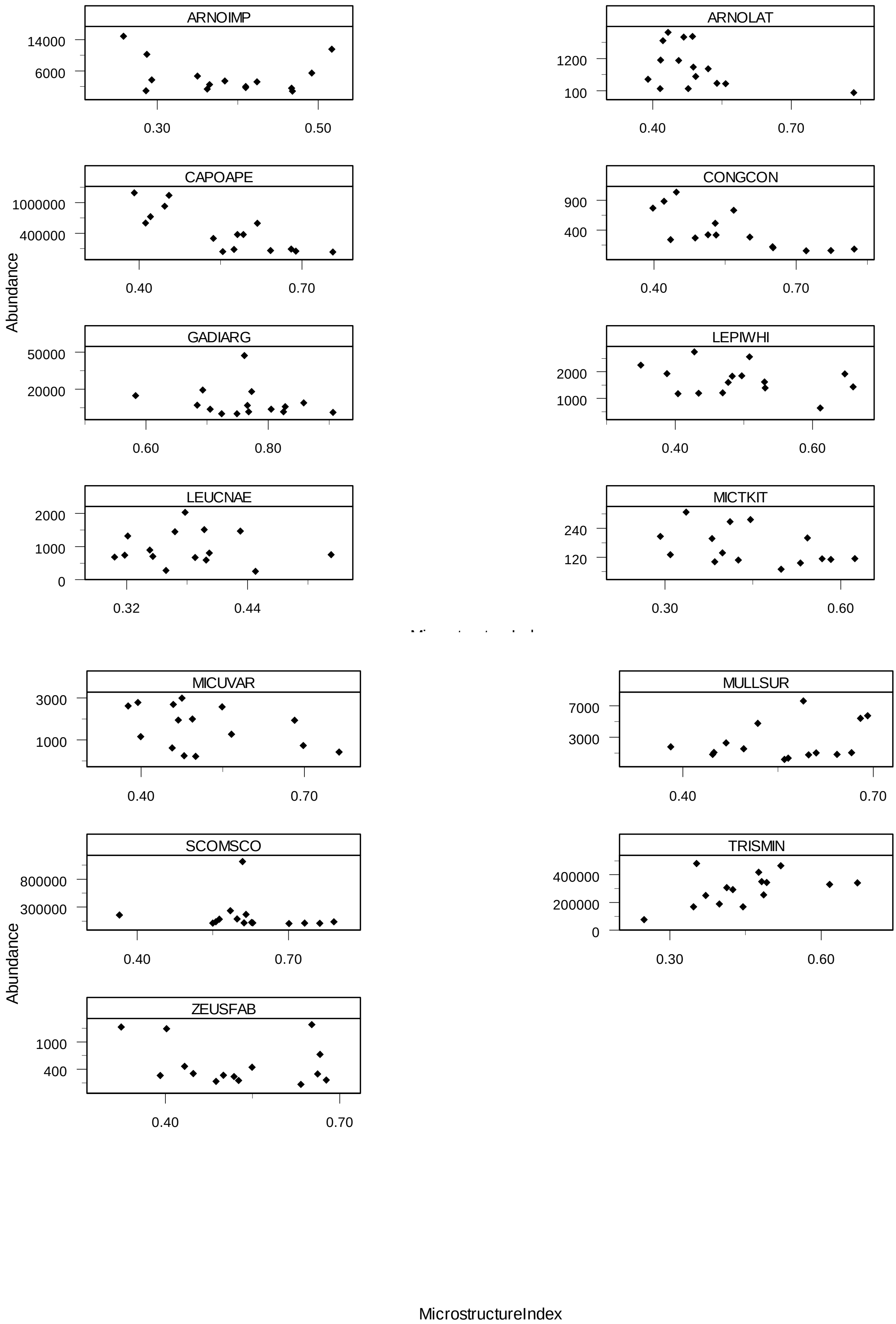


Figure 3.4-6: Scatter plot of species abundance versus microstructure index.

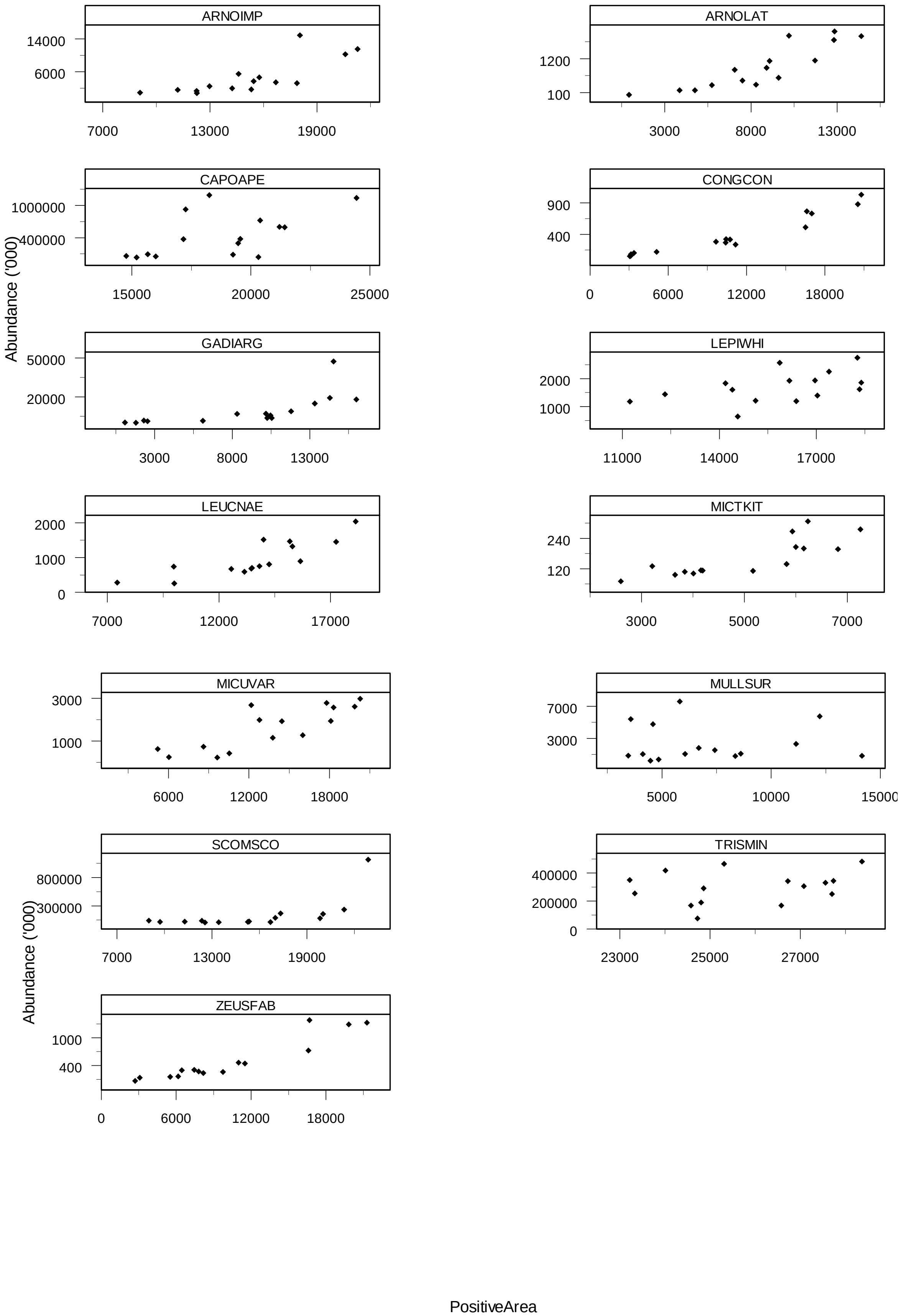


Figure 3.4-7: Scatter plot of species abundance versus positive area index.

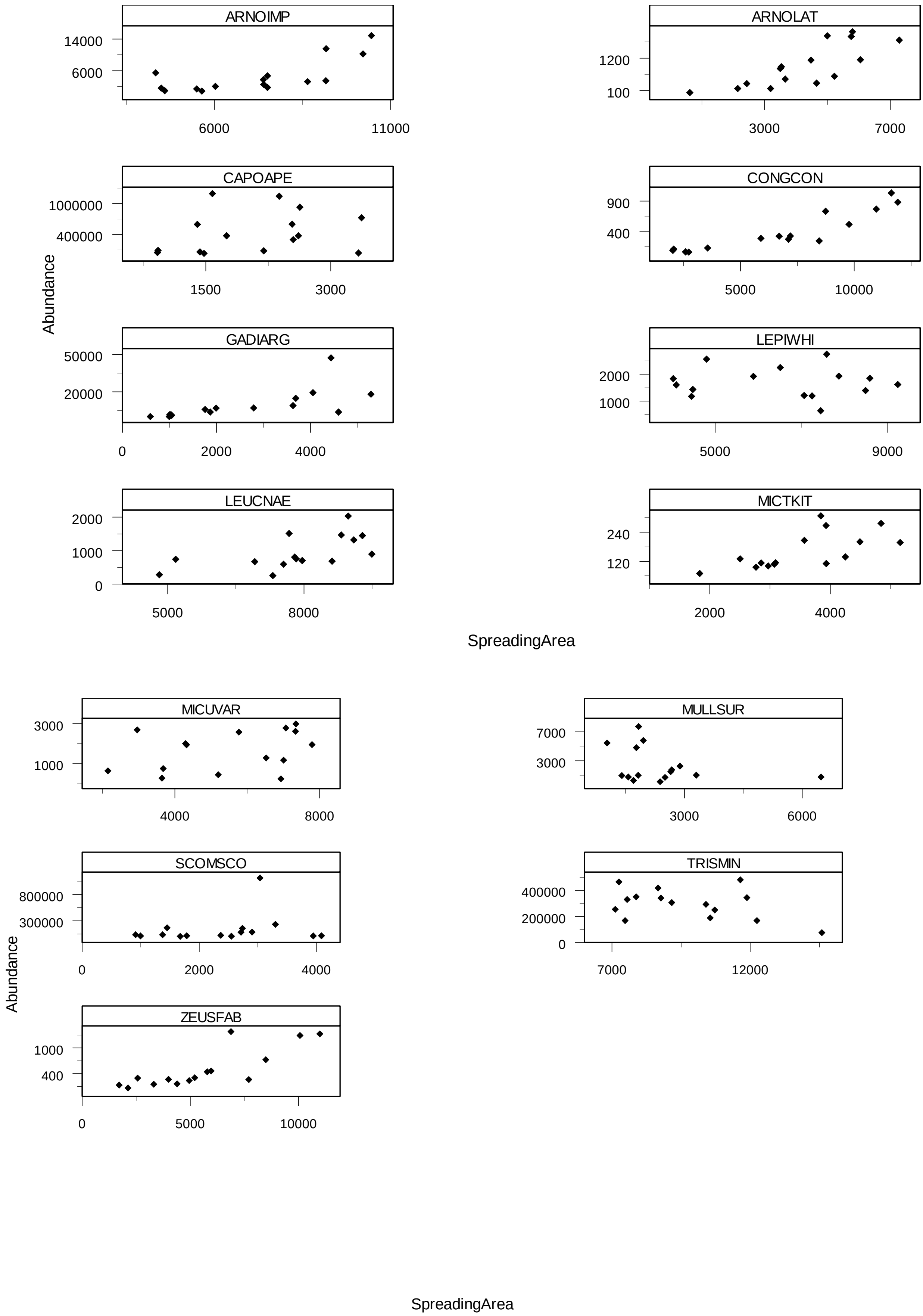


Figure 3.4-8: Scatter plot of species abundance versus spreading area index.

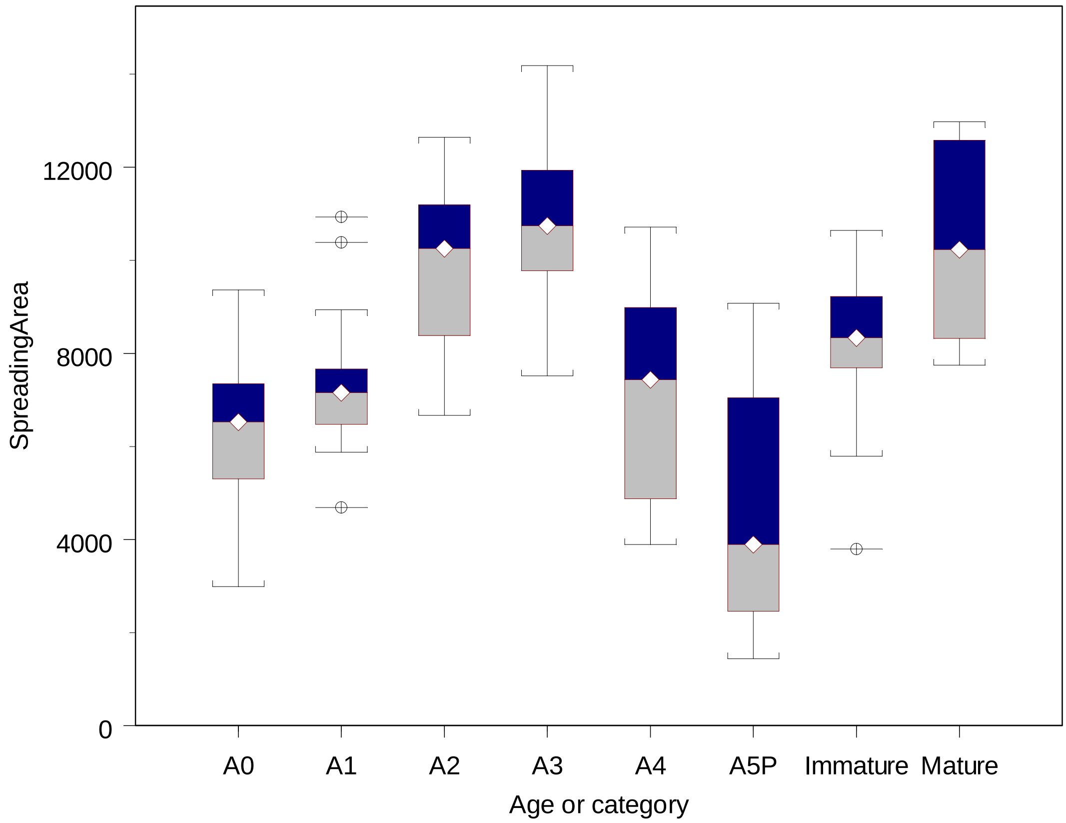


Figure 3.4-9: Average spreading area index for hake (*Merluccius merluccius*) age groups 1 to 5+, immature and mature individuals computed from the data collected during autumn surveys carried out from 1987 to 2004.

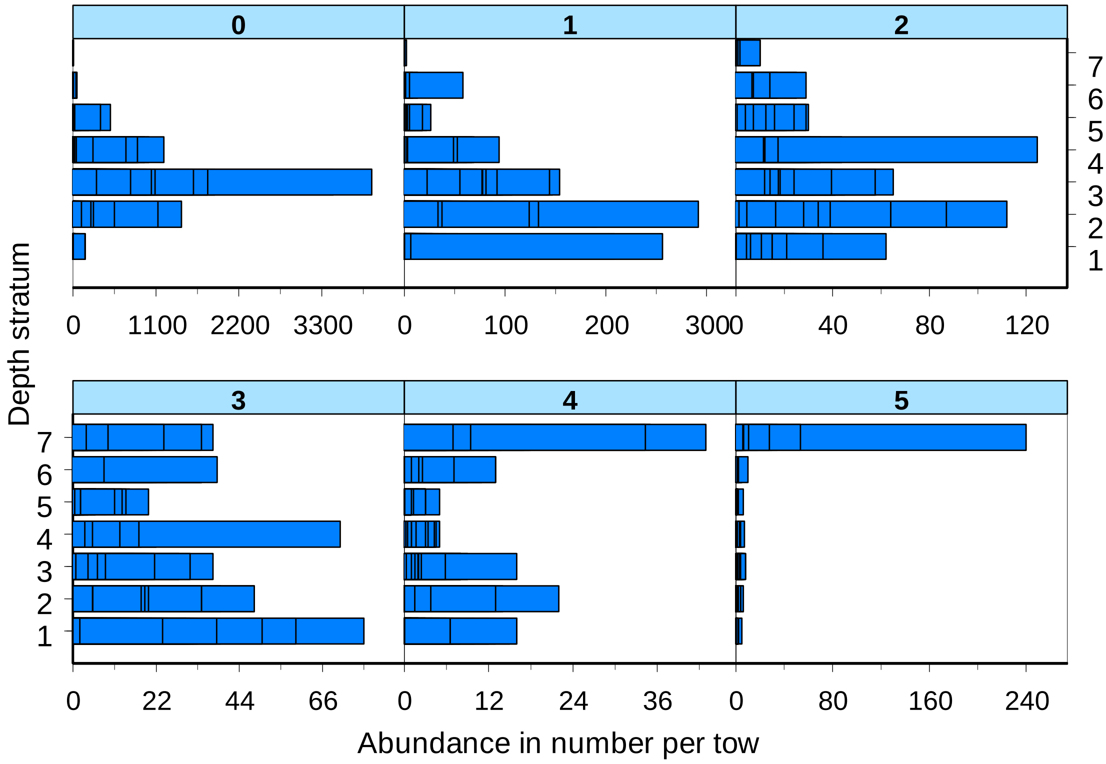


Figure 3.4-10: Average depth distribution of hake age groups 0 to 5+ in autumn. Depth range of stratum: 1 less than 30 m; 2 from 30 to 80 m; 3 from 80 to 120 m; 4 from 120 to 160 m; 5 from 160 to 200 m; 6 from 200 to 400 m; 7 from 400 to 600 m.

Table 3.4-1: Test for correlation between species abundance and five spatial indices using Kendall's tau-statistic p-value.

	Spatial indices				
Species	Anisotropy	Equivalent Area	Microstructure Index	Positive Area	Spreading Area
Arnoglossus imperialis	0.042	0.051	0.756	0.000	0.004
Arnoglossus laterna	0.672	0.001	0.924	0.000	0.000
Capros aper	0.979	0.010	0.998	0.017	0.128
Conger conger	0.771	0.000	0.999	0.000	0.000
Gadiculus argenteus	0.997	0.000	0.826	0.000	0.000
Lepidorhombus whiffiagonis	0.872	0.480	0.707	0.034	0.480
Leucoraja naevus	0.293	0.076	0.520	0.000	0.013
Microchirus variegatus	0.924	0.260	0.949	0.001	0.076
Microstomus kitt	0.174	0.006	0.892	0.000	0.004
Mullus surmuletus	0.559	0.707	0.200	0.293	0.598
Scomber scombrus	0.008	0.076	0.924	0.003	0.200
Trisopterus minutus	0.091	0.924	0.027	0.149	0.909
Zeus faber	0.441	0.006	0.826	0.000	0.000

Table 3.4-2: Correlation coefficient between species abundance and five spatial indices using Kendall's tau-statistic: estimate tau. In bold, estimate tau corresponding to p values less than 0.05.

ESTIMATE.TAU	SPATIAL INDICES				
SPECIES	ANISOTROPY	EQUIVALENT AREA	MICROSTRUCTURE INDEX	POSITIVE AREA	SPREADING AREA
Arnoglossus imperialis	**0.33**	0.31	-0.13	**0.64**	**0.50**
Arnoglossus laterna	-0.09	**0.58**	-0.28	**0.79**	**0.64**
Capros aper	-0.39	**0.45**	-0.56	**0.41**	0.22
Conger conger	-0.14	**0.66**	-0.58	**0.87**	**0.75**
Gadiculus argenteus	-0.52	**0.68**	-0.18	**0.77**	**0.75**
Lepidorhombus whiffiagonis	-0.22	0.01	-0.10	**0.35**	0.01
Leucoraja naevus	0.10	0.28	-0.01	**0.70**	**0.43**
Microchirus variegatus	-0.28	0.12	-0.31	**0.60**	0.28
Microstomus kitt	0.18	**0.49**	-0.24	**0.66**	**0.50**
Mullus surmuletus	-0.03	-0.10	0.16	0.10	-0.05
Scomber scombrus	**0.47**	0.28	-0.28	**0.52**	0.16
Trisopterus minutus	0.26	-0.28	**0.37**	0.20	-0.26
Zeus faber	0.03	**0.49**	-0.18	**0.79**	**0.70**

Table 3.4-3: Test for correlation between hake age group abundance and five spatial indices using Kendall's tau-statistic: p value. In bold:, values less than 0.05.

P.VALUE	SPATIAL INDICES				
AGE OR CATEGORY	ANISOTROPY	EQUIVALENT AREA	MICROSTRUCTURE INDEX	POSITIVE AREA	SPREADING AREA
A0	0.740	0.480	0.944	**0.010**	0.402
A1	0.200	0.924	0.826	**0.001**	0.987
A2	0.635	0.800	0.954	**0.000**	0.872
A3	**0.013**	0.365	0.461	**0.001**	0.520
A4	**0.010**	0.987	0.149	**0.000**	0.998
A5P	0.149	0.966	0.091	**0.001**	0.958
Immature	0.771	0.707	0.958	**0.030**	0.909
Mature	**0.015**	0.983	0.117	**0.008**	0.990

Table 3.4-4: Correlation coefficient between hake age group abundance and five spatial indices using Kendall's tau-statistic: estimate tau. In bold, estimate tau corresponding to p values less than 0.05

ESTIMATE.TAU	SPATIAL INDICES				
AGE OR CATEGORY	ANISOTROPY	EQUIVALENT AREA	MICROSTRUCTURE INDEX	POSITIVE AREA	SPREADING AREA
A0	-0.12	0.01	-0.30	**0.45**	0.05
A1	0.16	-0.28	-0.18	**0.58**	-0.43
A2	-0.07	-0.16	-0.32	**0.77**	-0.22
A3	**0.43**	0.07	0.02	**0.60**	-0.01
A4	**0.45**	-0.43	0.20	**0.83**	-0.56
A5P	0.20	-0.35	0.26	**0.60**	-0.33
Immature	-0.14	-0.10	-0.33	**0.36**	-0.26
Mature	**0.42**	-0.41	0.23	**0.47**	-0.45

3.5 Abundance-range size of case-study species

3.5.1 Introduction

Changes in distribution of six species, herring (*Clupeus harengus*), John Dory (*Zeus faber*), spurdog (*Squalus acanthias*), striped wolfish (*Anarhichas lupus*), northern wolffish (*A. denticulatus*) and spotted wolffish (*A. minor*) were examined over a period of 28 years in terms of changes in distribution, total area occupied and area occupied at the centre of mass was measured. Spatial indices; centre of gravity and inertia, anisotropy, count of spatial patches, positive area, spreading area, equivalent area and microstructure, described in the previous section were applied to selected species and areas within the surveyed portion of the northeast Atlantic (Figure. 3.5-1). In each case, a coherent data set comprising a single survey gear was selected for application of these indices.

3.5.2 Data sources and analysis

Data were obtained from seven demersal surveys from quarter's 3, 4 and 1 (See Table 3.5-1 for details of each of the survey gears and coverage). Figure 3.5-1 shows the locations of the sets and the accompanying caption describes the survey gears used. Because various different gears were used in these surveys (including several variations of GOV, plus Campelen and PHHT), the relative intensity of density of fish between areas where the different gears were deployed should be regarded with caution, as the survey gears are different. Because of different catchability among gears, degree of spatial variation in density is different among the different survey areas illustrated in Figure 3.5-1. For example, relative density is not directly comparable between the North Sea (GOV survey) and Barents Sea (Campelen), though changes in density patterns over time in each survey are is real.

Table 3.5-1: Summary details of demersal trawl surveys used in the present study.

SURVEY AND YEARS	AREA	GEAR	FOR FURTHER DETAILS OF GEAR AND SURVEY SEE:
Barents Sea (1981-2002)	Barents Sea (I and IIa)	Campelen trawl	Jakobsen *et al.* (1997); ICES (2004a)
IBTS – North Sea (1978-2005)	North Sea (IV) and Skagerrak (IIIa)	Standard GOV with ground gear A (or B in some areas)	ICES (2004b, 2005b)
IBTS – Scottish west coast (1990-2004)	North-west Scotland (VIa)	GOV with ground gear C	ICES (2004b, 2005b)
IBTS – English west coast (2004-2005)	Irish Sea (VIIa) and south-west Approaches (VIIe-h)	Modified GOV with ground gear D (hard ground) and ground gear A (fine ground)	ICES (2004b, 2005b)
English west coast survey (1982-2003)	Celtic Sea (VIIe-j)	Portuguese High Headline Trawl	Warnes and Jones (1995)
IBTS – EVEHOE (1987-2004)	Bay of Biscay (VIIIa-b) and Celtic Sea (VIIg-j)	GOV with ground gear A, but less kite	ICES (2004b, 2005b)
IBTS – Eastern Channel GFS (1988-2004)	Eastern English Channel (VIId)	Small GOV	ICES (2004b, 2005b)

Two gears, the PHHT (see Warnes and Jones (1995) for a description of the gear) and GOVA (standard GOV with ground gear A, but no kite) overlapped in time and space in the Celtic Sea. From 1990 to 2003, 1429 sets were used to compare the survey catch rate averaged over all sizes of fish between the two gears. A ratio of GOVA numbers per tow/PHHT numbers per tow was used to convert PHHT numbers to GOVA equivalents. Even though both of these gears have a cod end mesh size of 20 mm, the GOVA has a better graduation of mesh sizes down the length of the net, and so a stage-based conversion would have been more appropriate for this analysis. However, these data were not available for this analysis and, for the purpose of examining broadly based density distribution patterns; non-sized based conversion factor was deemed adequate.

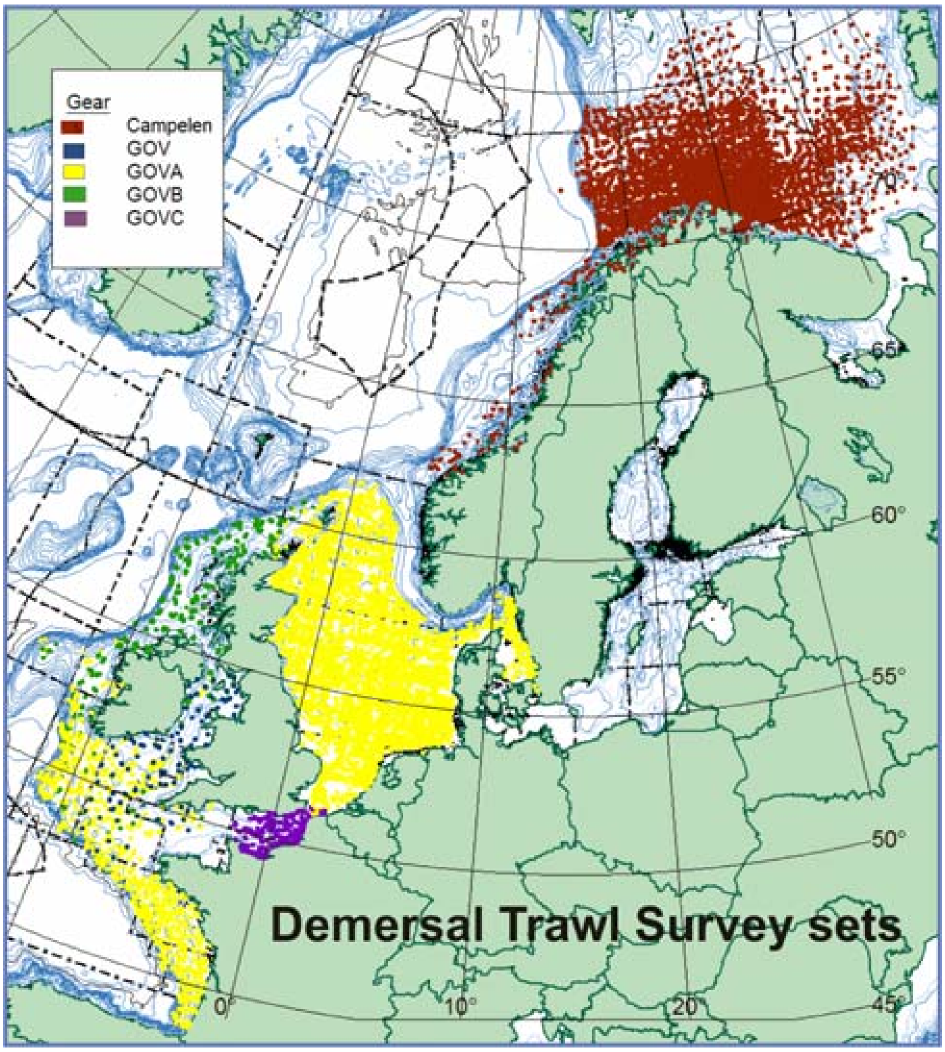


Figure 3.5-1: Distribution of sampling stations used in the present study, 1978-2005. Not all years are covered by each gear. Data correspond to quarter 3, 4 and to a lesser extent quarter 1. GOVA = standard GOV trawl with ground gear A (some nations use extra floats instead of a kite, or no kite), GOVB = GOV trawl used off NW Scotland with bobbin ground gear, GOV = modified GOV trawl used by UK in westerly surveys, GOVC = small GOV trawl used by France in CGFS. Refer to the reports of the International Bottom Trawl Survey Working Group (IBTSWG) and Study Group on Survey Trawl Gears (SGSTG) for descriptions of the GOV trawls (e.g. ICES, 2004b, 2005b). The UK Q1 survey with PHHT was standardized with the GOVA (see Section 3.5.2).

Survey data from the Bay of Biscay, Celtic Sea and Irish Sea was not available before 1982 and was limited in extent prior to 1987 (Figure 3.5-2). For this reason, for species that occurred both east and west of Britain, changes in area occupied are not comparable before and after 1987.

Surfaces depicting density or local abundance based on survey abundance were produced for each of the six case-study species over six time periods shown in Figure 3.5-13. The species selected included a wide-ranging pelagic teleost (herring), a wide-ranging elasmobranch (spurdog), a southern demersal teleost (John Dory) and northern demersal teleosts (three species of wolfish). Potential mapping in SPANS GIS was used to investigate spatial distribution. SPANS (Anon, 2000). Potential mapping transforms points (surveys set kg per hour) to density surfaces (areas of similar kg per tow) by placing a circle around each point and averaging the values of all points that fall within the circle. The circle size selected (30 km diameter) provided complete coverage of the survey area while minimizing gaps in the density surface and thus maximizing spatial resolution. The resulting map was then post-stratified into 15 classes defining density of the fish, each density class covering approximately the same amount of area. For each species, the density classes were held constant and were based on the average distribution over the 28-year period. The method is described in detail in Kulka (1998). Total area occupied, area occupied and the ratio of these two values were used to examine spatial changes over time, the latter indicating changes in degree of concentration.

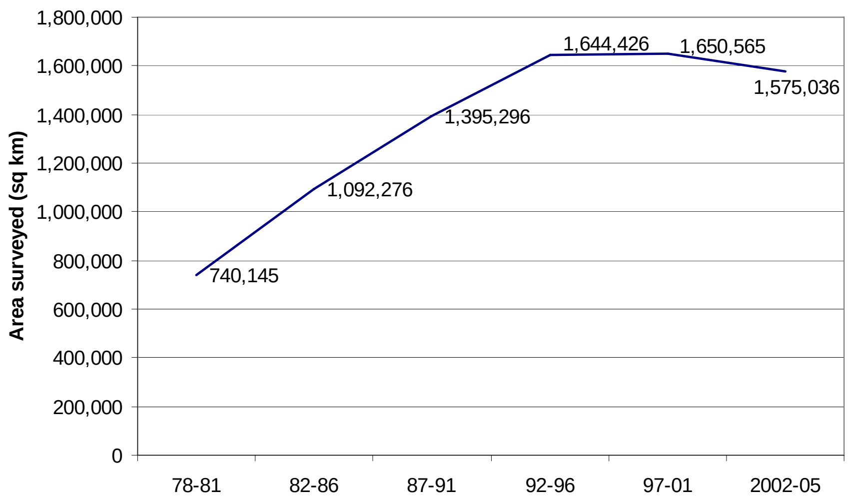


Figure 3.5-2: Area covered (km^2) by the six demersal surveys illustrated in Figure 3.5-11 during six time periods. The areas sampled by surveys have increased since the 1970s and has been more stable since the 1990s.

The spatial indices; centre of gravity and inertia, anisotropy, count of spatial patches, positive area, spreading area, equivalent area and microstructure, were applied to these case-study species and areas where the survey gear was the same for each analysis. For herring, the population in the North Sea and adjacent Skagerrak was studied using IBTS Quarter 1 survey data for the years 1989–2004. For John Dory, bottom trawl data covering the Bay of Biscay and Celtic Sea for the period 1997–2004 in the fourth quarter was chosen (French and English data). Spurdog analyses encompassed a region stretching from the central Celtic Sea to the north of Scotland for the period 1990–1997 in the fourth quarter (Scottish data). Striped wolfish was analysed for the Barents Sea in the first quarter for the years 1981-2002 (northern Norwegian survey data).

In addition, demersal Campelen survey data from the Barents Sea from 1981–2002 during January-March was used to examine changes in abundance. Data from a coastal Campelen survey targeting saithe and juvenile herring in October and November from 2004 and 2005 along the Norwegian coast were also used. There were several modifications in the Barents Sea survey configuration. The ground gear was changed in 1989. The mesh size was changed in 1994. In 1993 the sampling strategy changed from random stratified to a stratified regular grid (Jakobsen *et al*. 1997). In 1996 the strata layout was modified. Also a portion of the area was not systematically covered before 1993 when the area covered by the survey was expanded to the north and east. In 1997 and 1998, the Norwegian based survey did not cover the Russian economic zone. In most years the coverage towards north and east has been limited by the ice conditions. Since cod has been a target species in these surveys the area coverage since 1993 has been somewhat adopted to the observed distribution of cod.

3.5.3 Species analyses

The long-term distribution of six species, herring, John Dory, spurdog, striped wolffish, northern wolffish and spotted wolffish in the northeast Atlantic is illustrated over a wide area in Figure 3.5-3. Such a spatially wide treatment of demersal survey data has not been previously attempted. The purpose is to provide a spatial overview of species distribution.

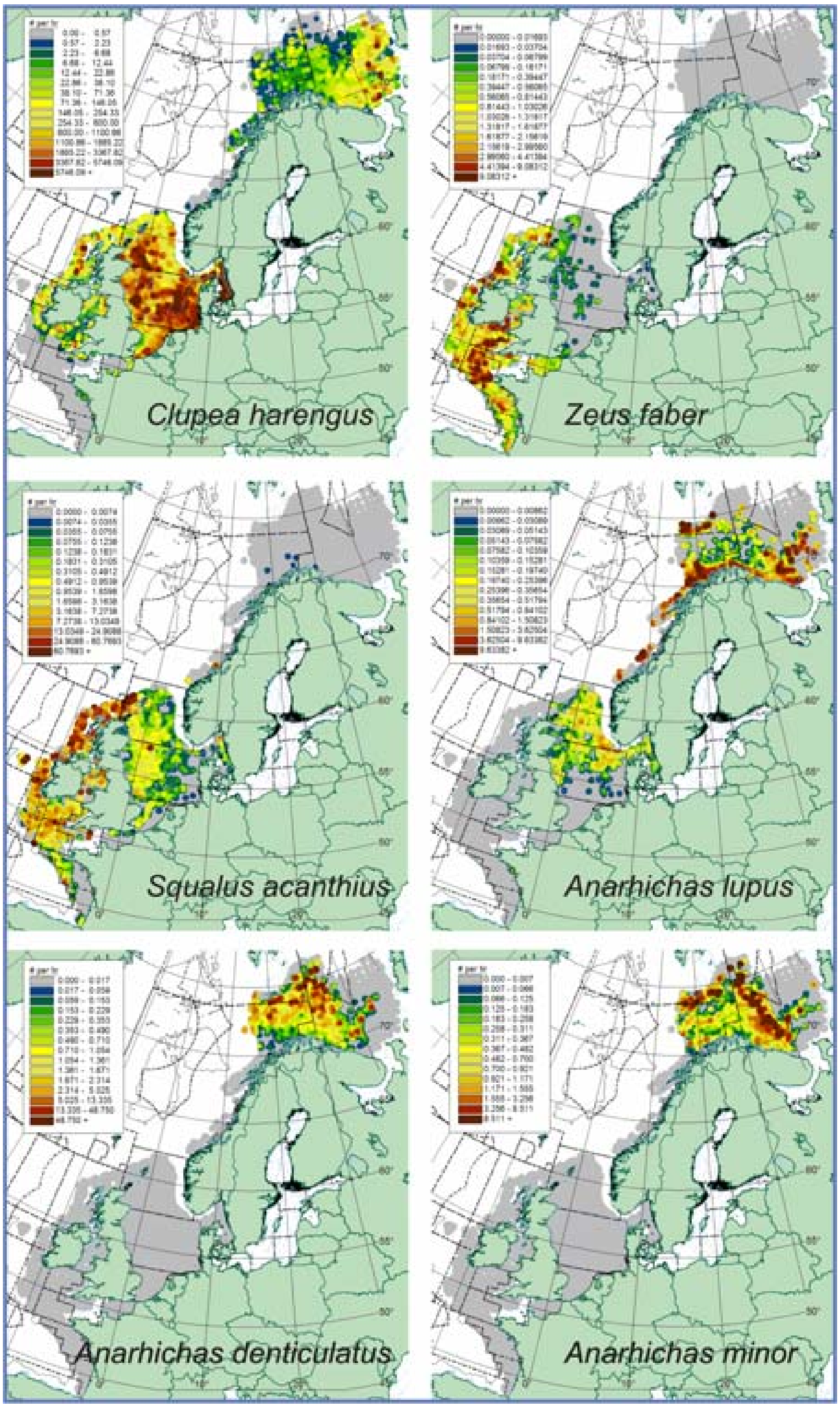


Figure 3.5-3: Distribution of six species based on a combination of data from six demersal surveys, 1978-2005 (refer to Figure 3.5-11). Grey areas are surveyed but with no catches. Red areas delineate the highest densities.

3.5.4 Herring

Herring was the most widespread of the six species covering 77% of the survey area (Figure 3.5-4). The only area where herring were largely absent from was the deeper parts of the Celtic Sea and the Bay of Biscay. Based on acoustic surveys, the distribution of this species is known to extend well beyond the demersal trawl survey area. Furthermore, the demersal surveys were observed to capture primarily juveniles of the species. Thus, the distribution of the total population of herring in the northeast Atlantic is under-represented by information contained in the demersal surveys. Densities appear higher in the North Sea compared to the Barents Sea but this may be a gear affect due to different catchability between gears and that the herring in the Barents Sea are mostly juvenile spring spawning herring (1 to 3–4 year olds).

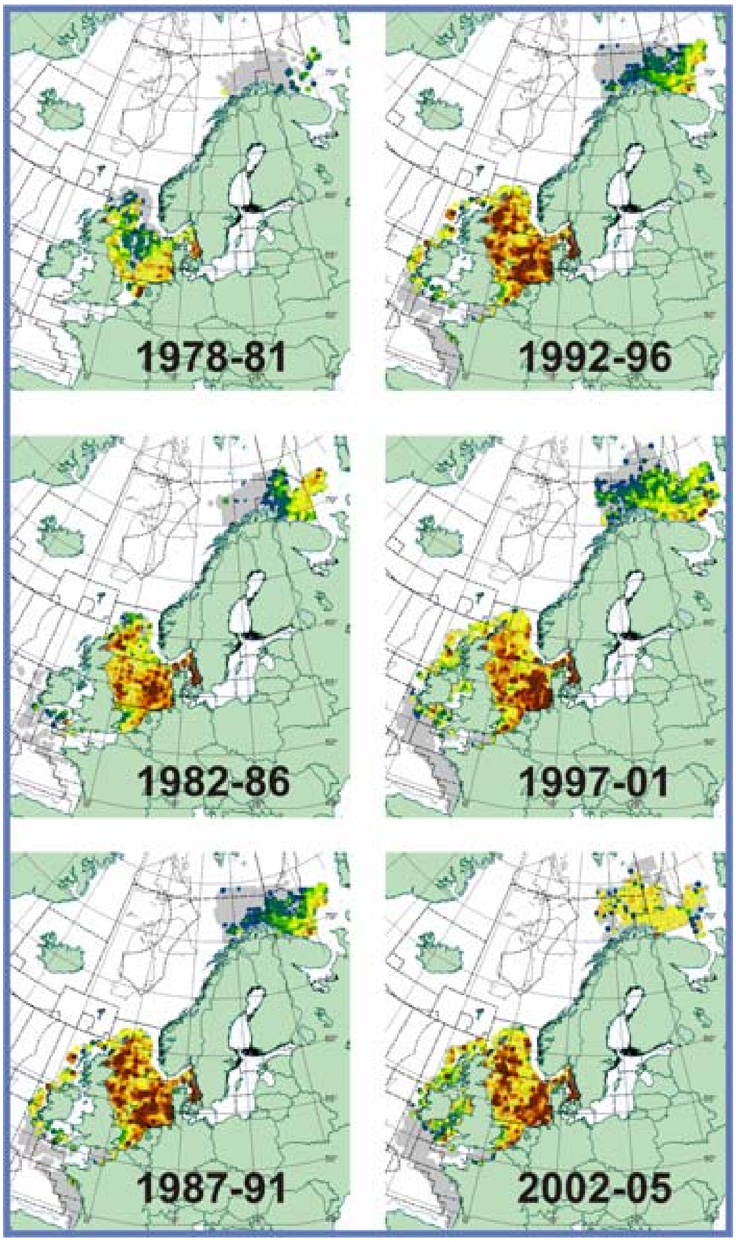


Figure 3.5-4: Distribution of herring during six periods using the combined demersal surveys (refer to Figure 3.5-11).

The distribution of herring in the North Sea showed spatial consistency over the entire period. The centre of mass was located mainly to the southeast with the lowest densities in the central part of the North Sea during all periods observed. In the Barents Sea, the highest concentration of herring was to the east. Larvae of spring spawning herring are transported passively into the east of the Barents Sea from the coast of Norway in summer. In winter juvenile Barents Sea herring has an eastern distribution because of favourable food and temperature conditions.

The area occupied by herring was relatively constant over the period of the surveys. However, the area of highest density increased substantially between 1978 and 1991 and was relatively stable thereafter (Figure 3.5-5). Given that much of the area of higher concentration of this species occurs east of the British Isles, this observed increase likely reflects an actual increase in the density of fish at the centre of mass. This pattern is also reflected in Figure 3.5-16, which shows an increase in degree of concentration from 1978 to 1991. This observation, of a consistent area occupied but an increase in density is consistent with the increase in abundance of herring during this period.

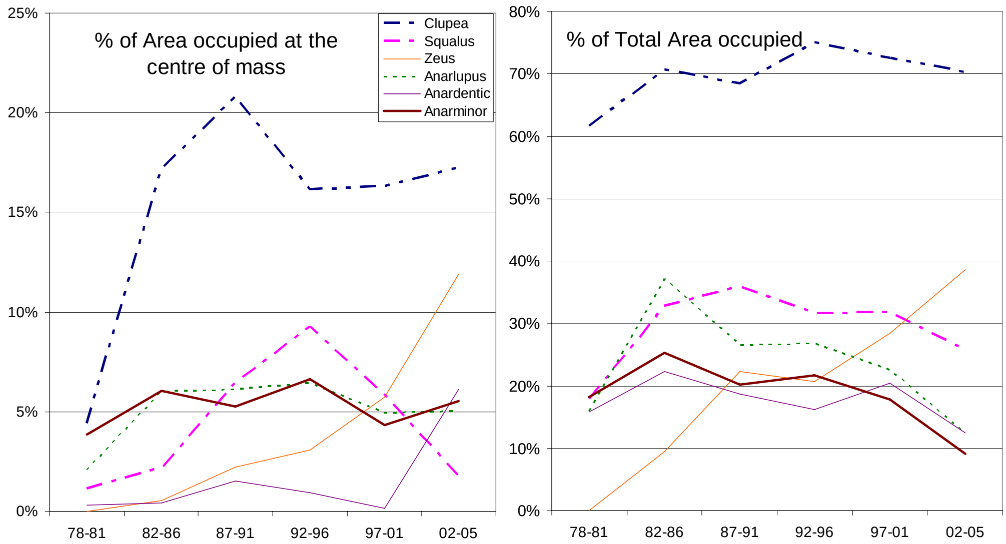


Figure 3.5-5: Change in percent of area occupied for six time periods. The first three periods include a smaller area surveyed refer to Figure 3.5-12) and thus may over or underestimate percent of area occupied.

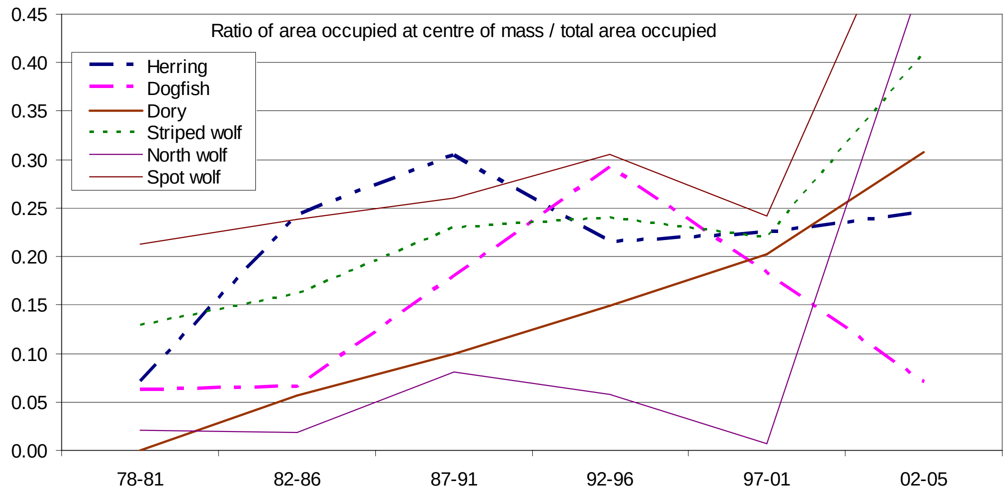


Figure 3.5-6: Ratio of area occupied at the centre of mass/total area occupied.

The centre of gravity for herring in the North Sea moved substantially from year to year, but was consistently located in the southern sector (Figure 3.5-7). The main axis of inertia was generally orientated east to west with the exception of 2004. This location coincides with the largest patch located to the west of Norway in that year (Figure 3.5-8). In all other years the most abundant patch was either found in the Kattegat or in the southern North Sea. Estimated herring abundance varied between years with no clear trend. No clear relationship between this abundance index and any of the spatial indices was apparent (Figure 3.5-9).

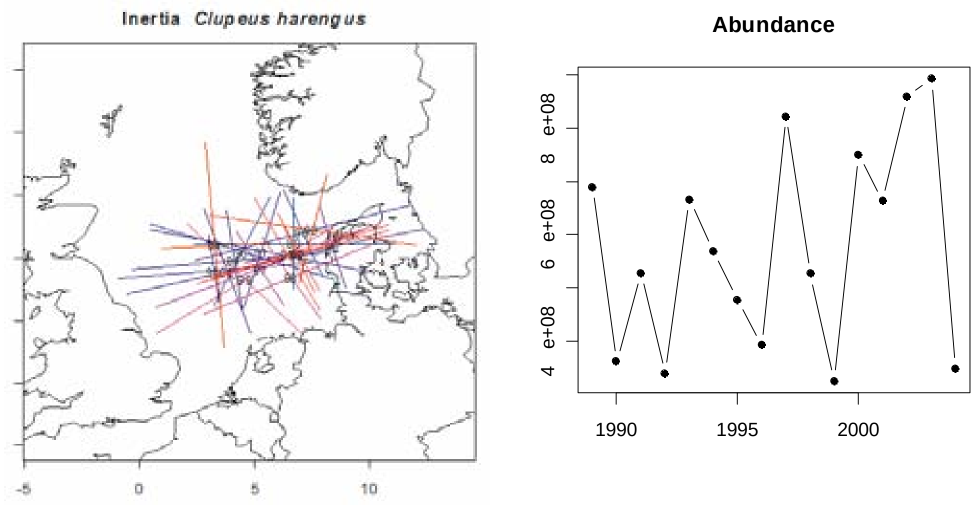


Figure 3.5-7: Change in centre of mass of herring in the North Sea and estimated abundance time series.

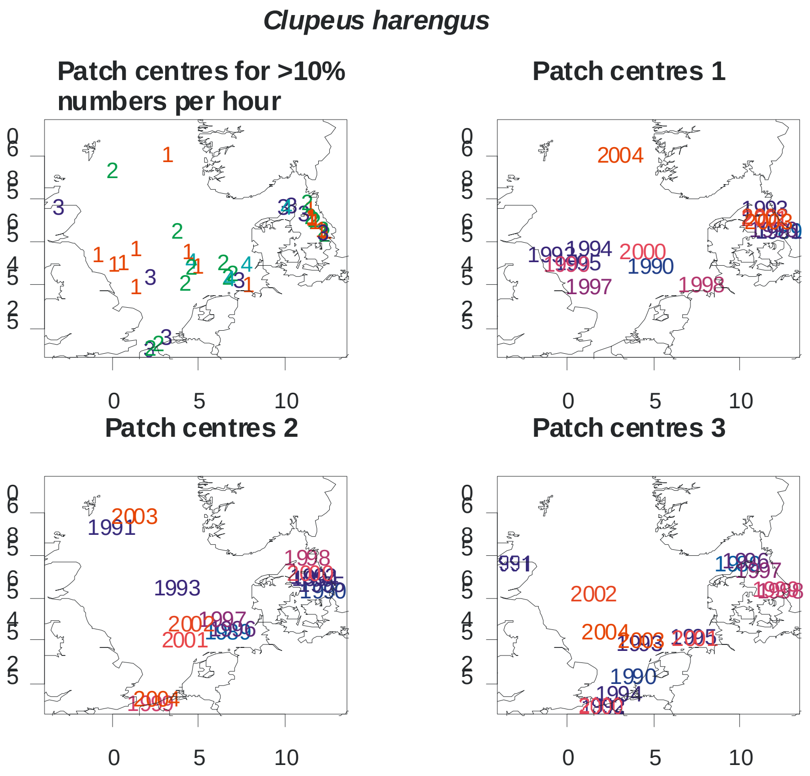


Figure 3.5-8: Location of first three most abundant spatial patches for *C. harengus* in the North Sea and Kattegat.

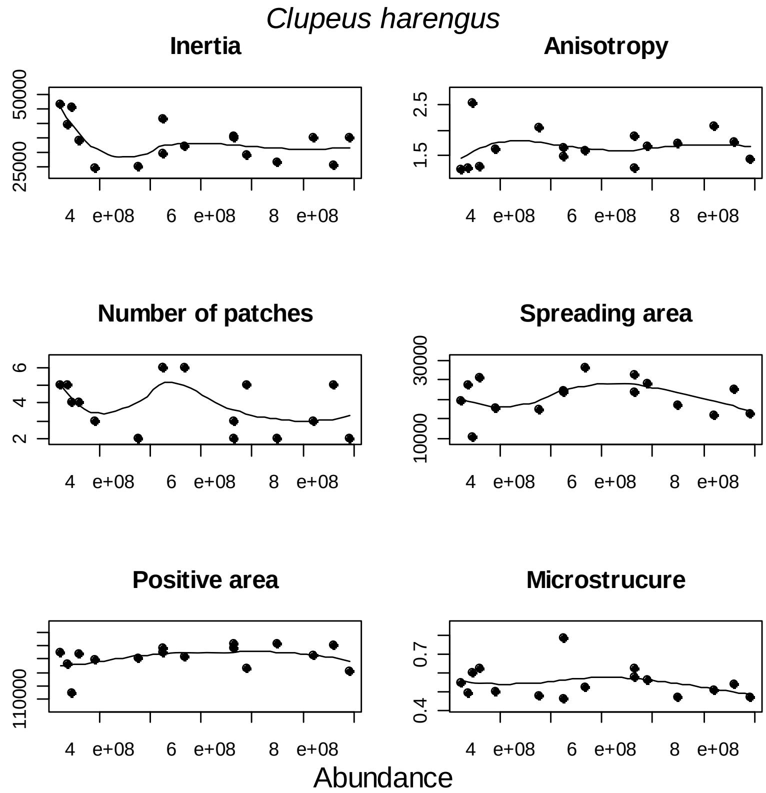


Figure 3.5-9: Spatial relationships (y axis) with abundance for herring.

3.5.5 John Dory

John Dory was constrained mainly to west and southwest of Britain over the entire period covering 38% of the survey area over the long term. The highest concentrations occurred in the Celtic Sea (Figure 3.5-3). John Dory underwent the greatest change of any of the six species, increasingly spreading into the survey area from the west (Figure 3.5-10). Prior to 1987, the density of John Dory was low west of Great Britain and was largely absent in the North Sea. Following the early 1990s, John Dory was more widely distributed and at greater density, and increasingly observed in the North Sea, mainly from the north. During this same period, the density was increasing particularly along the outer part of the Bay of Biscay and Celtic Sea. The reasons for this increase are not currently known.

The total area occupied by John Dory increased from about 21% in 1987–1991 to 38% of the surveyed area in the most recent period (Figure 3.5-5). The value for 1978-1990 underestimates the actual area occupied, given the low coverage by the survey west of Britain during this period. As well an increase in area occupied, areas of high concentration expanded at a greater rate than increase in total area over time (Figure 3.5-6).

The centre of gravity for John Dory was located in the Celtic Sea in all years (Figure 3.5-11). The main inertia axis was always orientated Northwest-Southeast following the coastline in the Bay of Biscay. Estimated abundance increased over the time series (Figure 3.5-11). The most abundant patch was always located in the Bay of Biscay in recent years, while it was situated in the Celtic Sea in the period 1989–2000 (Figure 3.5-12) Abundance was positively related to spreading area and positive area (Figure 3.5-13). The number of spatial patches was highest for the largest observed abundance, which was negatively related to abundance (Figure 3.5-13).

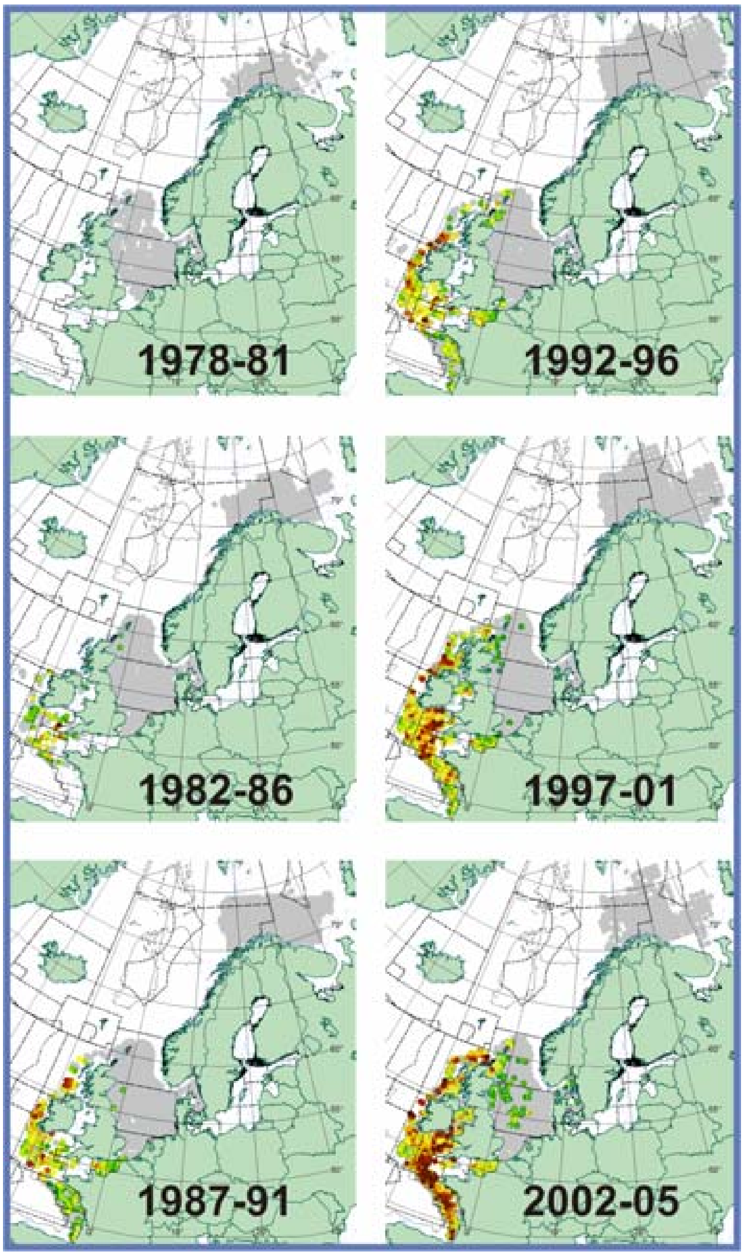


Figure 3.5-10: Distribution of John Dory during six periods using the combined demersal survey (refer to Figure 3.5-1).

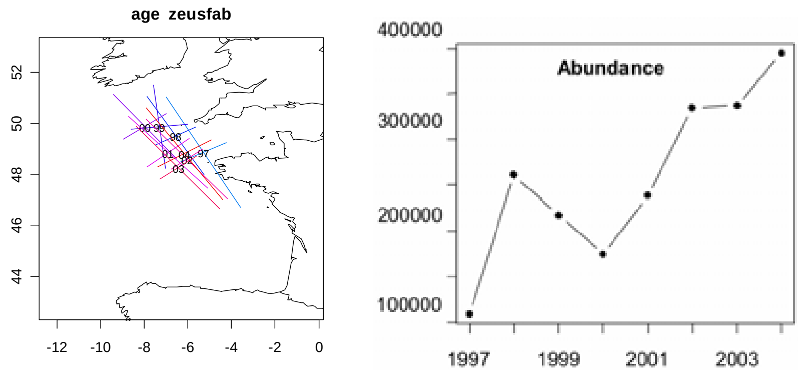


Figure 3.5-11: Change in inertia and centre of gravity of John Dory in the Bay of Biscay and Celtic Sea and estimated abundance time series.

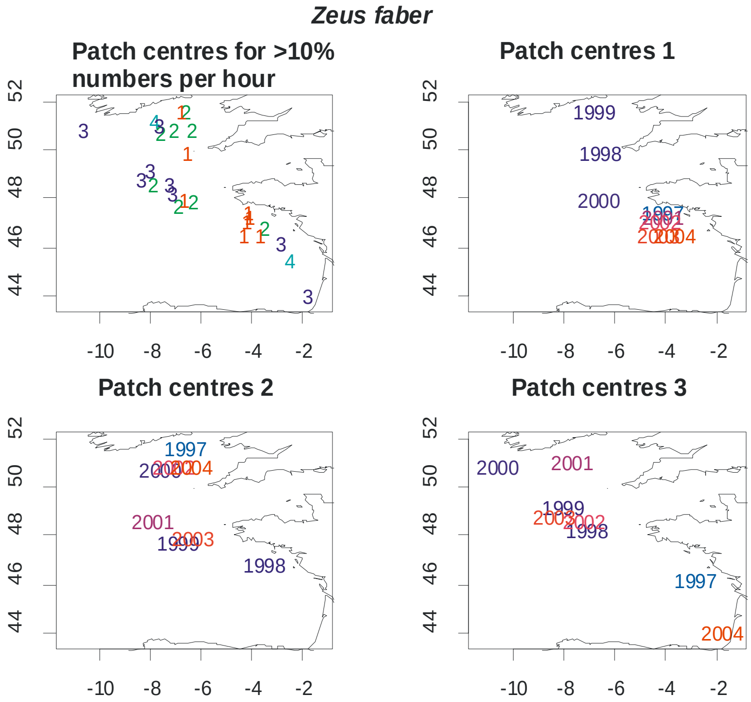


Figure 3.5-12: Location of first three most abundant spatial patches for Z. faber in the Bay of Biscay and Celtic Sea.

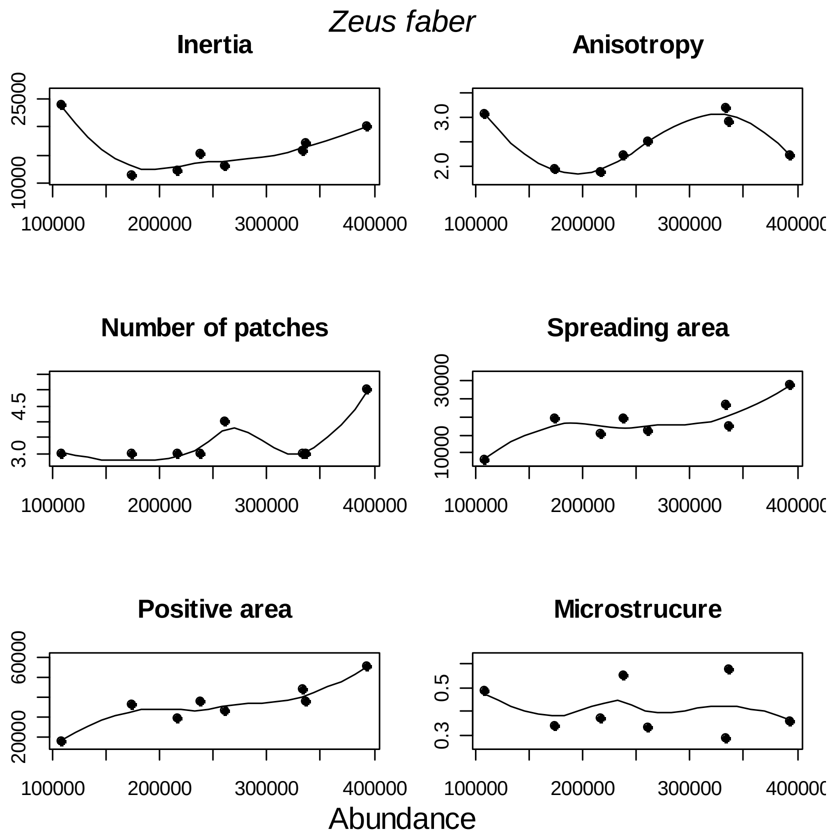


Figure 3.5-13: Spatial relationships (y axis) with abundance for John Dory.

3.5.6 Spurdog

Spurdog was restricted to the areas directly surrounding Britain in the Irish and Celtic Seas and in the eastern parts of the North Sea (Figure 3.5-3). These locations covered 42% of the survey area. The highest densities were observed to the east along the shelf edge.

During 1978–1981 and 2002–2005, spurdog in the North Sea were restricted to the northern half of the area, but were more widespread in the intermediate years from 1982–2001, extending to the English Channel (Figure 3.5-14). In all years when the Irish Sea was surveyed, spurdog were most concentrated toward the shelf edge. Concentrations were also observed in the Celtic Sea, and to a lesser extent in the Bay of Biscay.

In terms total area occupied, spurdog reached a peak in 1987–1991 and declined there after. The centre of mass, proportion of area occupied by high density concentrations peaked in 1992–1996 and declined rapidly thereafter (Figure 3.5-5). Concentration of spurdog peaked in 1992-96 (Figure 3.5-6). Spurdog are known to have declined in recent decades (Hammond and Ellis, 2005), and the proportion of survey hauls in which they occur has decreased in the last two decades (ICES, 2005c)

The centre of gravity for spurdog was located off Ireland in most years (Figure 3.5-14). The main inertia axis was always orientated Northeast-Southwest. Estimated abundance increased between 1990 and 1997 (Figure 3.5-15). No stability between years was found for the location of the three most abundant patches (Figure 3.5-16). A positive relationship between this abundance index and spreading area as well as anisotropy which was negatively related to abundance (Figure 3.5-17). Anisotropy expresses the difference in inertia (spatial variability of abundance) along the two main axes.

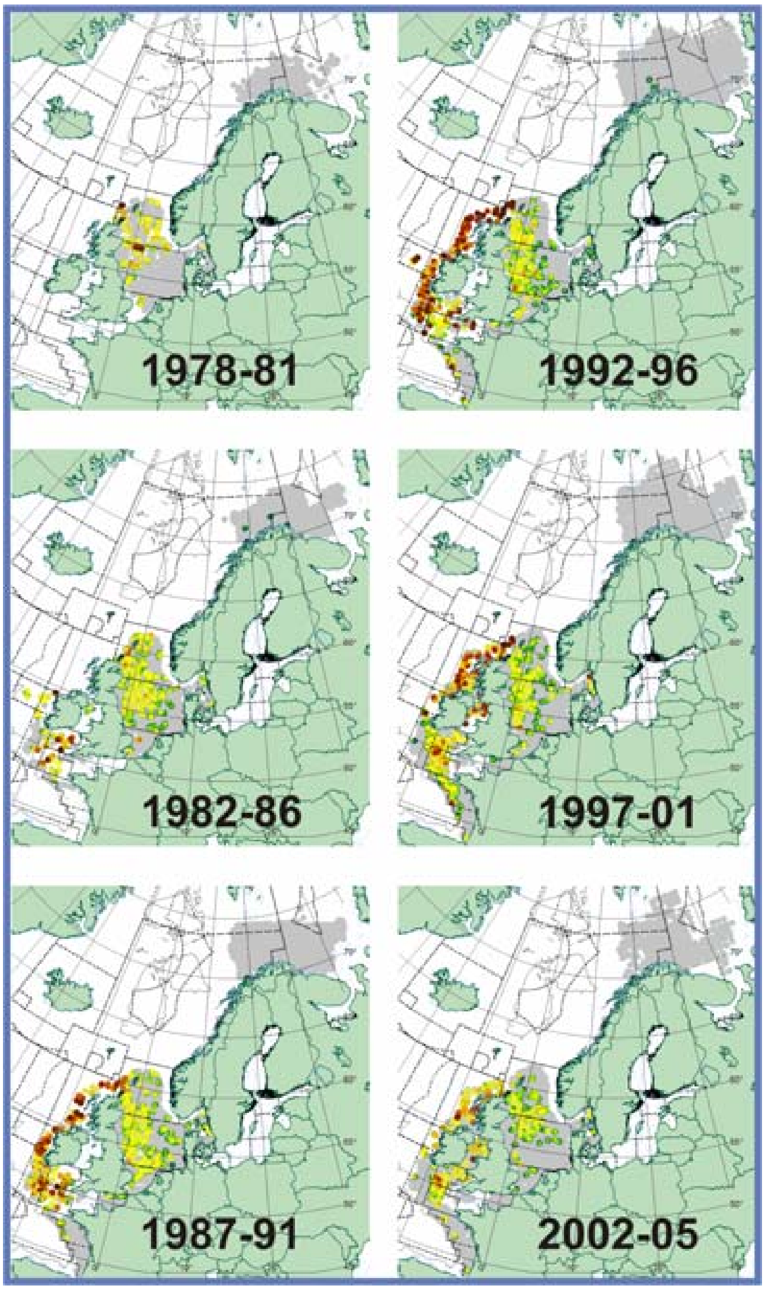


Figure 3.5-14: Distribution of spurdog during six periods using the combined demersal survey (refer to Figure 3.5-1).

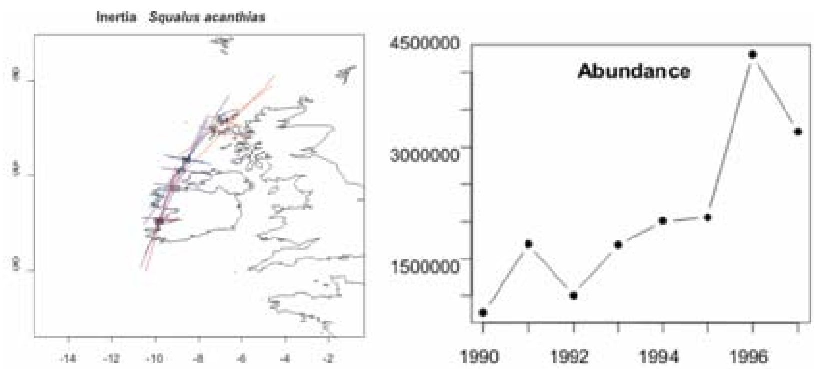


Figure 3.5-15: Change in inertia and centre of gravity of spurdog west off Ireland and Scotland and estimated abundance time series.

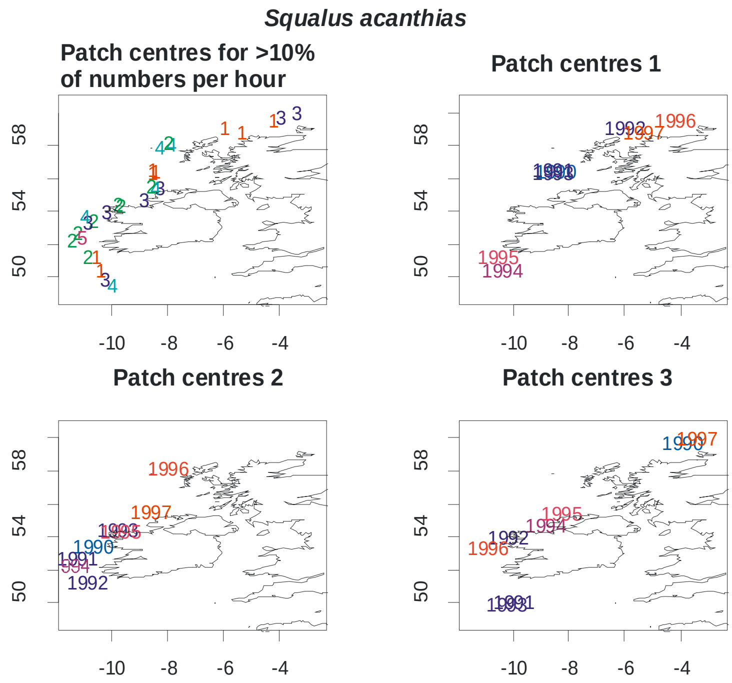


Figure 3.5-16: Location of first three most abundant spatial patches for Z. faber in the Bay of Biscay and Celtic Sea.

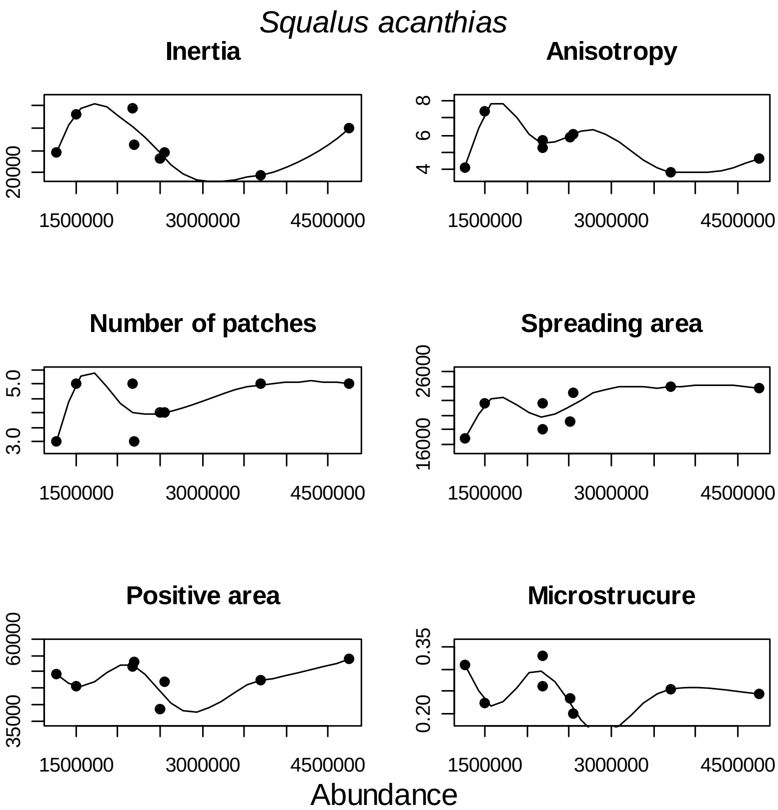


Figure 3.5-17: Spatial relationships (y axis) with abundance for spurdog.

3.5.7 Wolffish

Three species of wolffish that occur within the survey area (striped, northern and spotted) were located mainly in the western portion of the surveyed portion of the Barents Sea (Figures 3.5-16, 3.5-19, 3.5-20). The distribution of northern and spotted wolffish was very similar, occupying 20 and 22% of the area and only in the Barents Sea. The highest concentrations of both species occurred north and central parts of the Barents Sea survey area. Wolffish were largely absent from the eastern part of the survey area in the Barents Sea. Total area occupied by northern and spotted wolffish was fairly constant between 1978 and 1996 than declined thereafter (Figure 3.5-19).

The greatest decline occurred during the most recent period. Area occupied at the centre of mass for spotted wolffish was relatively constant over the entire period. However, for northern wolffish, extent of high density concentrations were small, <2%, until 2002, when the value increased to 6%. Whether this was the result of an increase in abundance or was due to immigration from surrounding, unsurveyed areas is unclear. A similar increase in density at centre of mass was noted for spotted wolffish. All three wolffish species became highly aggregated in the final period observed (Figure 3.5-18).

Striped wolffish had a wider distribution occupying the eastern part of the North Sea and along western Norway, as well as in the Barents Sea. It was most highly concentrated along the coast in contrast to the other two species (Figure 3.5-18). The total area occupied declined between 1982 and the most recent period while area occupied at the centre of mass was relatively constant during that period.

The centre of gravity for striped wolffish in the Barents Sea was variable between years (Figure 3.5-19). The main inertia axis was nearly always orientated east-west. Estimated abundance increased at the beginning and at the end of the time series (Figure 3.5-19 – note the figure shows log-abundance). The location of the three most abundant patches varied strongly between years with no clear pattern (Figure 3.5-20). Abundance (on log-scale) was positively related to inertia, spreading area and positive area (Figure 3.5-21). Thus higher abundances resulted in a larger area being occupied but also more spatial variability.

The ratio of area occupied at the centre of mass/total area occupied for all three species of wolffish and particularly for northern and spotted increased sharply during the last period observed (Figure 3.5-6). This was due to a high degree of concentration of the fish in the Barents Sea while the overall area occupied was reduced.

A comparison of the distribution of the wolffish in the northeast (this study) and northwest Atlantic (Kulka *et al*., 2004) showed similarities in the manner in which the three species of wolffish were distributed (compare Figures 3.5-18, 3.5-22, 3.5-23 and 3.5-24). Striped wolffish extended into more southerly areas (onto the southern Grand Banks in the northwest and North Sea in the northeast) while the other two species were distributed exclusively to the north (Labrador Shelf in the northwest, Barents Sea in the northeast). Kulka *et al*. (2004) noted that the wolffish species in the northwest Atlantic are temperature keepers, their distribution occurring over a narrow range of bottom temperatures.

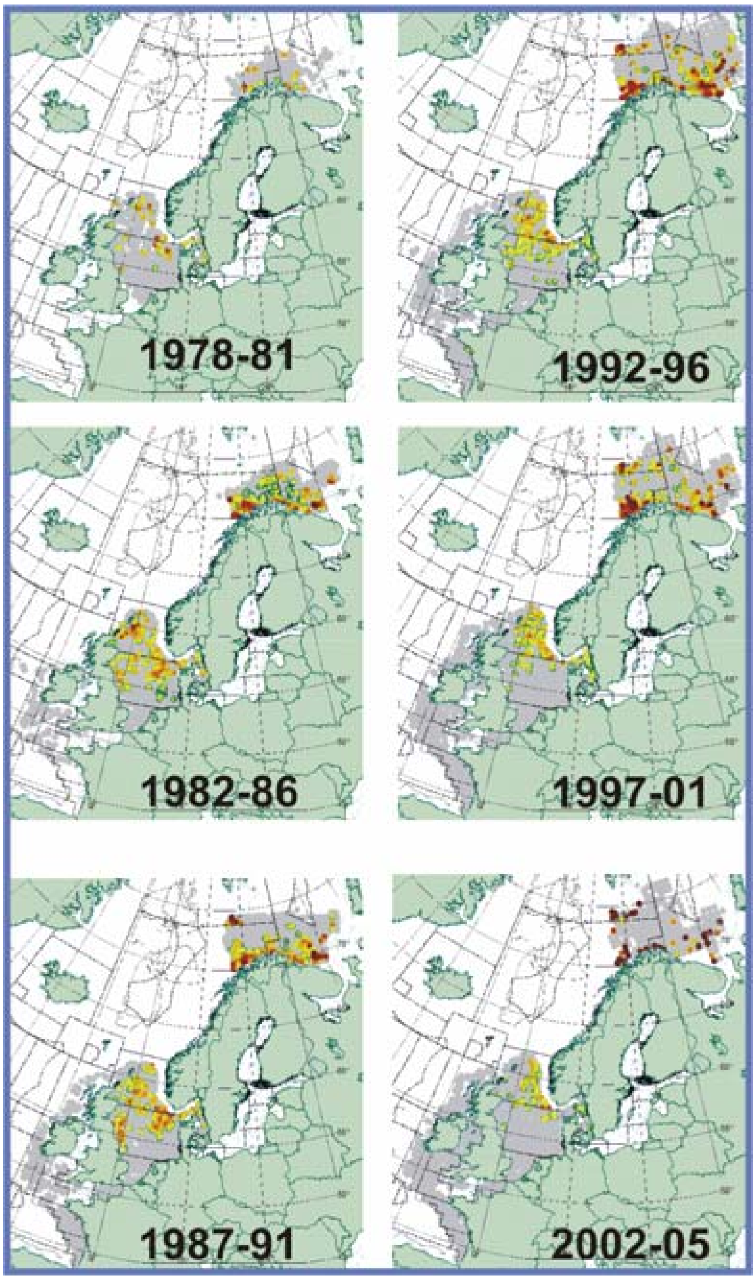


Figure 3.5-18: Distribution of A. lupus during six periods using the combined demersal survey (refer to Figure 3.5-1).

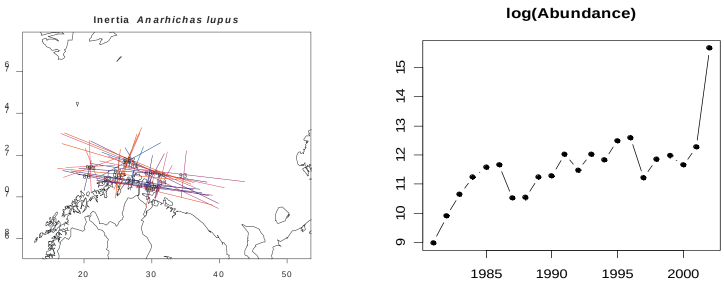


Figure 3.5-19: Change in inertia and centre of gravity of striped wolffish west off Ireland and Scotland and estimated abundance time series.

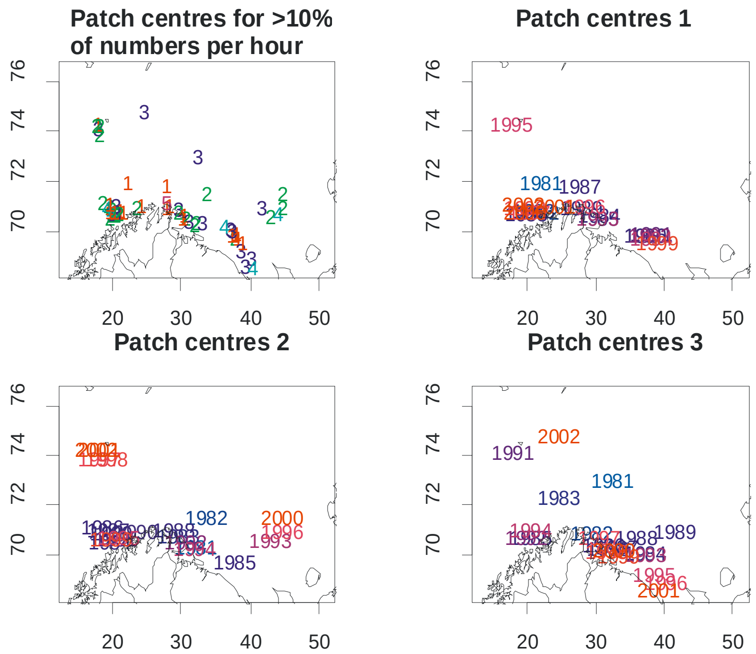


Figure 3.5-20: Location of first three most abundant spatial patches for *A. lupus* in the Barents Sea.

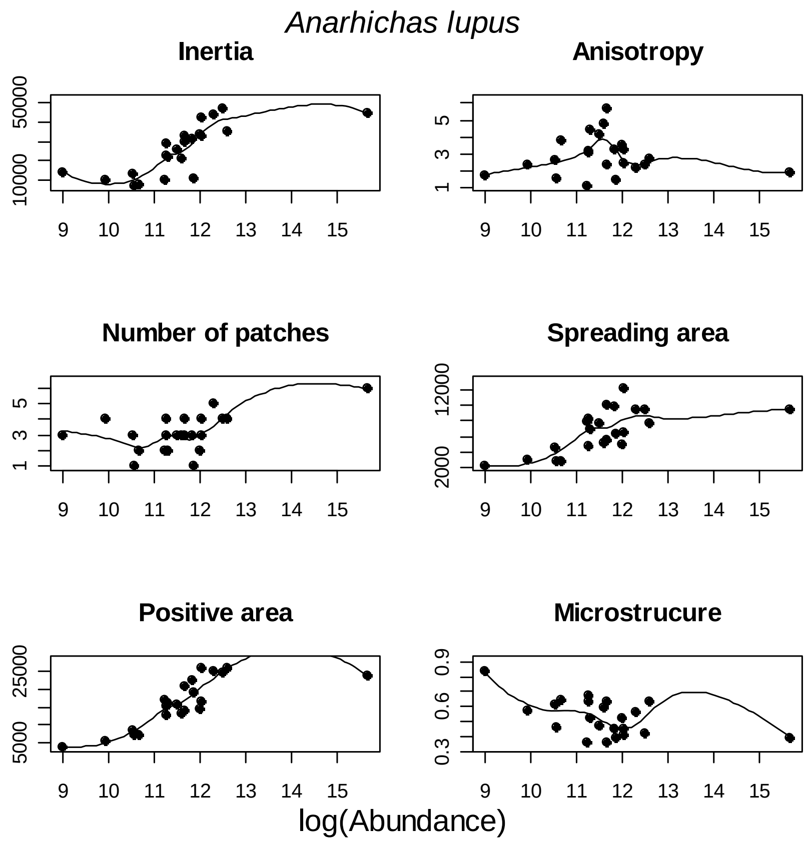


Figure 3.5-21: Spatial relationships (y axis) with abundance for A. *lupus*.

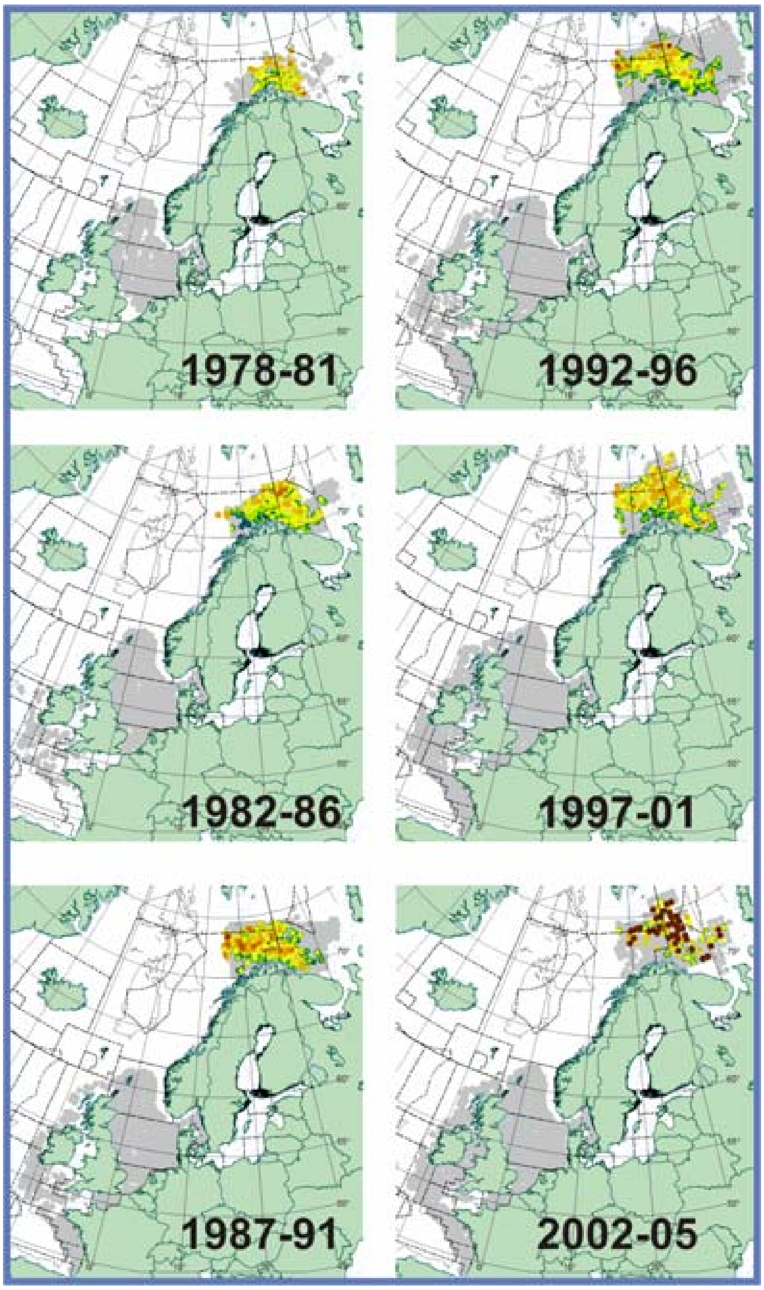


Figure 3.5-22: Distribution of *Anarhichas denticulatus* during six periods using the combined demersal survey (refer to Figure 3.5-1).

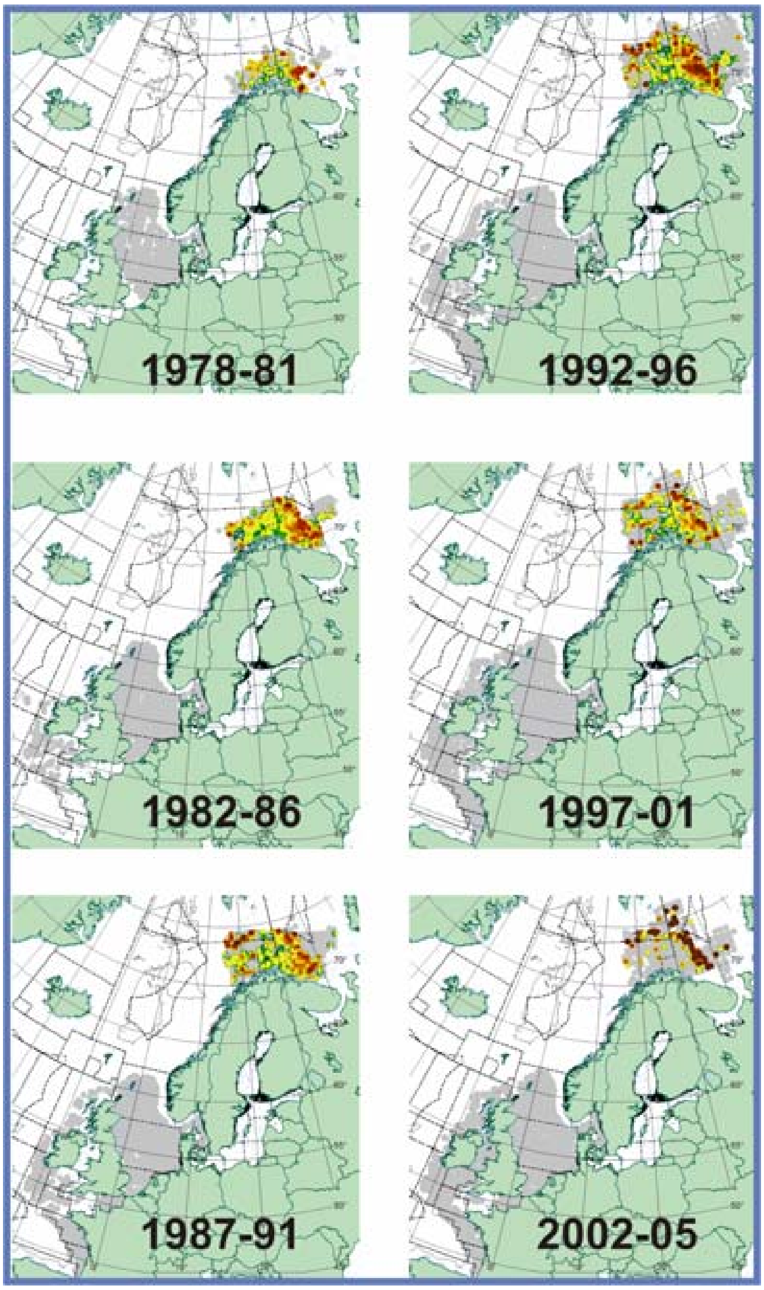


Figure 3.5-23: Distribution of *Anarhichas minor* during six periods using the combined demersal survey (refer to Figure 3.5-1).

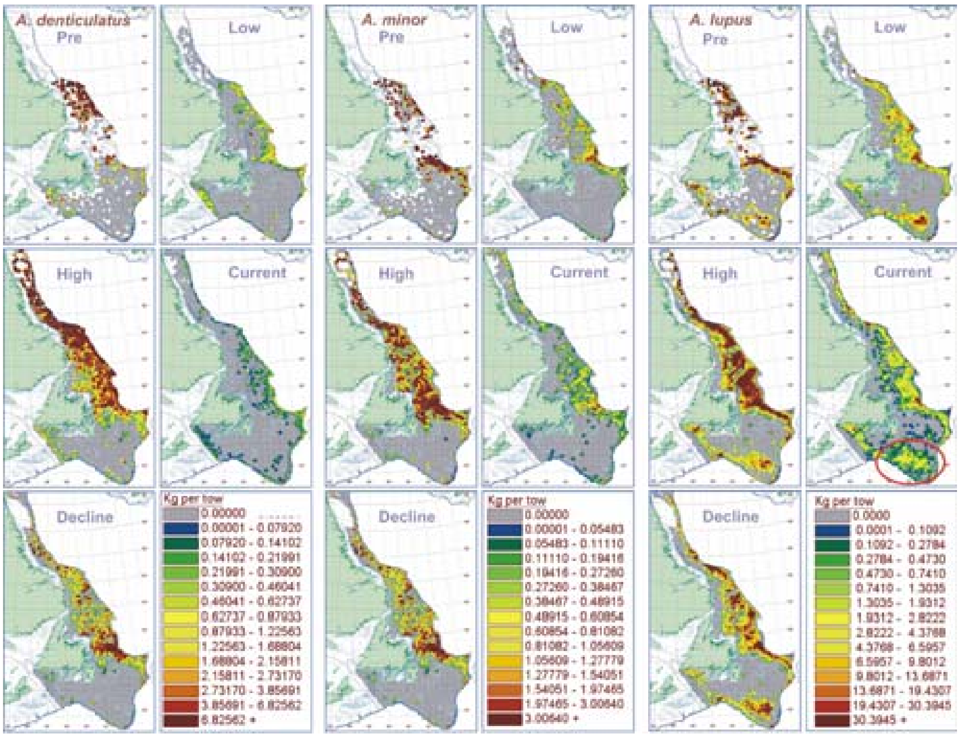


Figure 3.5-24: Distribution of the three wolffish species in the northwest Atlantic (after Kulka *et al.* 2004).

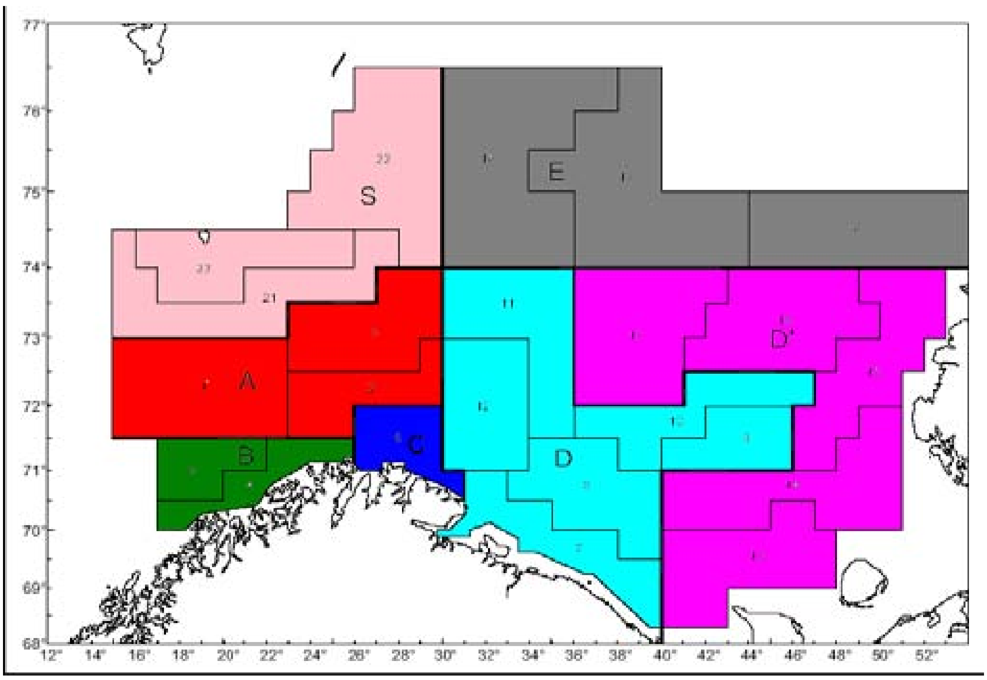
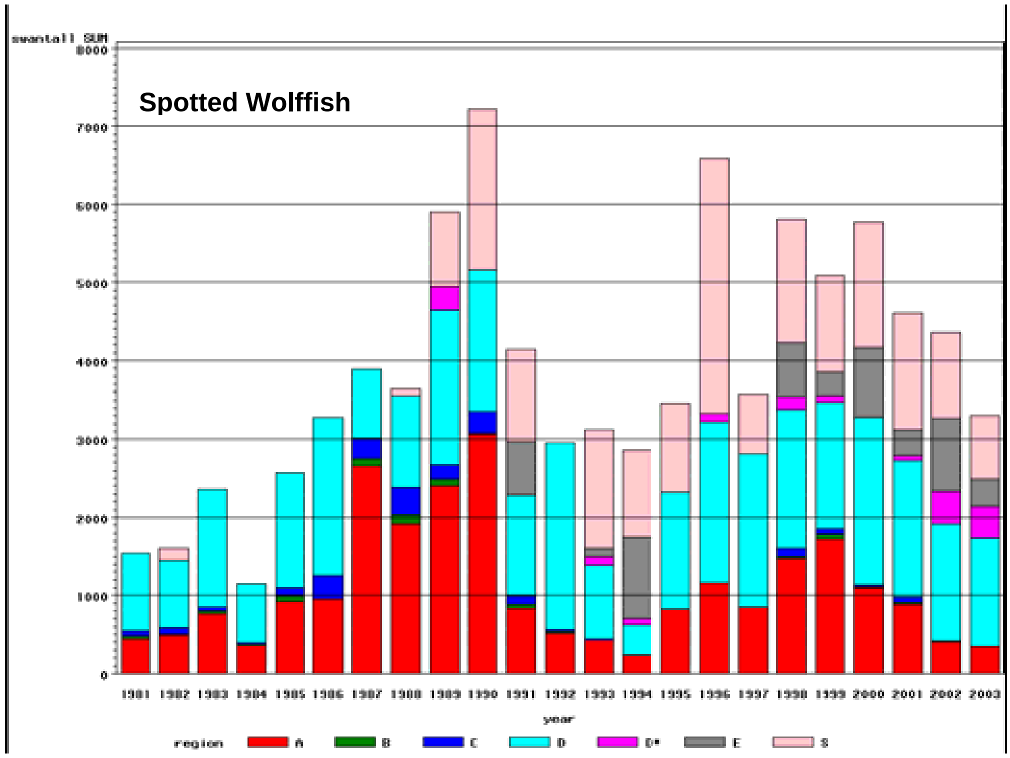


Figure 3.5-25: Norwegian trawl winter survey areas. Letters and colours correspond the bar segments in Figure 3.5-26.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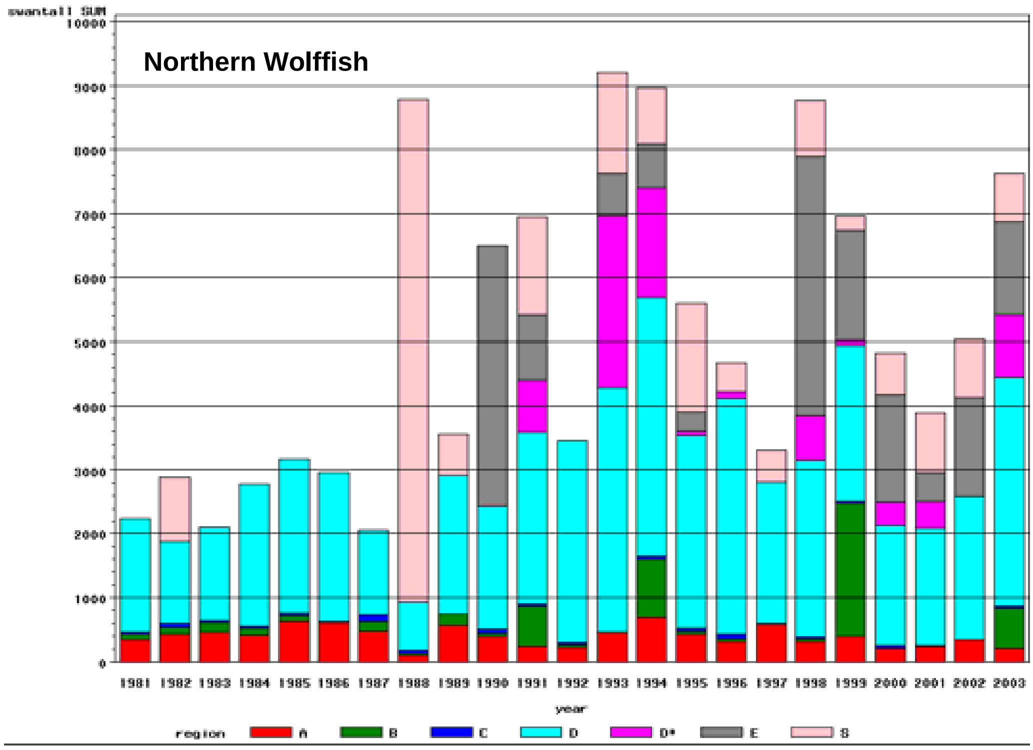
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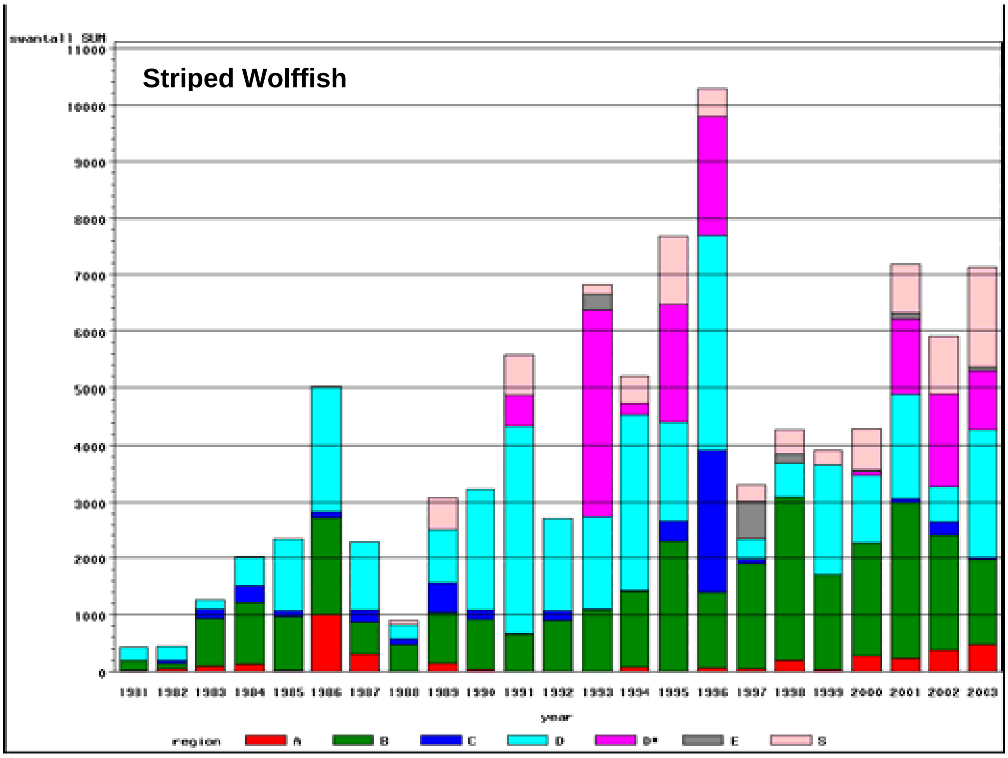

Figure 3.5-26: Swept area estimates (number caught) from IMR Barents Sea winter survey (after Wenneck 2005).

Figure 3.5-26 shows swept area estimates of the three wolfish species in the Barents Sea 1981–2003. The data is taken form a survey targeting cod. Therefore, for species other than cod, it is considered that the main areas of the Norwegian trawl survey areas ABCD give a more consistent time series than the full survey (Figure 3.5-25), and only areas A-D (red, green, blue and turquoise part of the column) should be compared across years. Abundance for the three wolfish species appeared stable, though there may have been a gradual decline in spotted wolffish over the last five years.

3.5.8 Summary

The above analyses shows that the six selected species had different distributions and underwent substantially different spatial changes over the period examined. Herring was most widespread occupying all of the survey area except the warmest areas (e.g. Bay of Biscay). Although area occupied changed little, it underwent an increase area of high concentration increased from 1978-1991, corresponding with an increase in abundance. John Dory, initially only distributed lightly along the western part of the survey area has increasingly occupied areas to the east including the northern North Sea and has become more concentrated along the Atlantic seaboard. The area occupied by spurdog has been relatively consistent over time. However, it has become increasingly less concentrated in the western part of the range. The distribution of two wolffish species, spotted and northern is constrained to the Barents Sea. Striped wolffish also occurs off western Norway and into the northern half of the Barents Sea. The most significant change in the distribution of all three wolffish species was an increase on their concentration in the Barents Sea, sharply so in the last period of observation. This increase in constituted a reduction in area occupied as well increased density. A comparison of distribution of the wolffishes in the Barents Sea in relation to bottom temperature indicates that they have a similar range of preferred temperatures. This observation is consistent with Kulka *et al*. (2004) who noted that wolffish species are temperature keepers.

Positive area and spreading area were positively related to abundance in the case of both wolffish and John Dory, and spreading was correlated with abundance of spurdog. None of the spatial indices showed any clear relationship with herring abundance. Thus wolffish, John Dory and spurdog populations were more spread out in space in years of higher overall abundance. These three species showed increasing trends in the areas and time periods studied, while herring fluctuated randomly, which might explain why no relationship was found for herring. For example, if there is some time lack between the spatial distribution of a species and the abundance at the time of the survey, it would be easier to detect a relationship between the spatial index and abundance in the case of continuously increasing or decreasing abundance time trends compared to a randomly fluctuating population.

3.6 Conclusions

The range and extent of distribution is often the only information reported for populations of fish. However, the spatial structure of fish populations is complex and changes in this structure are usually related to changes in abundance, as shown in a number of earlier studies (see Table 3.5-1). Perhaps more important indicators of change than increasing or decreasing in extent (area occupied) are the spatial elements within such as degree of aggregation, centre of gravity, inertia, anisotropy, count of spatial patches, positive area, spreading area, equivalent area and microstructure, that may occur within the perimeter of the overall population (or stock) area. Changes in these spatial characteristics relate to changes in the population, and so can be usefully applied to examine temporal changes in fish demography in relation to, for example, fisheries exploitation and environmental change.

Recommendation for future work on abundance-occupancy relationships could be to examine the patterns for selected species between stocks, e.g. Barents Sea stocks versus North Sea stocks; and between large areas selected NAFO stock versus selected ICES stocks. Studies of this nature in conjunction with knowledge of the fishery, the ecological and the environmental conditions in each area could provide insights into how fish aggregate on small and large scales in response to external conditions providing insights into the mechanisms of area-occupancy relationships. The mapping and geostatistical techniques developed here could be used for this work.

Application of this work in a management context could be aimed at trying to assess vulnerability of different species and populations to fishing effort (q – catchability) in the event of declining population size. That is, the likelihood of an acceleration in the decline as the population decreases in size. This would have implications for development of harvest control rules for populations approaching critical biomass levels (B_{lim}).

3.7 References

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4 International Bottom Trawl Survey

ToR g) liaise with IBTS to continue studies on the broadscale spatial and temporal patterns in selected fish species and communities along the European continental shelf of the eastern North Atlantic (e.g. the area covered by parts of ICES divisions VI-IX). Cross cut with ACFM groups and WGRED, SGRESP. Liaise with WGEF on identification and quantification of rare shark species.

4.1 Introduction

The International Bottom Trawl Working Group (IBTSWG) has its origin in the North Sea, the Skagerrak and the Kattegat where coordinated surveys have occurred since 1965.

The IBTSWG assumed responsibility for coordinating western and southern division surveys in 1994.

The ultimate goal should preferably be to combine all Eastern Atlantic survey data, in order to cover the spatial distribution of marine species in the entire area. However, up to this point in time, emphasis is put upon from the North Sea southwards and westwards (Table 4.1-1, Figure 4.1-1). It is of primary importance to address any bottlenecks that presently hinder the combination of the data.

Table 4.1-1: Overview of surveys presently covered by the IBTS Working Group.

Country	Survey	Q	gear	Design	Since	DATRAS
Den-Eng-Fra-Ger-Sco-Net-Nor-Swe	North Sea IBTS	1;3	GOV	by ICES rectangle	1965	Y
Scotland	West of Scotland (Rockall) - Deep Water Survey	3	GOV BT184	by ICES rectangle	1985	N
Scotland	Western Division Bottom Trawl Survey	1	GOV	by ICES rectangle	1981	Y
Scotland	Scottish Mackerel Recruit (SMR)	4	GOV	by ICES rectangle	1985	N
Ireland	West coast Groundfish Survey	4	rockhopper	by ICES rectangle	1990	N
Ireland	Irish Sea-Celtic Sea Groundfish Surveys	4	GOV	by ICES rectangle	1997	N
Northern Ireland	Irish Sea	1;4	rockhopper	stratified by depth and seabed-type with fixed stations	1992	N
England	Celtic Sea and Western Approaches Groundfish Survey	1	PHHT	fixed by area and depth strata	1981	N
England	Irish Sea and Celtic Sea	4	GOV	Fixed stations in strata	2004	N
France	Celtic Sea and Bay of Biscay Groundfish Survey	4	GOV	stratified random by area and depth	1987	Y
Spain	Porcupine	3	BACA	random stratified	2001	N
Spain	North Coast	4	BACA	stratified random by area and depth	1980	N
Spain	Gulf of Cadiz	2;4	BACA	stratified random by area and depth	1993	N
Portugal	Groundfish Survey	3;4	NCT	Fixed	1979	N

4.2 Conclusion

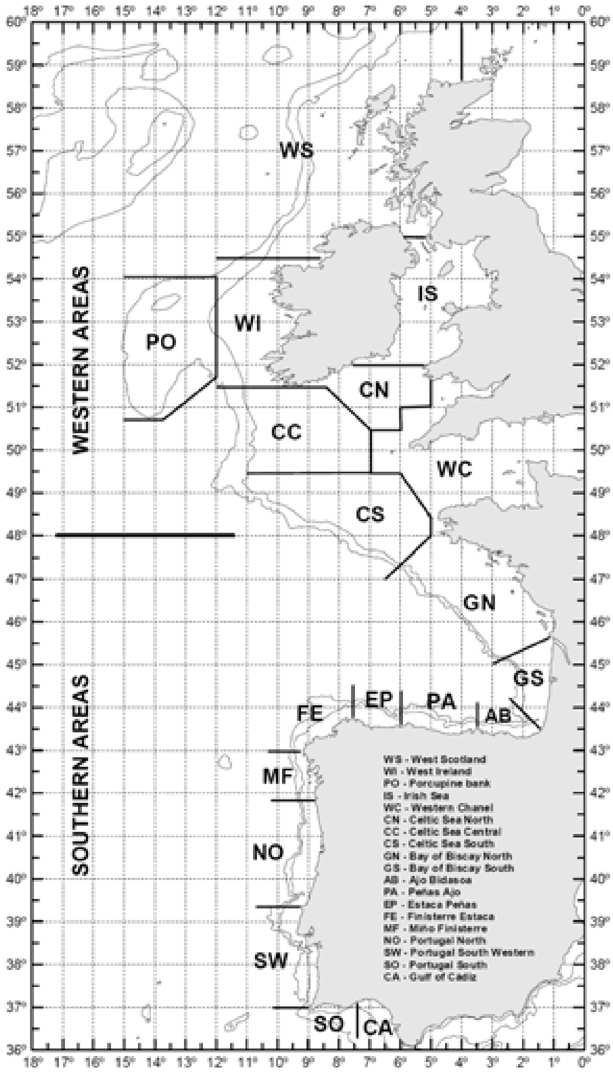
Each of the IBTS component surveys was designed for a different purpose, takes place in different season and with different gears, including several forms of GOV trawl. Owing to these differences and given the lack of suitable conversion factors, standardized catch rates are not available at the current time, and the datasets cannot yet be aggregated for quantitative analyses. However, combining the datasets in DATRAS should allow a variety of analyses, because there are other utilities that don't require standardized catch rates. For example, examining the broad-scale biogeographical patterns of fishes in the various surveys may provide useful information on presence/absence or for comparative species richness studies. Examples are given in Sections 3 and 4. For these studies, several datasets have been combined: the North Sea International Bottom Trawl Survey, the Scottish Western Division Bottom Trawl Survey, the historic English Celtic Sea and Western Approaches Groundfish Survey, the English Irish Sea and Celtic Sea Groundfish Survey, the French Celtic Sea and Bay of Biscay Groundfish Survey, the French Eastern Channel Groundfish Survey, and the Norwegian Barents Sea Groundfish Survey.

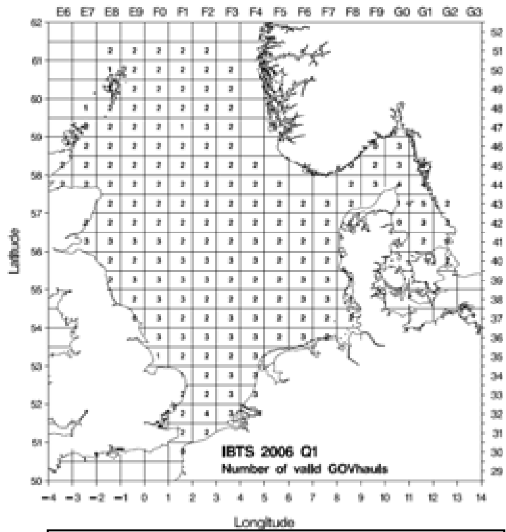
The inclusion of individual surveys in the IBTS database has only been possible because the institutes owning the data were 'represented' by members of WGFE, or because the data have already incorporated in a joint set, in this case the ICES-DATRAS system; however, data from the Irish, Northern Irish and Iberian surveys could not be made available. A system that combines several datasets like ICES-DATRAS has proven its value in two ways: first, by enabling the use of data collected by countries that were not represented at the meeting (Denmark, Sweden, and Russia), second, by facilitating their use because all data are presented in the same format.

For those surveys for which data were brought directly to the meeting through members, the integration of the different sets required considerable effort and this hampered the amount of work that could be carried out. However, some valuable results could be produced on selected topics, including information from areas as remote as the Barents Sea.

To facilitate future large-scale studies on changes in fish communities of the Northeast Atlantic, WGFE recommends:

- DATRAS is extended to allow incorporation of all information provided on other surveys than the ones currently included;
- All countries carrying out bottom trawl surveys should be invited to submit their data to DATRAS;
- IBTSWG develops further initiatives to implement these changes.





NB: Other bottom trawl surveys in the Eastern Atlantic area:

- France carries out a national groundfish survey in the Eastern Channel with a small GOV (quarter 4), which is not considered an IBTS-survey, but could fill the gap between the North Sea and the Celtic Sea.

- Norway has groundfish surveys (also non-IBTS) along its coast and in the Barents Sea, which extend the area covered by bottom trawl surveys in a northerly direction.

- The Baltic Sea is covered by the Baltic International Trawl Survey (BITS) and its data are already incorporated in DATRAS, although their use is presently restricted.

Figure 4.1-1: Overview of all IBTS-surveys, left: Western and Southern areas, right North Sea (example of 2006-Q1).

4.3 References

ICES. 2002. Manual for the international bottom trawl surveys in the western and southern areas. ICES CM 2002/D:03.

ICES. 2005. ICES IBTS Manual:

http://www.ices.dk/datacentre/datras/NSIBTSmanualRevVIIdraft.pdf

5 Essential fish habitat

5.1 Background

The concept of essential fish habitat (EFH) was introduced in the mid 1990s in the USA. In 1996, the USA Congress added habitat conservation measures to the Magnuson-Stevens Fishery Conservation and Management Act, which states “One of the greatest long-term threats to the viability of commercial and recreational fisheries is the continuing loss of marine, estuarine, and other aquatic habitats. Habitat considerations should receive increased attention for the conservation and management of fishery resources of the United States” (16 U.S.C. 1801 (A)(9)).

The USA Congress defined EFH as “those waters and substrate necessary to fish for spawning, breeding, feeding, or growth to maturity”, where **waters** are “aquatic areas and their associated physical, chemical, and biological properties that are used by fish and may include aquatic areas historically used by fish where appropriate”; **substrate** “includes sediment, hard bottom, structures underlying the waters, and associated biological communities”; **necessary** means “the habitat required to support a sustainable fishery and the managed species’ contribution to a healthy ecosystem”; and “spawning, breeding, feeding, or growth to maturity” covers a species’ full life cycle.

In 2003, WGFE was asked to: “Review the concept of essential fish habitat and consider what specify essential fish habitat for individual species or stocks”. WGFE (ICES, 2003a) reviewed and evaluated the concept and stated 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.”

In following years, ToRS regarding fish habitat concentrated on the study of habitat requirements of “commercial, threatened or rare species” and did so by focusing on some gadiform (cod-like) and pleuronectiform (flatfish) fishes (ICES, 2004) and selected deep-water species, examing data from Le Danois Bank, the Barents Sea, and Grand Bank and Labrador Shelf (ICES, 2005b).

5.2 Summary of habitat work undertaken by other ICES Working Groups

The Working Group Marine Habitat Mapping (WGMHM) (ICES, 2003b) reviewed ongoing and future marine habitat mapping proposals for the North Sea and the OSPAR area**.** The OSPAR Biodiversity Committee (BDC), as part of its work to implement Annex V of the Convention and the Biodiversity Strategy, has identified a need to prepare maps of seabed habitats. This is to meet both specific and immediate needs in relation to the protection of threatened habitats, the development of EcoQOs and longer-term goals regarding quality status assessments and implementation of an ecosystem-based approach to management of the marine environment. Such habitat mapping is a natural progression of the ongoing work of the Committee to develop a habitat classification system for the OSPAR area, in conjunction with the European Environment Agency (EEA) and ICES. Annex 4 of the ICES (2003*b*) listed the major habitat mapping programmes within the OSPAR area.

5.3 Benthos – Fish interactions

5.3.1 Benthos – Fish sampling programme

The GSBTS (German Small-scale Bottom Trawl Survey) includes coupled analyses of demersal fish fauna, epibenthos and hydrography in order to contribute to the physical and biological characterization of habitats of demersal North Sea fish. Within the GSBTS survey, which was initiated in 1987, six areas of the North Sea have been selected for these coupled analyses conducted yearly since 1999 to study the spatial and functional coupling between demersal fish and invertebrate epibenthic fauna (Figure 5.3-1; Ehrich 1988;Callaway *et al*., 2002, Hinz *et al*., 2005). The demersal fish fauna is sampled with the same gear as used in the IBTS Survey (GOV, 30-min hauls), while the GSBTS focuses on small-scale analyses to complement the large-scale international survey. Spatial resolution: typically between 20 and 30 hauls within the six 10x10 nautical mile squares, each sampled within three consecutive days. Epibenthos and sediment are sampled in tight spatial and temporal coupling to the GOV hauls, using a 2-m beam trawl and a van-Veen grab, respectively. Accompanying depth profiles of hydrographic parameters are taken. Methods and results of the survey will be presented in a review before the 2007 meeting of WGFE (Ehrich *et al*., in prep.) and should then be analysed in the framework of essential habitat description.

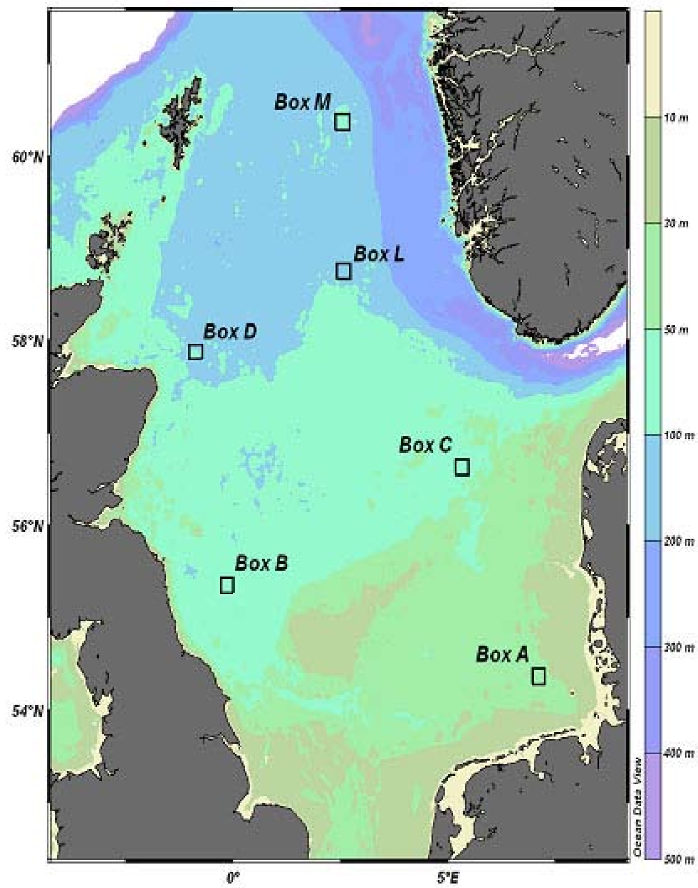


Figure 5.3-1: Location of sampling areas ("boxes") within the GSBTS, surveyed in coupled analyses of assemblages of epibenthos and fish.

5.3.2 Benthos–Fish trophic interactions

Fish–benthos relationships have special relevance in feeding habits, and the level of association between these components of the ecosystem will be marked by grade of feeding specializations of each fish species. It is well-known that flatfishes (*Pleuronectidae*), ventrally blind, except the Greenland halibut (*Reinhardtius hippoglossoides*), are strongly associated with the bottom relative to most fish species; therefore, changes or disturbances in the benthic habitat may influence their diet composition and condition of these species which could be reflected in their reproductive potential.

In the NAFO area, witch (*Glyptocephalus cynoglossus*) is the most specialized species, with a diet that consists almost exclusively of benthic prey (Román *et al.*, 2004). There also is a clear dependence on benthos in the cases of American plaice (*Hippoglossoides platessoides*) and yellowtail flounder (*Limanda ferruginea*). Other demersal species, as cod (*Gadus morhua*), wolffishes (*Anarhichas* sp.) and skates (*Rajidae*) have a more diverse diet (Rodríguez-Marín *et al*., 1994; Rodríguez-Marín, 1995; Román *et al*., 2004) and therefore may be less affected by changes in the benthos.

The data and some results on feeding habits are presented from the Spanish Bottom Trawl Research Survey "*Patuxa*" in NAFO Area Divs. 3NO in the period from 2002-2005 (Table 5.3-1).

Table 5.3-1: Characteristics of the Spanish Bottom Trawl Research Survey "*Platuxa*" 2002-2005 (NAFO Area, Divs. 3NO).

DATA OF SPANISH RESEARCH SURVEY *PLATUXA*				
RV	YEAR	GEAR	DATE (DAY/MONTH)	DEPTH RANGE (M)
Vizconde de Eza	2002	*Campelen 1800*	29/04 to 19/05	38 - 1540
Vizconde de Eza	2003	*Campelen 1800*	11/05 to 02/06	38 - 1666
Vizconde de Eza	2004	*Campelen 1800*	06/06 to 24/06	43 - 1460
Vizconde de Eza	2005	*Campelen 1800*	10/06 to 29/06	47 - 1438

The stomach contents of main species in the catch were analyzed on board ship. The results obtained in the study of the diet composition of spotted wolffish (*Anarhichas minor*), Northern wolffish (*Anarhichas denticulatus*), Atlantic wolffish (*Anarhichas lupus*), witch, and yellowtail flounder are summarised below. Characteristics of the sampled individuals are shown in Tables 5.3-2 and 5.3-3, and show the results in weight (%) and number (%) of the main prey groups.

Table 5.3-2: Individuals sampled (NAFO, Div. 3NO, 2002–2005).

SPECIES	LENGTH (CM)		DEPTH RANGE (M)	NO. INDIVIDUALS SAMPLED				
	Min.	Max.		2002	2003	2004	2005	*Total*
Anarhichas denticulatus	20	111	57 - 1458	14	39	48	88	*189*
Anarhichas lupus	6	124	51 - 1337	189	166	150	285	*790*
Anarhichas minor	11	102	176 - 751	4	9	65	36	*114*
Glyptocephalus cynoglossus	6	61	43 - 1450	419	581	278	350	*1628*
Limanda ferruginea	5	60	38 - 156	645	777	527	536	*2485*

Table 5.3-3: Main prey group (% weight and number) by year.

SPECIE	PREY GROUP	WEIGHT (%)				NUMBER (%)			
		2002	2003	2004	2005	2002	2003	2004	2005
A. denticulatus	Pisces	90.2	23.1	11.3	13.5	9.1	0.5	1.7	0.3
	Crustacea	0.7	1.4	5.9	10.3	36.4	2.4	23.9	54.3
	Mollusca	2.2				4.5			
	Echinodermata		2.2	4.1	3.4		1.3	1.1	1.7
	Other invertebrates	6.1	69.4	77.6	71.6	40.9	95.4	72.2	43.5
	Others	0.8	3.9	1.2	1.3	9.1	0.3	1.1	0.3
A. lupus	Pisces	3.4	2.3	1.0	2.9	1.1	1.0	0.1	0.3
	Crustacea	2.5	23.3	11.9	14.3	9.5	12.9	13.3	8.5
	Mollusca	80.2	37.2	39.4	34.9	62.4	19.6	33.4	59.0
	Echinodermata	0.2	3.0	27.4	44.4	5.3	8.6	28.7	28.4
	Other invertebrates	10.9	34.1	20.1	3.1	19.0	57.7	23.8	1.8
	Other/not identified	2.9	0.0	0.1	0.4	2.7	0.2	0.7	2.0
A. minor	Pisces			24.2	34.7			1.9	0.4
	Crustacea	4.0	62.5	7.6	4.7	4.5	22.7	8.6	6.7
	Mollusca			1.0	0.1			31.9	4.0

Specie	Prey group	Weight (%)				Number (%)			
		2002	2003	2004	2005	2002	2003	2004	2005
	Echinodermata	96.0	32.2	67.0	60.5	95.5	68.2	56.7	87.9
	Other invertebrates		5.3	0.1	0.1		9.1	0.8	0.9
G. cynoglossus	Pisces		1.4	5.0	5.9		0.2	0.3	0.3
	Crustacea	15.3	3.7	15.3	10.2	32.5	21.7	38.6	32.7
	Mollusca	2.1	0.4	0.5	1.0	8.4	3.3	3.9	6.9
	Echinodermata	0.0	0.1	0.0	11.6	0.1	0.5	0.0	0.7
	Other Invertebrates	80.6	93.2	79.1	70.8	58.2	72.3	56.8	57.0
	Other/not identified	2.0	1.2	0.1	0.6	0.8	1.9	0.4	2.4
L. ferruginea	Pisces	49.2	30.0	37.6	56.9	1.5	0.8	0.8	1.4
	Crustacea	24.6	28.2	32.3	24.2	67.1	62.1	68.9	91.3
	Mollusca	2.3	2.7	3.1	0.8	3.4	1.2	12.1	1.9
	Echinodermata	1.8	1.2	1.6	0.7	0.8	0.6	0.6	0.2
	Other invertebrates	20.3	37.2	24.1	16.3	27.1	35.1	17.4	5.0
	Other/not identified	1.8	0.6	1.3	1.0	0.2	0.1	0.2	0.3

Analysis of the fish diet shows that invertebrates constitute an important diet component of some fish species, with most of the invertebrates fed upon being benthic. Benthic invertebrates reach 90% in weight and number in the stomach contents of some cases. Changes in the prey composition in wolffish diet have been observed within recent years. For example, the prey of *A. denticulatus* has shifted to a lower trophic level, with fish being the major prey in 2002, and molluscs and other invertebrates thereafter. Dominant prey in *A. lupus* has changed to molluscs and echinoderms, while *A. minor* became more piscivorous. It would be necessary to conduct a more extended quantitative analysis to establish the importance of these changes.

In the cases of species more specialized, or less opportunistic, detailed studies of the diet changes in the medium term may indicate changes in the ecosystem, particularly changes in the abundance of certain benthic organisms. On the contrary, a more constant diet in selective predators may indicate stability in the benthic ecosystem. The medium-term studies of the diet of witch, which seems quite stable, could be a good indicator to detect changes in the benthic habitat in the area NAFO.

In the life cycle of most fish species, the association with the benthic habitat is more marked in the first years of the life, when the diet is composed almost exclusively of benthic invertebrates. For this reason, the feeding studies focusing on the first years of life of demersal species can be very illustrative of the benthic association (and habitat), while in the adult stage, its relationship with the benthos become weaker.

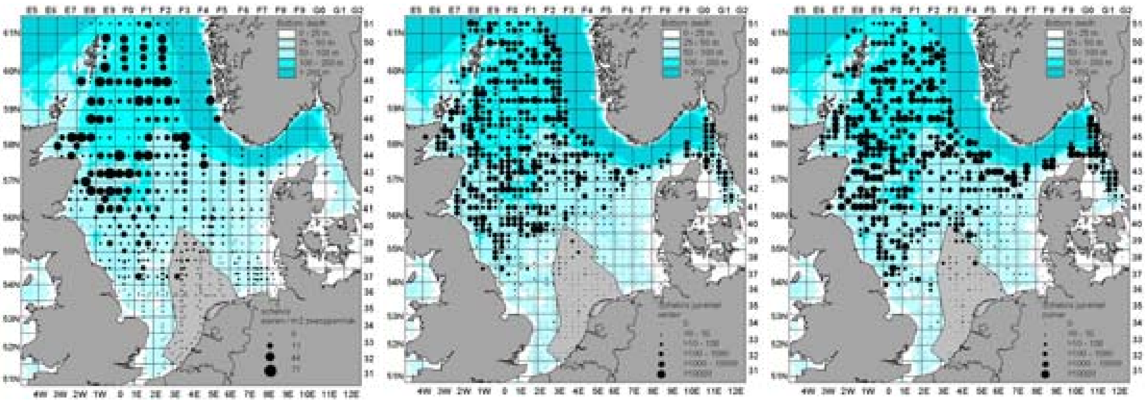
Studies of feeding behaviour should not be limited to commercially important species but should also focus on species that give the highest level of information on benthic habitats and can best serve as indicators of changes in benthos communities and trophic relationships.

5.4 Nursery areas of North Sea fishes

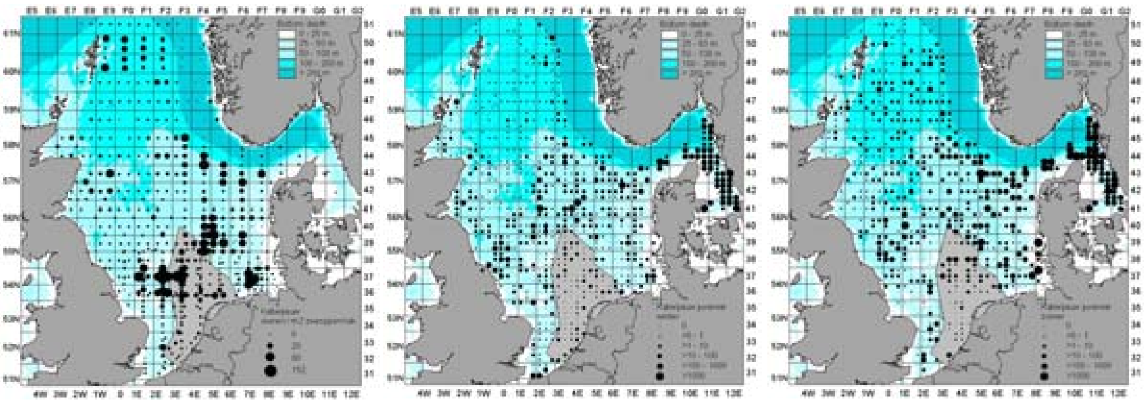
This Section describes the spatial use of the North Sea for spawning and nursery grounds for 7 important fish species (herring, whiting, haddock, cod, mackerel, plaice and sole). Information on spawning can to some extent be estimated from the catches of mature adults but more precise data can be provided from egg surveys. This is because spawning may occur in regions that are not accessible to fishing gear and unlike adult fish; eggs do not actively avoid sampling gears. For all species examined here, the location of the spawning areas can be determined by the distribution of the youngest development stages (stage 1). The spawning areas are determined based on several egg-surveys: PLACES (Plaice and Cod Egg Survey)

(Fox *et al.*, 2005), the Herring Larval surveys (ICES, 2005a), the Mackerel Egg survey (ICES, 2005c), and the 1989 Egg Surveys in the southeastern North Sea (van der Land, 1991).

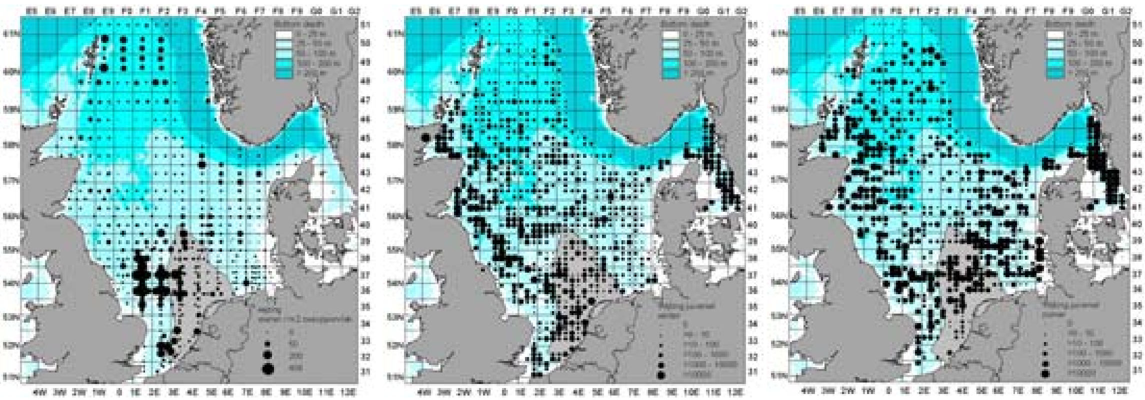
The location of the nursery areas are visualized by plotting the average distribution (1995-2004) of 0 and 1 year olds, on a scale of 1/9th ICES-rectangle (ter Hofstede *et al.*, 2005) (Figure 5.4-1). A distinction was made between the winter (quarter 1) and summer (quarter 3) distribution. The data originates from the International Bottom Trawl Survey (IBTS) for roundfish and the Dutch Beam Trawl Survey (BTS) for flatfish.



A) Haddock eggs B) Haddock juveniles winter C) Haddock juveniles summer

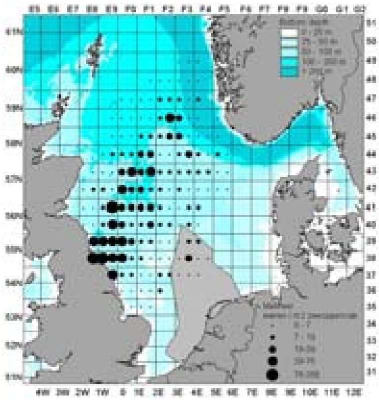


A) Cod eggs B) Cod juveniles winter C) Cod juveniles summer

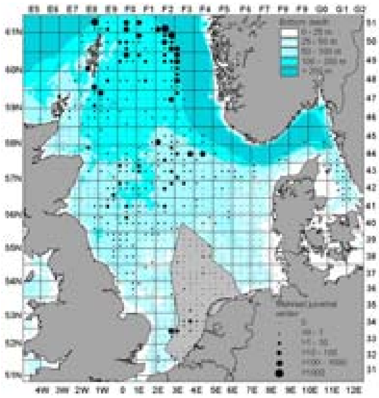


A) Whiting eggs B) Whiting juveniles winter C) Whiting juveniles summer

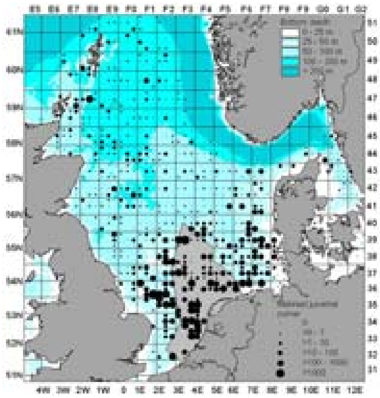
Figure 5.4-1: Distribution of eggs (A) and juveniles during winter (B) and summer (C). Data are derived from egg surveys and trawl surveys (see text). (Grey area: Dutch Exclusive Economic Zone.)



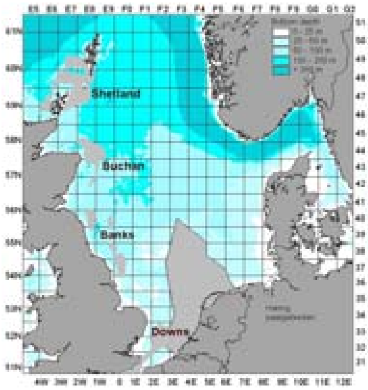
A) Mackerel eggs



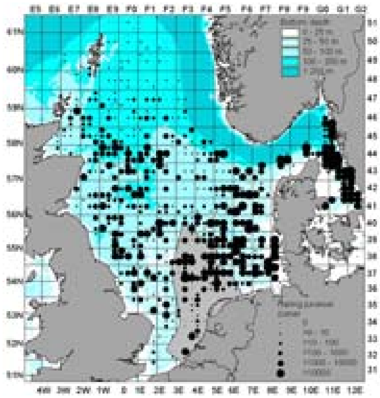
B) Mackerel juveniles winter



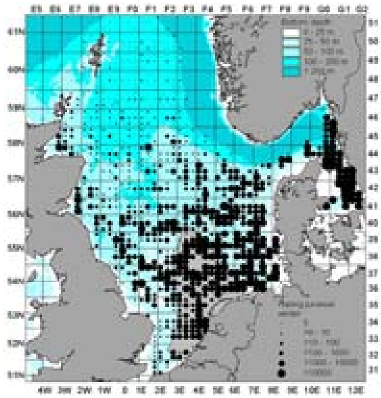
C) Mackerel juveniles summer



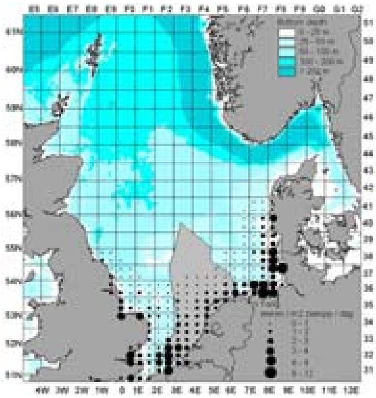
A) Herring eggs



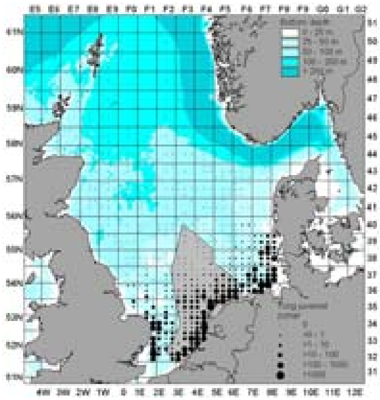
B) Herring juveniles winter



C) Herring juveniles summer

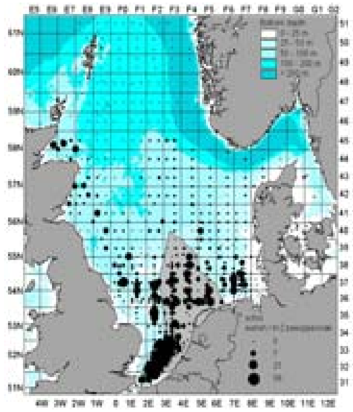


A) Sole eggs

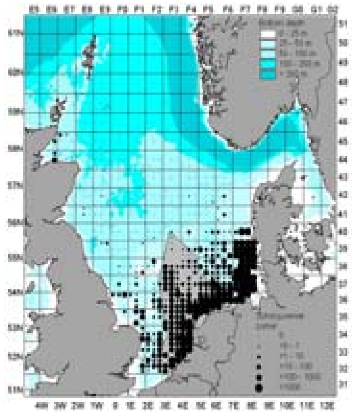


C) Sole juveniles summer

Figure 5.4-1 Continued: Distribution of eggs (A) and juveniles during winter (B) and summer (C). Data are derived from egg surveys and trawl surveys (see text). (Grey area: Dutch Exclusive Economic Zone.)



A) Plaice eggs



C) Plaice juveniles summer

Figure 5.4-1 Continued: Distribution of eggs (A) and juveniles during winter (B) and summer (C). Data are derived from egg surveys and trawl surveys (see text). (Grey area: Dutch Exclusive Economic Zone.)

As becomes clear from the maps, the various species show large differences in their spatial use of the North Sea during early life-history stages. This might indicate that the entire North Sea is of use for spawning and nursery by various fish communities. This should be further examined in the near future by generating more distribution maps illustrating the spatial distribution of key life-history stages for a wider range of species.

5.5 ICES-FishMap

ICES-FishMap (http://www.ices.dk/marineworld/ices-fishmap.asp) is an electronic atlas of 15 North Sea fish species that uses data collected during the North Sea IBTS in the period 1983-2004. It is the outcome of an EU-funded project under the same name, and was a cooperative exercise involving the Netherlands Institute for Fisheries Research (RIVO), the Centre for Environment, Fisheries and Aquaculture Science (CEFAS), and the Secretariat of the International Council for the Exploration of the Sea (ICES).

The advantage of an electronic atlas is that it allows an annual update and that it is flexible in selecting periods to allow changes in the fish fauna to be studied. The ICES-FishMap is considered to be a preliminary update of the 1993 Atlas of North Sea Fishes (Knijn *et al.*, 1993), and so far covers 15 species. The ultimate aim is to produce an electronic and paper atlas for a much larger area than the North Sea that provides information on all sampled fish species.

ICES-FishMap allows the creation of distribution maps (North Sea, Skagerrak and Kattegat) for the 15 fish species, by selecting on years, quarters, ages and size-classes. These data are derived from the DATRAS survey database kept at the ICES Secretariat and will be updated annually.

ICES-FishMap also offers a short summary of relevant information for each of the 15 species (basic pages), and a detailed section by species on the distribution, life history and exploitation (pdf files) (Figure 5.5-1). In addition, ICES-FishMap supplies information on the surveys used, the factors affecting the distribution, the fish communities, and the limitations of the data presented.

Fig 5.5-1: A sample of an ICES FishMap page for herring showing a map of spatial distribution and information on the biology of the species

5.6 Pupping grounds of spurdog, *Squalus acanthias*

A large catch of spurdog, *S. acanthias*, was made during the course of an annual groundfish survey in the Irish Sea, west of the Lleyn Pensinula, Wales (52.95ºN, 4.86ºW). The catch of spurdog comprised 5 tonnes (98.6%) of females and 71 kg (1.4%) of males. A random sub-sample of 210 females were measured, with 8 healthy specimens tagged and released, and the remaining specimens dissected for maturity and fecundity examination. The length range of the females examined was 82–117 cm and of the 202 dissected, one was mature but lacking candles and embryos, two contained candled embryos, 142 (70%) contained pups with yolk sacs (i.e. pups about one year old) and 57 (28%) contained term pups (Figure 5.6-1). Due to the relatively high proportion of gravid females with full-term pups, it is possible that this area is a pupping ground for spurdog.

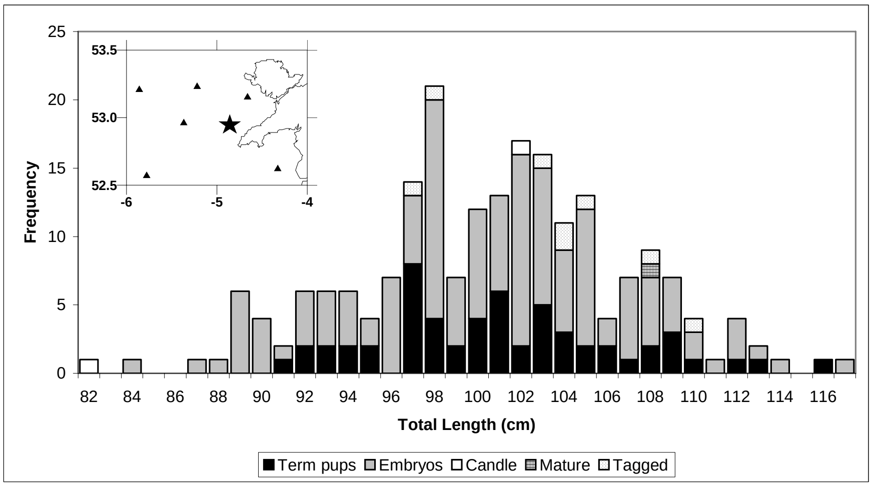


Figure 5.6-1: Length-frequency and maturity stages of female spurdog caught west of the Lleyn Peninsula (catch station denoted ★ on insert).

5.7 Conclusions and recommendations

Knowledge of EFH is important in interpreting many other work areas conducted within the group (e.g. abundance-occupancy, threatened fish species etc.). In terms of the future direction of WGFE in this work area**,** WGFE suggested a focus on the following areas:

- Study the functional coupling between fish and their biotic and abiotic environment to identify the characteristics of essential habitats for fish species (and life-history stages) of interest. Examine the distributions of demersal and pelagic fish in relation to habitat properties, and identify those ecological, physiological and behavioural components that may affect the distribution of fish.
- Review ongoing research activities in national and international programs that can help to isolate potential mechanisms that restrict fish to certain habitats, including habitat selection in single species and multi-species scenarios.
- Mapping of contemporary species distributions using broadscale data (e.g. see Section 3) to identify species of localised abundance and areas of importance to various life-history stages of commercial and vulnerable species.
- Liaise with relevant ICES working groups (e.g. WGMHM) in order to build upon activities in the areas of habitat mapping, with respect to both maps of abiotic habitat parameters and biotic components (e.g. benthic communities and biotopes). Such broadscale maps are fundamental to relating the distribution of fish to the distribution, structure and function of sea floor habitats, and for identifying important fish habitats.
- It is suggested that WGFE explore the utility of using IBTS and other national data to identify the broadscale distribution of nursery grounds of commercial and vulnerable fish species in the ICES area.

5.8 References

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6 Relative catchability of fishes

6.1 Introduction

Reliable estimates of biomass are required for ecosystem modelling and are an essential part of describing ecosystems (Harley *et al.*, 2001). For most species, biomass is estimated from scientific research surveys such as the International Bottom Trawl Survey (IBTS) in the North Sea. This is not an absolute estimate of biomass as species and size classes are sampled disproportionately by the survey gear. In order to obtain more accurate estimates of biomass the catchability of a species to the survey gear must be known.

6.2 Case study 1: Estimating bottom trawl catchability of several species by RV "Thalassa" in the French Groundfish survey

6.2.1 Introduction

Several models for estimating bottom trawl catchability were developed and tested on real and simulated data. The results have been published in Trenkel and Skaug (2005). Here an extract from that work is provided, including estimates of catchability for nine specieshake caught by the French groundfish survey in the Bay of Biscay and Celtic Sea (Evohe survey).

6.2.2 Models

We model the statistical distribution of catch data as a mixture of two processes, population abundance and random trawl efficiency, making the following assumptions 1) individual fish are randomly distributed in space (ignoring the vertical component) and do not form large schools; 2) trawl catchability is a random variable in the range (0.1); 3) the width of the swept area is the same for all hauls, but trawled distance can vary. A series of three models was developed. Model 1 is the baseline model, model 2 includes the effects of body size on trawl catchability and model 3 takes account of the relationship between age and population abundance.

6.2.2.1 Model 1

For the *i*th haul, denote by y_i the number of individuals in the catch conditional on n_i individuals present in the trawl path of which a proportion q_i is caught, where i=1, …m, and the haul efficiencies q_i are independent random variables. Modelling the number of individuals in the trawl path by a Poisson distribution, we have

$$n_i \sim \text{Poisson}(\lambda) \qquad n_i = 1,2,\ldots \qquad (1)$$

The capture proportions q_i are modelled by a normally distributed random variable p_i which a logistic transformation puts into the appropriate range $q_i \in (0,1)$.

$$p_i \sim \text{N}(\mu, \sigma^2) \qquad (2)$$

$$q_i = \exp(p_i)/(1+\exp(p_i)) \qquad (3)$$

The above formulation corresponds to a random effects model for catchability. The resulting model of catch numbers is

$$\text{C}_\text{i} \sim \text{Poisson}(q_i\, \lambda\, 2/\text{D}_\text{i}) \qquad i=1, \ldots m \qquad (4)$$

where D_i is the distance trawled by haul *i*, such that $2/\text{D}_\text{i}$ is a factor for standardising to the nominal trawl distance of 2 nm.

6.2.2.2 Model 2

Modelling the mean of the random capture variable as a linear function of the average body length (l_i) in hauls *i*, we get

$$\mu_i = a + b\, \mathrm{l}_\mathrm{i} \tag{7}$$

$$p_i \sim \mathrm{N}(\mu_i, \sigma^2) \tag{2a}$$

This means that the capture proportion q_i (eq. 3) is a logistic function of body length, which corresponds to a classical selectivity model. The model for population abundance remains as before (eq 1). We use average body length in a haul to represent the length effects of a given species.

6.2.2.3 Model 3

Modelling population abundance as a decreasing function of age, we obtain

$$\lambda_i = d\ \mathrm{exp}(-\ c\ \mathrm{age}) \tag{8}$$

$$n_i \sim \mathrm{Poisson}(\lambda_i) \tag{1a}$$

This model corresponds to a classical population dynamics model where *c* is total mortality and *d* average recruitment. The random effects catchability model remains as in model 1 (eqs. 2 and 3). Age can be estimated from body length assuming a growth function. Using the inverted von Bertalanffy growth model, we estimated mean age in a haul by first estimating the age of all individuals using growth parameters.

6.2.3 Data

For fitting the different models, catch data from repetitive fishing hauls carried out in the Celtic Sea in 1996 on board RV “Thalassa” (introduced in 1996) with the standard 36/47 GOV bottom survey trawl were used.

6.2.4 Results

Model estimates for models 1 and 2 are given in Table 6.2-1 for those species with enough individuals to allow model fitting. The relationship between mean capture proportion, μ (eq. 7) for model 2 shows an increasing catchability with mean length. The probability distributions for the capture proportions q (eq 3) for model 2 are also shown. Among the four species shown, megrim (*Lepidorhombus whiffigonis)* has the highest capture proportions while horse mackerel (*Trachurus trachurus*) has the lowest (Figure 6.2-1). This difference is easily explained by body size and but probably also body shape and behaviour. For further results see Trenkel and Skaug (2005).

Table 6.2-1: Model parameter estimates (standard deviations in brackets) for model 1 and 2 for selected species.

Species		**Model 1**				**Model 2**		
	Code	λ	μ	log(σ)	λ	a	b	Log(σ)
Argentina silana	ARGESIL	93 (14)	-1.4 (0.4)	0.65 (0.2)	109 (24)	5 (<0.1)	-0.41 (0.03)	0.5 (0.2)
Argentina sphaerana	ARGESPH	294 (20)	-0.62 (0.4)	0.7 (0.2)	331 (30)	-12.84 (7.0)	0.66 (0.4)	0.55 (0.2)
Arnolatus imperialis	ARNOIMP	27 (11)	-1.08 (0.7)	0.17 (0.3)	30 (13)	1.43 (1.8)	-0.19 (0.1)	0.11 (0.3)
Gadicula argentina	GADIARG	74 (13)	-1.5 (0.7)	1.09 (0.2)	95 (15)	4.6 (1.7)	-0.69 (0.2)	0.52 (0.2)
Lepidprhombus whiffigonis	LEPIWHI	58 (57)	-0.88 (1.5)	-0.61 (0.5)	22 (1)	-17 (<0.1)	0.75 (0.03)	1 (0.4)
Micromestitius poutassou	MICMPOU	9927 (223)	-2.71 (0.6)	1.12 (0.2)	10660 (315)	-17 (<0.1)	0.67 (0.02)	1.01 (0.2)
Trachurus trachurus	TRACTRA	2394 (383)	-3.15 (0.5)	0.77 (0.2)	2823 (812)	-5.81 (1.0)	0.11 (0.04)	0.61 (0.2)
Trisopterus minutus	TRISMIN	860 (82)	-0.3 (0.3)	0.06 (0.2)	-	-	-	-
Trisopterus minutus	TRISMIN	860 (82)	-0.3 (0.3)	0.06 (0.2)	-	-	-	-

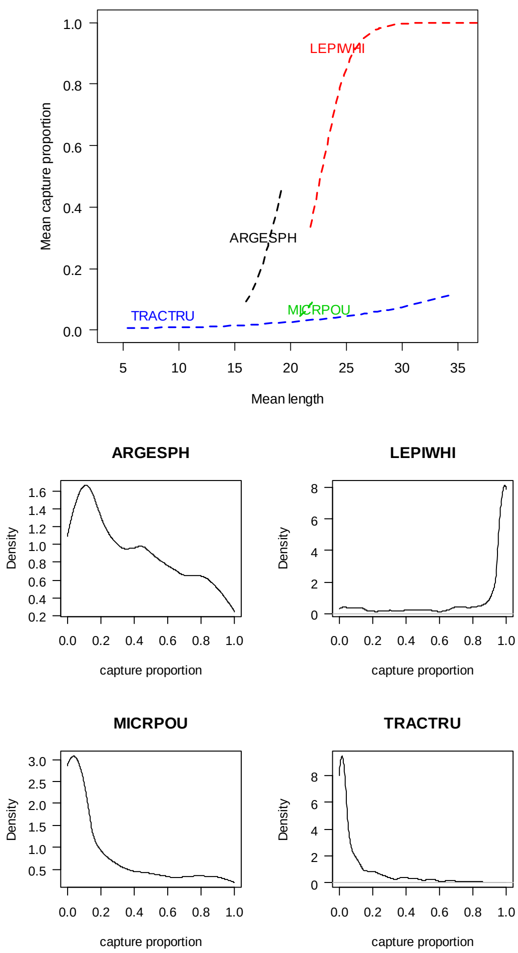

Figure 6.2-1: Mean capture proportions and distributions of capture proportions for model 2 for selected species from the French Celtic Sea groundfish survey using a GOV bottom trawl.

6.3 Case study 2: Estimation of the demersal fish biomass of the North Sea

6.3.1 Introduction

In order to estimate the absolute abundance of fish species from research surveys the catchability (q) of the gear must be known, the q value is analogous to a probability of capture. A q value less than 1 indicates that not all fish that were in the path of a trawl were actually caught by the trawl. A q value greater than 1 implies that the trawl gear was able to herd fish in to the path of the gear (Harley *et al*., 2001). This interpretation at the individual level is applied to entire surveys but there are other reasons at the survey level why a catchability value could be greater or less than 1. For example if a survey detects an anomalously dense aggregation of fish in a year then a q value may be greater than 1 while if a survey fails to detect any aggregations of fish then this could further decrease the probability of capture q value.

Using the IBTS data series and the VPA, estimates of catchability at length for 5 demersal fish species were made. These catchability values were used to estimate biomass of all demersal fish in the North Sea for the time period 1998 to 2004.

6.3.2 Methods

The International Bottom Trawl Survey (IBTS) data for Quarter 3 (Q3) for the time period 1998 to 2004 was used to calculate the catch of demersal fish in the GOV (Grande Ouverture Verticale). Other GOV trawls carried in Q3, though not submitted to ICES, were also used. Data were ‘cleaned’ using the methods described in Daan (2001). Only valid tows of 30 minutes in duration were used in the analysis and only demersal fish were considered. Appendix 6.3-1 lists all species caught in the Q3 IBTS and indicates those which were considered demersal fish. Over the period 1998 to 2004, 139 different species were caught in the Q3 IBTS, 104 of which were demersal.

Figure 1 shows the extent of ICES area IV. The shaded portion represents the area covered by the IBTS survey over the time period 1998 to 2004; areas in white are not covered by the IBTS. As fishing did not take place in all the statistical rectangles within ICES areas IV, a raising factor *RF* was used to multiply the biomass estimates up to the total area of area IV

$$RF = \frac{A_{ICES}}{A_{IBTS}}$$

where A_{ICES} is the area of area IV in Km^2 and A_{IBTS} is the area covered by the IBTS survey in Km^2. To take account of the fact that fish were not evenly distributed across the North Sea, raising factors were determined for the five separate sub-areas indicated in Figure 6.3-1.

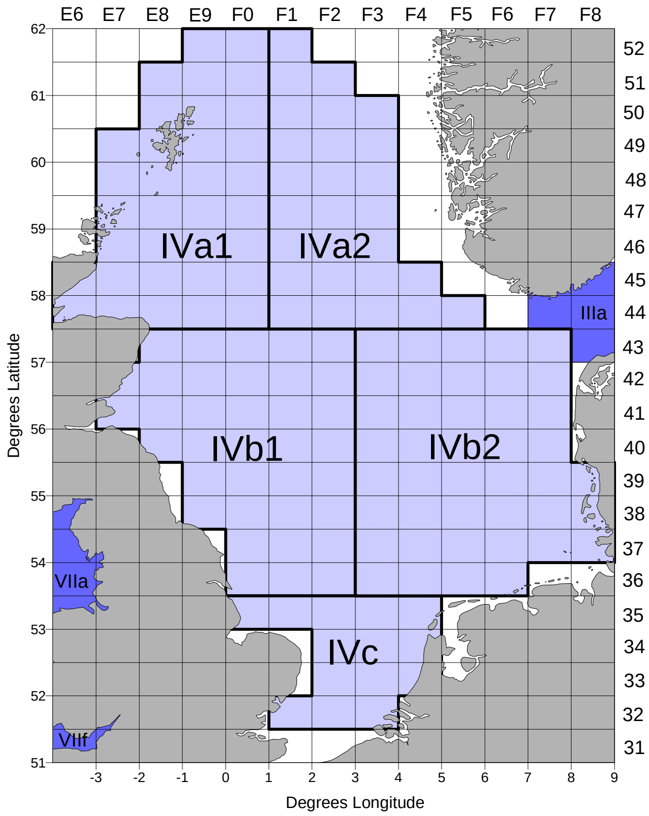

Figure 6.3-1: The extent of ICES area IV. Areas shaded white are not sampled during the IBTS survey. The area covered by the survey was divided in to five sub areas.

In each year, there were at least two statistical rectangles within statistical area IV where no fishing took place. For these missing rectangles an IBTS derived biomass estimate value was interpolated based on the mean of the IBTS derived biomass estimates in the surrounding statistical rectangles.

The VPA is an estimate of the number of fish in each age class on 1 January in each year (ICES, 2005) and assumes that the catch at age is known without error. The Q3 IBTS data was collected in July to September. Some level of mortality would have taken place in the 9 months since the date of the VPA estimates. In order for the two estimates of fish number at age to be comparable, a level of mortality was applied to the VPA data.

$$N_{survey} = N_{Jan1} \times \exp^{(-p*z)}$$

where N_{Jan1} was the number of fish in each age-class at the beginning of each year, z was the total mortality for the whole year (fishing mortality F plus natural mortality M) and p was the time of year the survey takes place (0.75 for a survey carried out in Q3).

There are five demersal species for which a VPA assessment is carried out by the Working Group for the Assessment of Demersal Stocks in the North Sea and Skagerrak (WGNSSK) and which are regularly caught in the IBTS survey (haddock *Melanogrammus aeglefinus*, whiting *Merlangius merlangus*, cod *Gadus morhua*, Norway pout *Trisopterus esmarkii* and plaice, *Pleuronectes platessa*). Each of the assessed species have different ICES areas included in their assessment. Table 6.3-1 shows the ICES areas which are included in the assessment of haddock, whiting, cod, Norway pout and plaice.

Table 6.3-1: ICES areas included in the assessment of each VPA species and the proportion of the total assessed area that they represent.

SPECIES	ICES AREA COVERED IN VPA ASSESSMENT	PROPORTION OF TOTAL AREA (%)
Haddock	IIIa	9.1
	IV	90.9
Whiting	IV	92.1
	VIId	8.6
Cod	IIIa	8.4
	IV	84.3
	VIId	7.2
Norway pout	IIIa	9.1
	IV	90.9
Plaice	IV	100

In order to compare the results from the IBTS (which only covers area IV) with the VPA the proportion of the total assessed area which was made up by ICES area IV was calculated. This proportion was multiplied by the biomass estimates from WGNSSK to give a biomass just for ICES area IV that would be comparable with the IBTS biomass estimates.

Length frequency data from the IBTS was converted to number at age data using age-length keys collected on each survey.

Due to differences in the behaviour of haddock, whiting, cod, Norway pout and plaice, different swept area measurements were used to estimate *CPUE* of each species. For haddock and whiting the area swept by the doors was used as these species are thought to be herded by the doors. The area swept by the wings was used to estimate the *CPUE* of cod, Norway pout and plaice as these species are not thought to be herded by the doors.

Once the number per age-class had been calculated then the catchability q could be estimated (for each age-class) using the following equation:

$$q = \frac{CPUE}{N}$$

where *CPUE* was equal to the number per age-class caught in the IBTS Q3 survey and N was the number in a particular age class as estimated by the VPA.

Figure 6.3-2 shows that when catchability at age was calculated for each assessed species, some estimates of q for haddock and whiting exceeded 1.

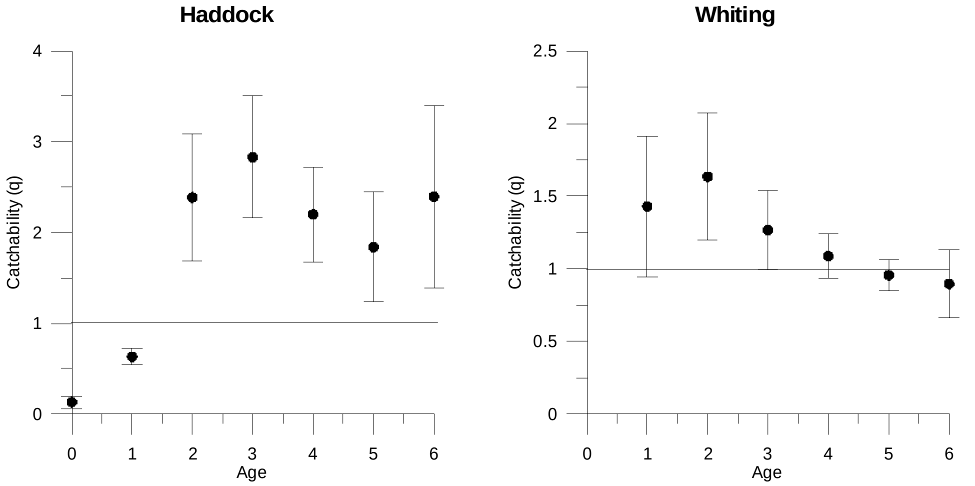

Figure 6.3-2: The catchability at age of haddock and whiting estimated using the mean density per statistical rectangle.

When examining the IBTS data it was apparent that where there were multiple hauls within a statistical rectangle there were often large difference between catch rates. Using the average catch within a statistical rectangle was biasing the density values and making them too high giving values of *q* greater than 1. By using the geometric mean densities of each statistical rectangle, values of *q* were reduced. Figure 6.3-3 shows that even using the geometric mean some catchabilities were still greater than 1. For the purpose of this analysis catchabilities greater than 1 were assumed to be equal to 1.

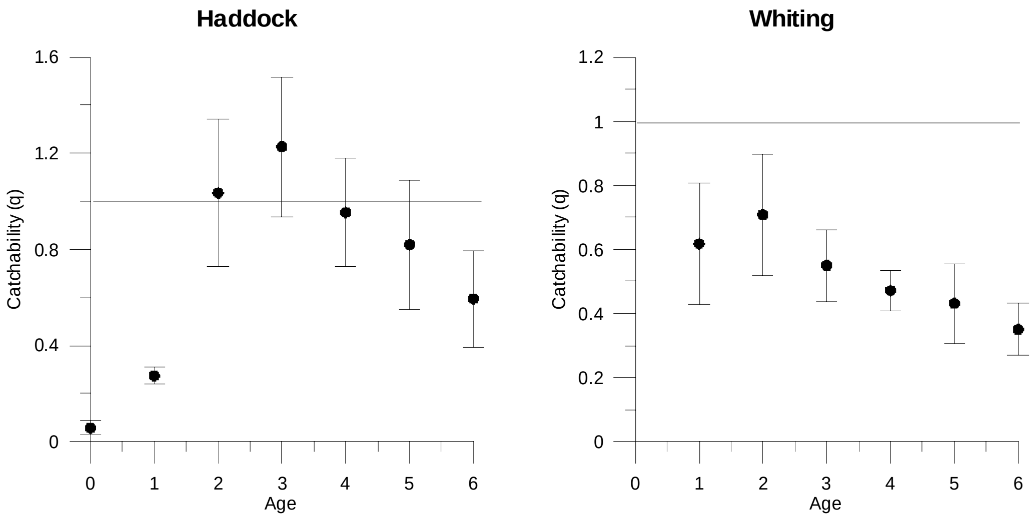

Figure 6.3-3: The catchability of haddock and whiting estimated using the geometric mean per statistical rectangle.

In order to estimate catchability at length, catchability at age was converted to catchability at length using the age-length keys collected on the Q3 IBTS surveys. Catchability at length for the 5 assessed species can be seen in Figure 6.3-4.

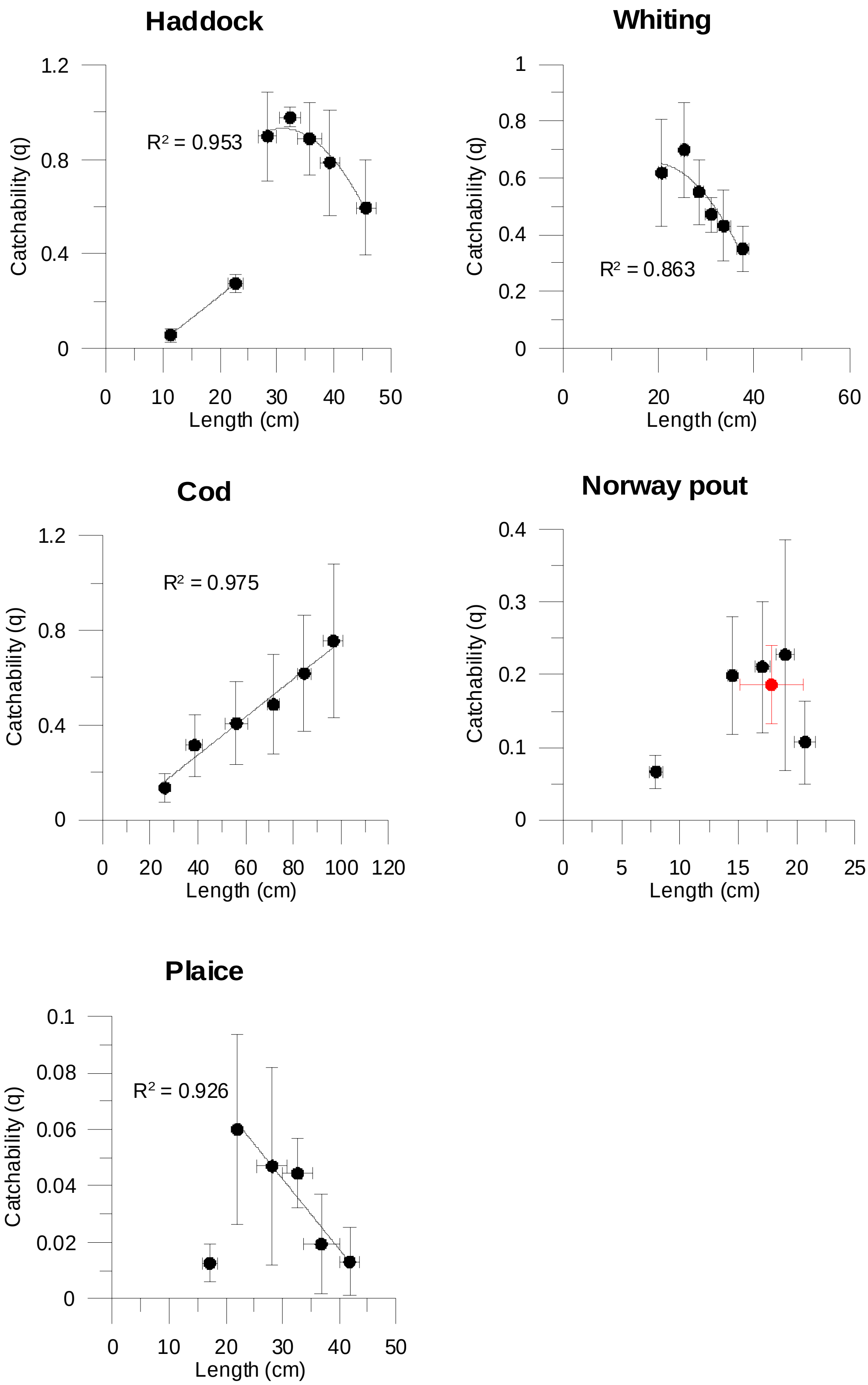

Figure 6.3-4: Catchability at length for each assessed species.

To calculate catchability based on size class, the average length at age for each species was determined and plotted against *q* for each of the assessed species and a relationship fitted through the data (Figure 6.3-4). In some cases more than one relationship was used. For Norway pout there was no significant difference between the *q* of the four largest size classes so the average q of the 0 age class was used for the smallest size classes and the average q of the 1+ age groups was applied to the larger sizes. Once the most appropriate relationship between length and q was established catchability was calculated for every 1 cm length.

Catchability values could only be estimated for 5 of the species routinely caught in the IBTS. In order to get a total biomass of all the demersal fish in the North Sea, catchabilities for all demersal species both commercial and non-commercial need to be established.

In order to allocate the 99 non-assessed demersal species to one of the 5 catchability groups catch ratios between the GOV and 8-metre Beam Trawl (8BT) were examined. Data from the 8BT survey which was carried out by the Netherlands at the same time as the IBTS Q3 survey in the southern North Sea were used to compare the catches of demersal species in each gear. In order to find GOV trawls which were carried out at the same time and in the same area as an 8-metre Beam Trawl (8BT) each statistical rectangle was divided in to 9 'mini' rectangles (Figure 6.3-5).

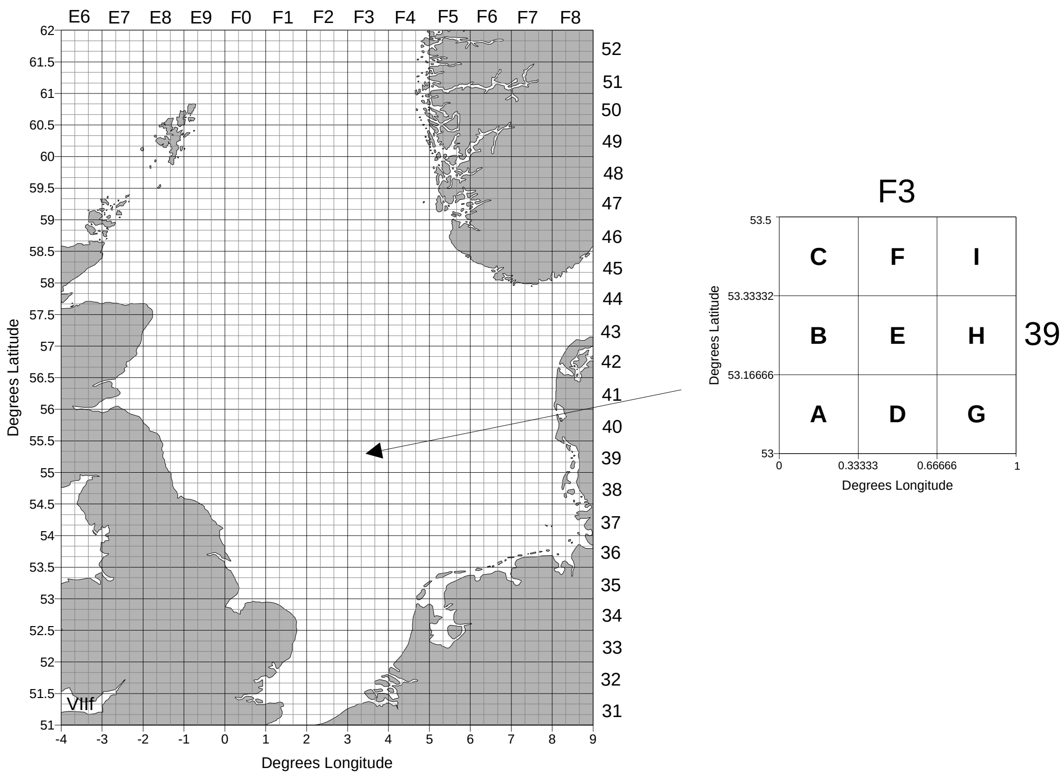

Figure 6.3-5: Division of statistical rectangles in to 'mini' rectangles.

If a valid GOV and 8BT haul were located within the same mini statistical rectangle in the same year they were considered a 'paired haul'. All fish caught in each haul were assigned a 5 cm length class and the density of each species at each length class calculated corrected to area swept by the gear. A catch ratio was then calculated:

$$CR = Log\left(\frac{N_{GOV+0.01}}{N_{8BT+0.01}}\right)$$

where CR is the catch ratio, N_{GOV} was the number of fish caught by the GOV and N_{8BT} was the number of fish caught by the 8BT. If catch ratio was positive then more fish were caught in the GOV, if catch ratio was negative then more fish were caught in the 8BT. Only positive paired hauls were used when considering catch ratio's i.e. where both hauls contained fish of the same species and length class.

Catch ratio was plotted against *q* for each of the assessed species. The non-assessed demersal species were placed in a group based on the similarity of their catch ratio's at length class to one of the assessed species. Figures 6.3-6 to 6.3-9 show catch ratio plotted against length for each of the assessed species. It is clear from Figure 6.3-6 that both haddock and whiting were caught in greater number in the GOV even when using the area swept by the doors. As a catchability at length for both species was estimated they were each assigned a group of their own. Norway pout was caught very effectively in the GOV (Figure 6.3-7). Poor cod was also placed in this group mainly due to the similarity of the body size and shape of the species, even though it is clear from Figure 6.3-7 that poor cod was caught less effectively with increasing size in the GOV. Figure 6.3-8 shows the catch ratio's of the group where cod is the standard species. Common dab, grey gurnard, long rough dab, lemon sole and red mullet were also placed in this group on account of their catch ratio at length being similar to that of cod. Figure 6.3-9 shows the catch ratio's of the group where plaice is the standard species. Angler, dragonet, hooknose, sand goby, starry ray, witch, megrim, cuckoo ray and four-bearded rockling were placed in this group on account of their catch ratio at length being similar to that of plaice. Species where there was no catch ratio information were placed in to a group on the basis of similarities in their behaviour and habitat to the standard species. Using this method all 104 demersal species were allocated to one of 5 groups, with at least one of the assessed species the standard species for that group. The group to which each species was assigned is shown in Table 6.3-2.

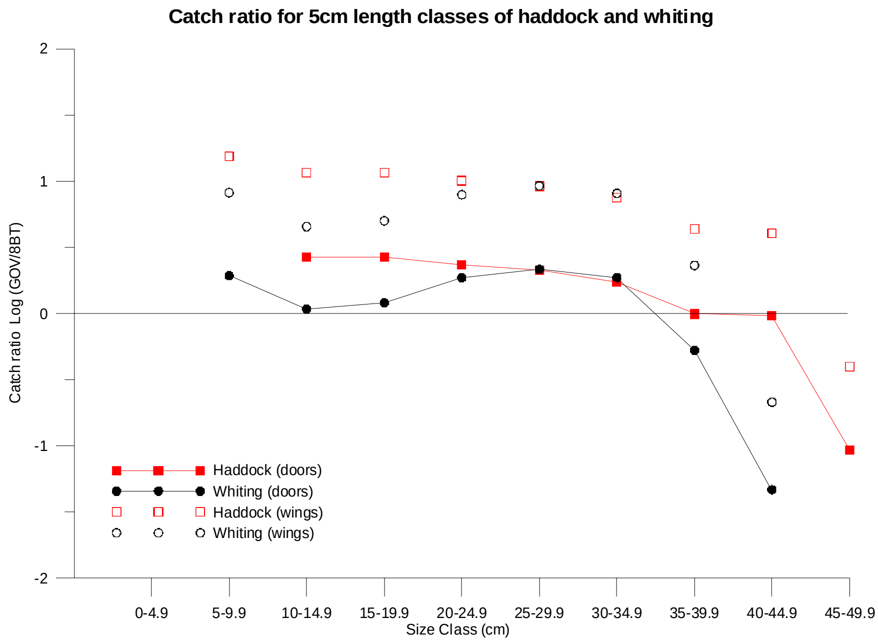

Figure 6.3-6: Catch ratio for 5 cm length classes of haddock and whiting for both the area swept by the doors and the area swept by the wings.

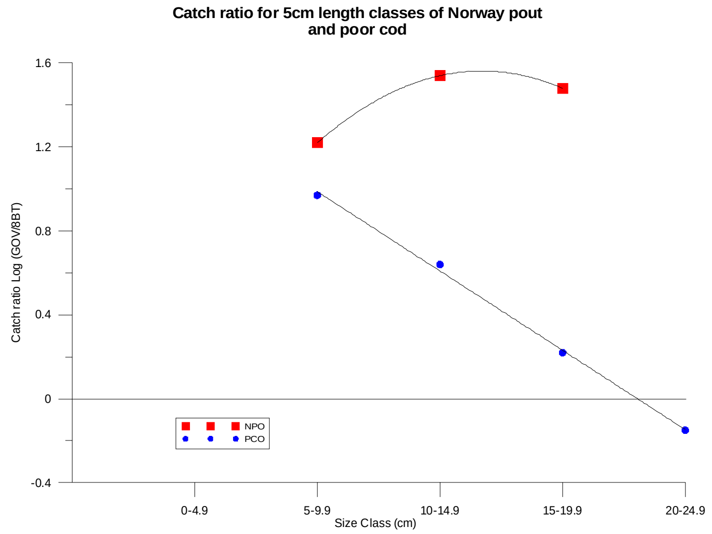

Figure 6.3-7: Catch ratio for 5 cm length classes of Norway pout (NPO) and poor cod (PCO).

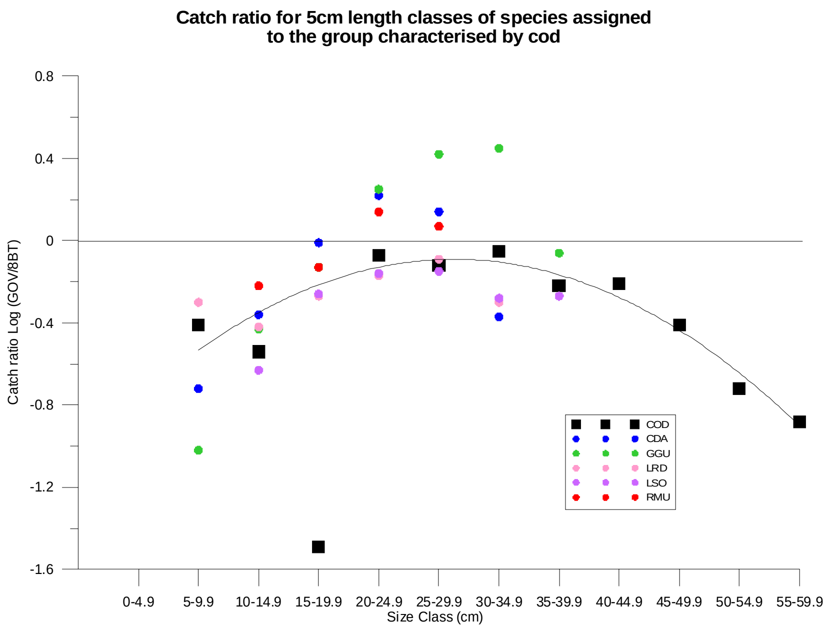

Figure 6.3-8: Catch ratio 5 cm length classes of cod (COD). The catch ratios of common dab (CDA), grey gurnard (GGU), long rough dab (LRD), lemon sole (LSO) and red mullet (RMU) are also shown.

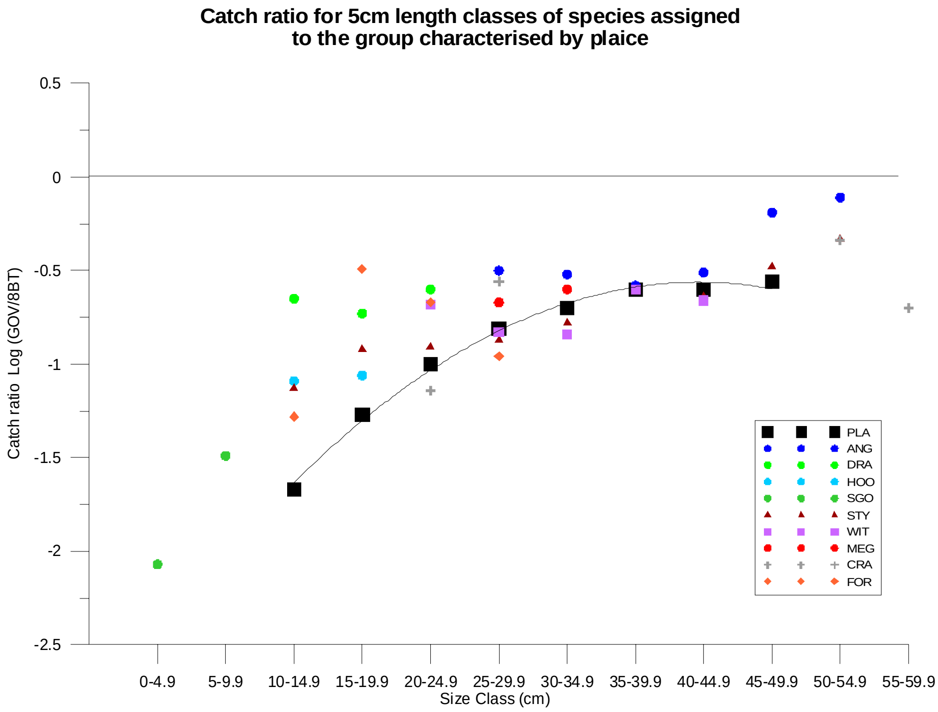

Figure 6.3-9: Catch ratio for 5 cm length classes of plaice (PLA). The catch ratio of angler (ANG), dragonet (DRA), hooknose (HOO), sand goby (SGO), starry ray (STY), witch (WIT), megrim (MEG), cuckoo ray (CRA) and four-bearded rockling (FOR) are also shown.

Table 6.3-2: List of species assigned to each catchability group. Records which are shaded show the standard species for that group.

Group	Species Code	Common name	Scientific name	Swept area measure
1	HAD	Haddock	*Melanogrammus aeglefinus*	Door
2	WHI	Whiting	*Merlangius merlangus*	Door
3	NPO	Norway pout	*Trisopterus esmarkii*	Wings
3	PCO	Poor cod	*Trisopterus minutus*	Wings
4	COD	Cod	*Gadus morhua*	Wings
4	BFI	Boarfish	*Capros aper*	Wings
4	BIB	Bib	*Trisopterus luscus*	Wings
4	BLI	Blue ling	*Molva dypterygia*	Wings
4	BLM	Bluemouth	*Helicolenus dactylopterus*	Wings
4	BMD	Black mouthed dogfish	*Galeus melastomus*	Wings
4	BRI	Brill	*Scophthalmus rhombus*	Wings
4	BRO	Bull rout	*Myoxocephalus scorpius*	Wings
4	BSB	Black sea bream	*Spondyliosoma cantharus*	Wings
4	BUT	Butterfish	*Pholis gunnellus*	Wings
4	CAT	Catfish	*Anarhichas lupus*	Wings
4	CDA	Common dab	*Limanda limanda*	Wings
4	CEE	Conger eel	*Conger conger*	Wings
4	CHI	Rabbit ratfish	*Chimaera monstrosa*	Wings
4	CSB	Couch's sea bream	*Pagrus pagrus*	Wings

Group	Species Code	Common name	Scientific name	Swept area measure
4	FLO	Flounder	*Platichthys flesus*	Wings
4	GFO	Greater forkbeard	*Phycis blennoides*	Wings
4	GGU	Grey gurnard	*Eutrigla gurnardus*	Wings
4	GOM	Golden grey mullet	*Liza aurata*	Wings
4	GRM	Flathead (Grey) mullet	*Mugil cephalus*	Wings
4	GWE	Greater weever	*Trachinus draco*	Wings
4	HAK	Hake	*Merluccius merluccius*	Wings
4	HAL	Halibut	*Hippoglossus hippoglossus*	Wings
4	LIN	Ling	*Molva molva*	Wings
4	LRD	Long rough dab	*Hippoglossoides platessoides*	Wings
4	LSD	Lesser spotted dogfish	*Scyliorhinus canicula*	Wings
4	LSO	Lemon sole	*Microstomus kitt*	Wings
4	LYT	Pollack	*Pollachius pollachius*	Wings
4	MSC	Moustache sculpin	*Triglops murrayi*	Wings
4	NHA	Norway haddock	*Sebastes viviparus*	Wings
4	NTO	Norwegian topknot	*Phrynorhombus norvegicus*	Wings
4	RED	Redfish (*marinus*)	*Sebastes marinus*	Wings
4	RGU	Red gurnard	*Aspitrigla cuculus*	Wings
4	RMM	Red mullet	*Mullus barbatus*	Wings
4	RMU	Striped red mullet	*Mullus surmuletus*	Wings
4	SAI	Saithe	*Pollachius virens*	Wings
4	SCA	Spotted catfish	*Anarhichas minor*	Wings
4	SHO	Smooth hound	*Mustelus mustelus*	Wings
4	SNB	Snake blenny	*Lumpenus lampretaeformis*	Wings
4	SPB	Spotted snake blenny	*Leptoclinus maculatus*	Wings
4	SPU	Spurdog	*Squalus acanthias*	Wings
4	SSC	Sea scorpion	*Taurulus bubalis*	Wings
4	SSH	Starry smooth hound	*Mustelus asterias*	Wings
4	TOP	Tope	*Galeorhinus galeus*	Wings
4	TOR	Torsk	*Brosme brosme*	Wings
4	TPK	Topknot	*Zeugopterus punctatus*	Wings
4	TSC	Twohorn sculpin	*Icelus bicornis*	Wings
4	TUB	Tub gurnard	*Trigla lucerna*	Wings
4	TUR	Turbot	*Psetta maxima*	Wings
4	VBE	Velvet belly	*Etmopterus spinax*	Wings
4	WEE	Lesser weever	*Echiichthys vipera*	Wings
5	PLA	Plaice	*Pleuronectes platessa*	Wings
5	ANG	Angler	*Lophius piscatorius*	Wings
5	BAN	Black bellied angler	*Lophius budegassa*	Wings
5	BGO	Black goby	*Gobius niger*	Wings
5	BRA	Blond ray	*Raja brachyura*	Wings
5	CGO	Crystal goby	*Crystallogobius linearis*	Wings
5	COG	Common goby	*Pomatoschistus microps*	Wings
5	CRA	Cuckoo ray	*Leucoraja naevus*	Wings
5	DRA	Dragonet	*Callionymus lyra*	Wings
5	DSO	Dover sole	*Solea vulgaris*	Wings
5	EEE	Esmark's eelpout	*Lycodes esmarkii*	Wings
5	FGO	Fries's goby	*Lesueurigobius friesii*	Wings
5	FIR	Five-bearded rockling	*Ciliata mustela*	Wings
5	FOR	Four-bearded rockling	*Enchelyopus cimbrius*	Wings
5	GPI	Great pipefish	*Syngnathus acus*	Wings

GROUP	SPECIES CODE	COMMON NAME	SCIENTIFIC NAME	SWEPT AREA MEASURE
5	HAG	Hagfish	*Myxine glutinosa*	Wings
5	HOO	Hooknose	*Agonus cataphractus*	Wings
5	ISC	Imperial scaldfish	*Arnoglossus imperialis*	Wings
5	LNS	Long nosed skate	*Dipturus oxyrinchus*	Wings
5	LUM	Lumpsucker	*Cyclopterus lumpus*	Wings
5	MEG	Megrim	*Lepidorhombus whiffiagonis*	Wings
5	MSS	Montagu's sea snail	*Liparis montagui*	Wings
5	NPI	Nilsson's pipefish	*Syngnathus rostellatus*	Wings
5	NRO	Northern rockling	*Ciliata septentrionalis*	Wings
5	PBU	Pricklebacks unidentified	*Stichaeidae*	Wings
5	PEF	Pearlfish	*Echiodon drummondii*	Wings
5	PGO	Painted goby	*Pomatoschistus pictus*	Wings
5	RDR	Reticulated dragonet	*Callionymus reticulatus*	Wings
5	SAR	Sandy ray	*Leucoraja circularis*	Wings
5	SCF	Scaldfish	*Arnoglossus laterna*	Wings
5	SDR	Spotted dragonet	*Callionymus maculatus*	Wings
5	SGO	Sand goby	*Pomatoschistus minutus*	Wings
5	SKA	Skate	*Dipturus batis*	Wings
5	SLA	Sea Lamprey	*Petromyzon marinus*	Wings
5	SOL	Solenette	*Solea lutea*	Wings
5	SPI	Snake pipefish	*Entelurus aequoreus*	Wings
5	SPY	Spotted ray	*Raja montagui*	Wings
5	SRA	Shagreen ray	*Leucoraja fullonica*	Wings
5	SSN	Sea snail	*Liparis liparis*	Wings
5	SSO	Sand sole	*Solea lascaris*	Wings
5	STY	Starry ray	*Amblyraja radiata*	Wings
5	TBR	Three-bearded rockling	*Gaidropsarus vulgaris*	Wings
5	TCL	Two-sptted clingfish	*Diplecogaster bimaculata*	Wings
5	TFI	Tadpole fish	*Raniceps raninus*	Wings
5	TRA	Thornback ray	*Raja clavata*	Wings
5	TSO	Thickback sole	*Microchirus variegatus*	Wings
5	VBL	Viviparous blenny	*Zoarces viviparus*	Wings
5	VEE	Vahl's eelpout	*Lycodes vahlii*	Wings
5	WIT	Witch	*Glyptocephalus cynoglossus*	Wings

Using the methods above three different biomass estimates were produced:

- The first estimate was calculated using the catchability at length for each standard species and applying it to all species within that group.
- The second estimate was calculated using the catchability at length for each standard species. For all other species in the group the average catchability of the standard species was used
- The third estimate was calculated using the catchability at length for each standard species. For all other species in groups 4 and 5 (characterised by cod and plaice) the average catchability at length of cod and plaice was used.

6.3.3 Results

The total demersal fish biomass of the North Sea as estimated using the three methods described is shown in Figure 6.3-10a. Using method 1, biomass estimates range from 6 million tonnes in 2004 to 12 million tonnes in 2000. Method 2 gives the lowest biomass, a low of 3.5 million tonnes in 2004 and a high of 9.5 million tonnes in 2000. Method 3 gives very similar biomass estimates to method 1. Due to the fact that the GOV is very poor at catching small fish, the total biomass of fish greater than 10 cm is shown in Figure 6.3-10b. On average the biomass of fish greater than 10 cm were 0.5 million tonnes less than those of total fish biomass. The main exception was in 1999 when fish below 10 cm contributed 3.5 million tonnes to the total biomass.

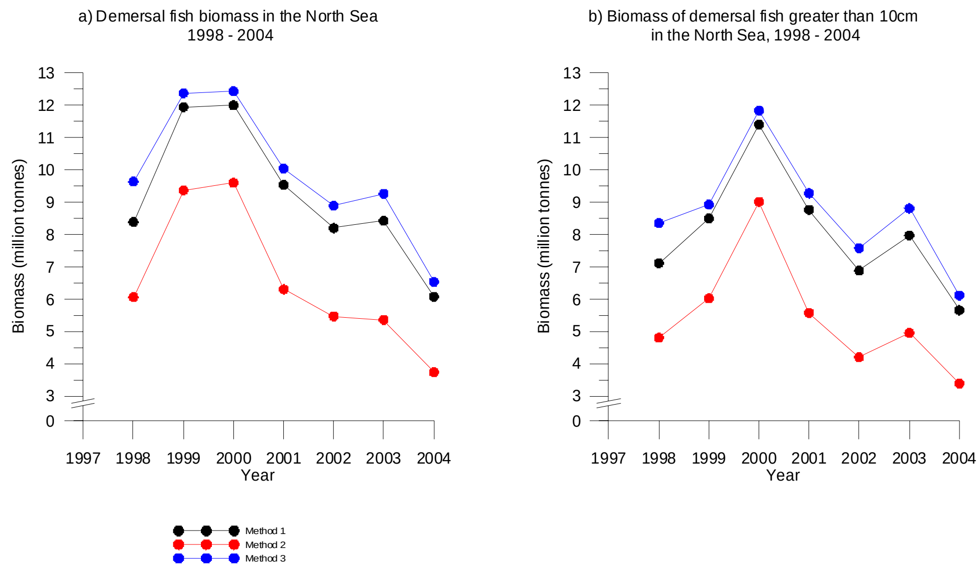

Figure 6.3-10: North Sea biomass estimates produced using the three different methods, a) shows total demersal fish biomass, b) shows the biomass of demersal fish >10cm.

6.3.4 Discussion

Previous attempts at calculating total North Sea biomass by Sparholt (1990) estimated biomass in the first quarter at 5.9 million tonnes 13.1 million tonnes in the third quarter. The three estimates of demersal fish biomass made in here are all in line with those estimated by Sparholt. When estimating the catchability of the GOV in the way described here, several things that should be considered are;

- As most species are placed in the cod and plaice groups, the total demersal fish biomass is essentially driven by the catchability of these two species.
- Fish below 10 cm are not sampled well by the GOV and therefore the biomass estimates of small species should be used with caution.

Appendix 6.3-1: List of all species caught in valid tows in the North Sea Q3 IBTS, 1998 to 2004. Only species considered demersal were considered in the analysis.

Species Code	Common name	Scientific name	Type
ANC	Anchovy	*Engraulis encrasicolus*	Pelagic
ANG	Angler	*Lophius piscatorius*	Demersal
ARU	Argentines	*Argentinidae*	Pelagic
ASH	Allis shad	*Alosa alosa*	Pelagic
BAN	Black bellied angler	*Lophius budegassa*	Demersal
BAS	Bass	*Dicentrarchus labrax*	Pelagic
BFI	Boarfish	*Capros aper*	Demersal
BGO	Black goby	*Gobius niger*	Demersal
BIB	Bib	*Trisopterus luscus*	Demersal
BLI	Blue ling	*Molva dypterygia*	Demersal
BLM	Bluemouth	*Helicolenus dactylopterus*	Demersal
BMD	Black mouthed dogfish	*Galeus melastomus*	Demersal
BRA	Blond ray	*Raja brachyura*	Demersal
BRI	Brill	*Scophthalmus rhombus*	Demersal
BRO	Bull rout	*Myoxocephalus scorpius*	Demersal
BSB	Black sea bream	*Spondyliosoma cantharus*	Demersal
BUT	Butterfish	*Pholis gunnellus*	Demersal
BWH	Blue whiting	*Micromesistius poutassou*	Pelagic
CAT	Catfish	*Anarhichas lupus*	Demersal
CBS	Corbin's sandeel	*Hyperoplus immaculatus*	Pelagic
CDA	Common dab	*Limanda limanda*	Demersal
CEE	Conger eel	*Conger conger*	Demersal
CGO	Crystal goby	*Crystallogobius linearis*	Demersal
CHI	Rabbit ratfish	*Chimaera monstrosa*	Demersal
COD	Cod	*Gadus morhua*	Demersal
COG	Common goby	*Pomatoschistus microps*	Demersal
CRA	Cuckoo ray	*Leucoraja naevus*	Demersal
CSA	Common sandeel	*Ammodytes tobianus*	Pelagic
CSB	Couch's sea bream	*Pagruss pagrus*	Demersal
DRA	Dragonet	*Callionymus lyra*	Demersal
DSO	Dover sole	*Solea solea*	Demersal
EEE	Esmark's eelpout	*Lycodes esmarkii*	Demersal
EEL	European eel	*Anguilla anguilla*	Pelagic
FGO	Fries's goby	*Lesueurigobius friesii*	Demersal
FIR	Five-bearded rockling	*Ciliata mustela*	Demersal
FLO	Flounder	*Platichthys flesus*	Demersal
FOR	Four-bearded rockling	*Enchelyopus cimbrius*	Demersal
FST	Fifteen spined stickleback	*Spinachia spinachia*	Pelagic
GAR	Greater argentine	*Argentina silus*	Pelagic
GFO	Greater forkbeard	*Phycis blennoides*	Demersal
GGU	Grey gurnard	*Eutrigla gurnardus*	Demersal
GOM	Golden grey mullet	*Liza aurata*	Demersal
GPI	Great pipefish	*Syngnathus acus*	Demersal
GRF	Garfish	*Belone belone*	Pelagic
GRM	Flathead (Grey) mullet	*Mugil cephalus*	Demersal
GSA	Greater sandeel	*Hyperoplus lanceolatus*	Pelagic
GWE	Greater weever	*Trachinus draco*	Demersal
HAD	Haddock	*Melanogrammus aeglefinus*	Demersal
HAG	Hagfish	*Myxine glutinosa*	Demersal

Species Code	Common name	Scientific name	Type
HAK	Hake	*Merluccius merluccius*	Demersal
HAL	Halibut	*Hippoglossus hippoglossus*	Demersal
HER	Herring	*Clupea harengus*	Pelagic
HMA	Horse mackerel	*Trachurus trachurus*	Pelagic
HOO	Hooknose	*Agonus cataphractus*	Demersal
ISC	Imperial scaldfish	*Arnoglossus imperialis*	Demersal
JDO	John dory	*Zeus faber*	Pelagic
LAR	Lesser argentine	*Argentina sphyraena*	Pelagic
LBA	Ribbon barracudina	*Arctozenus risso*	Pelagic
LIN	Ling	*Molva molva*	Demersal
LNS	Long nosed skate	*Dipturus oxyrinchus*	Demersal
LRD	Long rough dab	*Hippoglossoides platessoides*	Demersal
LSD	Lesser spotted dogfish	*Scyliorhinus canicula*	Demersal
LSO	Lemon sole	*Microstomus kitt*	Demersal
LUM	Lumpsucker	*Cyclopterus lumpus*	Demersal
LYT	Pollack	*Pollachius pollachius*	Demersal
MAC	Mackerel	*Scomber scombrus*	Pelagic
MEG	Megrim	*Lepidorhombus whiffiagonis*	Demersal
MSC	Moustache sculpin	*Triglops murrayi*	Demersal
MSS	Montagu's sea snail	*Liparis montagui*	Demersal
NHA	Norway haddock	*Sebastes viviparus*	Demersal
NPI	Nilsson's pipefish	*Syngnathus rostellatus*	Demersal
NPO	Norway pout	*Trisopterus esmarkii*	Demersal
NRO	Northern rockling	*Ciliata septentrionalis*	Demersal
NTO	Norwegian topknot	*Phrynorhombus norvegicus*	Demersal
PBU	Pricklebacks unidentified	*Stichaeidae*	Demersal
PCO	Poor cod	*Trisopterus minutus*	Demersal
PEA	Pearlside	*Maurolicus muelleri*	Pelagic
PEF	Pearlfish	*Echiodon drummondii*	Demersal
PGO	Painted goby	*Pomatoschistus pictus*	Demersal
PIL	Pilchard	*Sardina pilchardus*	Pelagic
PLA	Plaice	*Pleuronectes platessa*	Demersal
RDR	Reticulated dragonet	*Callionymus reticulatus*	Demersal
RED	Redfish (*marinus*)	*Sebastes marinus*	Demersal
RGU	Red gurnard	*Aspitrigla cuculus*	Demersal
RLA	European river lamprey	*Lampetra fluviatilis*	Pelagic
RMM	Red mullet	*Mullus barbatus*	Demersal
RMU	Striped red mullet	*Mullus surmuletus*	Demersal
RSA	Raitt's sandeel	*Ammodytes marinus*	Pelagic
SAI	Saithe	*Pollachius virens*	Demersal
SAL	Salmon	*Salmo salar*	Pelagic
SAR	Sandy ray	*Leucoraja circularis*	Demersal
SCA	Spotted catfish	*Anarhichas minor*	Demersal
SCF	Scaldfish	*Arnoglossus laterna*	Demersal
SDR	Spotted dragonet	*Callionymus maculatus*	Demersal
SGO	Sand goby	*Pomatoschistus minutus*	Demersal
SHO	Smooth hound	*Mustelus mustelus*	Demersal
SJD	Silvery John Dory	*Zenopsis conchifera*	Pelagic
SKA	Skate	*Dipturus batis*	Demersal
SLA	Sea Lamprey	*Petromyzon marinus*	Demersal
SNB	Snake blenny	*Lumpenus lampretaeformis*	Demersal
SOL	Solenette	*Solea lutea*	Demersal

SPECIES CODE	COMMON NAME	SCIENTIFIC NAME	TYPE
SPB	Spotted snake blenny	*Leptoclinus maculatus*	Demersal
SPI	Snake pipefish	*Entelurus aequoreus*	Demersal
SPO	Silvery pout	*Gadiculus argenteus*	Pelagic
SPR	Sprat	*Sprattus sprattus*	Pelagic
SPU	Spurdog	*Squalus acanthias*	Demersal
SPY	Spotted ray	*Raja montagui*	Demersal
SRA	Shagreen ray	*Leucoraja fullonica*	Demersal
SSA	Smooth sandeel	*Gymnammodytes semisquamatus*	Pelagic
SSC	Sea scorpion	*Taurulus bubalis*	Demersal
SSH	Starry smooth hound	*Mustelus asterias*	Demersal
SSN	Sea snail	*Liparis liparis*	Demersal
SSO	Sand sole	*Solea lascaris*	Demersal
STY	Starry ray	*Amblyraja radiata*	Demersal
SUN	Sandeels unidentified	*Ammodytidae*	Pelagic
TBR	Three-bearded rockling	*Gaidropsarus vulgaris*	Demersal
TCL	Two-sptted clingfish	*Diplecogaster bimaculata*	Demersal
TFI	Tadpole fish	*Raniceps raninus*	Demersal
TOP	Tope	*Galeorhinus galeus*	Demersal
TOR	Torsk	*Brosme brosme*	Demersal
TPK	Topknot	*Zeugopterus punctatus*	Demersal
TRA	Thornback ray	*Raja clavata*	Demersal
TRO	Trout	*Salmo trutta*	Pelagic
TSC	Twohorn sculpin	*Icelus bicornis*	Demersal
TSO	Thickback sole	*Microchirus variegatus*	Demersal
TST	Three-spined stickleback	*Gasterosteus aculeatus*	Pelagic
TUB	Tub gurnard	*Trigla lucerna*	Demersal
TUR	Turbot	*Psetta maxima*	Demersal
TWS	Twaite shad	*Alosa fallax*	Pelagic
VBE	Velvet belly	*Etmopterus spinax*	Demersal
VBL	Viviparous blenny	*Zoarces viviparus*	Demersal
VEE	Vahl's eelpout	*Lycodes vahlii*	Demersal
WEE	Lesser weever	*Echiichthys vipera*	Demersal
WHI	Whiting	*Merlangius merlangus*	Demersal
WIT	Witch	*Glyptocephalus cynoglossus*	Demersal

6.4 References

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7 Upcoming nature conservation issues for marine fishes

7.1 Introduction

WGFE had the following ToR: "d) address any upcoming nature conservation issues for marine fishes including their value as indicators in the context of the Water Framework Directive". This refers to the recommendations in WGFE 2004 Report, which included "...address any upcoming nature conservation issues for marine fishes".

Nature conservation issues are of increasing importance in the ICES area. ICES needs to be prepared for future tasks and challenges, and should be in a position to provide advice to address upcoming marine nature conservation issues.

7.2 Upcoming conservation issues

The following conservation issues are considered to be relevant for ICES area:

- UN: Conservation of marine species and habitats in international waters (UN)
- IUCN Red List of Threatened Species
- CITES convention
- CMS (Bonn convention)
- OSPAR List of Threatened or Declining Habitats and Species
- HELCOM List of Threatened and/or Declining Habitats and Species
- EU Habitats Directive
- EU Water Framework Directive
- COSEWIC / SARA
- USA Endangered Species Act

Many of the above have been reviewed by WGEF in preceding reports (ICES, 2004; 2005)

7.2.1 UN: Conservation of marine species and habitats in international waters

On 29 November 2005, the United Nations General Assembly (UN GA) reaffirmed its call for nations to take 'urgent action' to protect deep-sea corals, seamounts and hydrothermal vent ecosystems from damage by bottom trawl fishing, but stopped short of agreeing to declare a halt to the practice in international waters. A report released by UNEP in 2004 singled out bottom trawl fishing, the most widely used method of fishing deep-sea bottom species such as orange roughy, deep-sea halibut and grenadiers on the high seas, as the greatest threat to deep-water coral ecosystems.

7.2.2 IUCN Red List of Threatened Species

The aims and objectives of the IUCN (http://www.iucn.org/) and its Red List of Threatened Species were discussed in detail in a preceding WGFE report (ICES, 2004). The IUCN has various specialist groups (e.g. Coral Reef Fish Specialist Group, Grouper and Wrasse Specialist Group, Salmon Specialist Group, Shark Specialist Group), and the Shark Specialist Group (SSG) is currently attempting to assess the status of all chondrichthyan species. To this end, the SSG organised a meeting to assess the chondrichthyan fishes of the North-east Atlantic (Peterborough, UK, 13–15 February 2006).

7.2.3 CITES

Aims and objectives of the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) (http://www.cites.org/) were discussed in detail in a preceding WGFE report (ICES, 2004). The appendices of the convention list species for which international trade is permitted only in exceptional circumstances (Appendix I) and those for which trade must be controlled in order to avoid utilization incompatible with their survival (Appendix II). Marine fishes included in these appendices were listed in the report of ICES WGFE 2004.

Amendments to Appendices I and II can be made at the Conference of Parties, where proposals from member states are discussed and put to the vote. Under EC regulations, wildlife trade is an area of exclusive community competence and as such, any such proposals from EC Member States have to be approved by a qualified majority of the committee on trade in wild fauna and flora. If the committee is unable to reach a decision, it must be referred to the council. Proposals to add porbeagle shark (*Lamna nasus*) and spurdog (*Squalus acanthias*) to Appendix II of the convention at the 13th CoP in 2004 were rejected by the council but it is likely that revised proposals for these species, and possibly new proposals for other species, will be put before the committee by Member States before the next CoP in 2007.

7.2.4 Convention on the Conservation of Migratory Species of Wild Animals

The Convention on the Conservation of Migratory Species of Wild Animals (CMS, also known as the Bonn convention, http://www.cms.int/) is an intergovernmental treaty, concluded under the aegis of the United Nations Environment Programme. Migratory species threatened with extinction are listed on Appendix I of the Convention. CMS Parties strive towards strictly protecting these animals, conserving or restoring their habitats, mitigating obstacles to migration and controlling other factors that might endanger them. CMS encourages concerted action among range states including the negotiation of regional agreements and memoranda of understanding.

Three marine fish species, white shark, basking shark and whale shark, have recently been added to appendix I. Several countries within ICES are range states for the basking shark and some may possibly be range states for white shark. The majority of ICES member states are parties to the convention and advice on these species may be solicited.

7.2.5 OSPAR List of threatened or declining habitats and species

Aims and objectives of the Convention for the Protection of the Marine Environment of the North-east Atlantic (OSPAR) (http://www.ospar.org/) were discussed in detail in a preceding WGFE report (ICES, 2004). The 2004[1] Initial OSPAR List of Threatened and/or Declining Species and Habitats (Reference Number: 2004-06) may be subject of future revision. In 2005, the European Commission (DG Environment) asked OSPAR for a list of candidate habitats and species for expanding the EU Habitats Directive (see Section 7.8) to cover marine habitats and species. Candidates may include fish species that are on the existing 2004 list.

7.2.6 HELCOM List of Threatened and/or declining habitats and species

Aims and objectives of the Convention for the Protection of the Marine Environment of the Baltic Sea area (HELCOM) (http://www.helcom.fi/) were discussed in detail in a preceding

1 Note: OSPAR 2003 adopted the Initial List of Threatened and/or Declining Species and Habitats. OSPAR 2004 updated this list with the addition of two further fish species and four further habitats and made some further editorial changes.

WGFE report (ICES, 2004). The Helsinki Commission (HELCOM) held a HELCOM HABITAT Workshop on the Development of an Initial List of Threatened and/or Declining Marine Species and Biotopes/Habitats, Berlin, 1–3 March 2006 (Table 7-1). The Nature Protection and Biodiversity Group (HELCOM HABITAT) was asked to decide on this initial list at its next meeting. The list also includes proposals for candidate habitats and species for the EU Habitats Directive.

Table 7-1: Regions covered by the HELCOM Initial List of Threatened and/or Declining Species of Lampreys and Fishes of the Baltic Sea region.

Abbreviation	Region	Comments
SK	Skagerrak	Not included in HELCOM Area; given as additional information on species distribution
KA	Kattegat	Including areas south of Skagen/Denmark and Göteborg/Sweden
WB	Western Baltic	Including Belt seas and Kieler Bucht
SB	Southern Baltic	Including Skane/Sweden, Bornholm, Mecklenburger Bucht, coast of Poland and Kaliningrad area of Russia
CB	Central Baltic	Including Baltic proper around Öland, Gotland towards Åland, Hiumaa and Saaremaa islands
RI	Gulf of Riga	Including areas east of Sääre/Estonia and Kolka/Latvia, south of Virtsu/Estonia
BO	Gulf of Bothnia	Including areas north of Åland Islands
FI	Gulf of Finland	Including areas east of Hangö/Finland

7.2.7 EU Habitats Directive

Aims and objectives of the EU Habitats Directive (92/43/EEC) (http://europa.eu.int/comm/environment/nature/nature_conservation/eu_nature_legislation/habitats_directive/index_en.htm) were discussed in detail in a preceding WGFE report (ICES, 2004). The EU Commission (DG Environment) is close to adopting a Community network of NATURA 2000 sites in its member states for terrestrial and inland water habitats and species; this network will probably be finalised by the end of 2006. Though the annexes I-V of the EU Habitats Directive are closed for terrestrial and inland water habitats and species (except for habitats and species proposed by new member states), the annexes shall be reviewed in order to include a set of marine habitats and species, to fulfil the recently formulated 'Marine Strategy to save Europe's seas and oceans' (http://europa.eu.int/comm/environment/water/marine.htm). The EU Commission has asked OSPAR, HELCOM and the Barcelona Convention to suggest marine habitats and species as candidates to expand annexes I-V. ICES should be prepared to give advice in this respect.

Some EU member states have already established a set of initial NATURA 2000 sites both in territorial waters and in the EEZ (e.g. sites in the German EEZ, http://www.bfn.de/marinehabitate/de/downloads/erlaeuterungstexte/Karte1_Schutzgebiete_mit_Koordinaten.pdf and http://www.bfn.de/marinehabitate/de/downloads/erlaeuterungstexte/Karte6_Schutzgebiete_mit_Koordinaten.pdf). The aims of these Special Areas for Conservation (SAC) will be to conserve the specific habitat types for which they are designated. ICES may be asked to provide advice on the management and monitoring of such habitats / species.

7.2.8 EU Water Framework Directive

The aim of the EU Water Framework Directive (2000/60/EC) (http://europa.eu.int/comm/environment/water/water-framework/index_en.html) is to establish a framework of EU Community action in the field of water policy, and a framework for the protection of inland surface waters (including transitional waters and coastal waters), and groundwater (Art 1). The directive aims to protect and enhance the status of aquatic ecosystems and the aquatic environment, ensuring the reduction of pollution and the protection of territorial and marine waters. Art 1(16) stresses the necessity of "further integration of protection and sustainable management of water into other Community policy such as ... fisheries ...". Art 1(17) specifies that "an effective and coherent water policy must take account of the vulnerability of aquatic ecosystems located near the coast and estuaries or in gulfs or relatively closed seas, as their equilibrium is strongly influenced by the quality of inland waters flowing into them. Protection of water status within river basins will provide economic benefits by contributing towards the protection of fish populations, including coastal fish populations".

In the directive [Art 2(6)], 'transitional waters' are defined as bodies of surface waters in the vicinity of river mouths which are partly saline in character as a result of their proximity to coastal waters but which are substantially influenced by freshwater flows; this would include water bodies otherwise named estuaries; the inland boundary would be the limits of high tide brackish water influence. The whole body of the Baltic Sea would be part of transitional waters, as it falls under the definition of transitional waters over its whole area, especially as 'relatively closed seas' are specially addressed under Art. 1(17).

Although the WFD does not require fish monitoring in coastal waters, surveillance monitoring of fishes is required every three years in transitional water (Annex V, 1.3.4). An 'excellent ecological status' of fish communities in transitional waters was defined in Annex V as "Species composition and abundance is consistent with undisturbed conditions" and 'good ecological status' as: "The abundance of the disturbance-sensitive species shows slight signs of distortion from type-specific conditions attributable to anthropogenic impacts on physicochemical or hydromorphological quality elements."

There has been progress in European estuarine fisheries science in recent years (e.g. Elliot and Hemingway, 2002; Elliot *et al*., 1999). Unfortunately, no estuarine fisheries ecologists could attend the 2006 WGFE meeting, and no data were supplied. It may be that other ICES WGs, such as the Baltic International Fish Survey Working Group (WGBIFS) and the Study Group on Baltic Fish and Fisheries Issues in the BSRP (SGBFFI) are better placed to assist in the provision of advice on the WFD in terms of the Baltic Sea. In terms of other transitional waters, the Study Group on the Status of Diadromous Fish Species (SGSDFS), which will be dissolved from 2006, may have been a more appropriate forum to address this ToR, though the Diadromous Fish Committee and/or Living Resources Committee could give consideration to forming a group specifically to examine estuarine ecosystems and transitional waters.

7.3 Inventories of fish species

The necessary inventories of fish species include the following items:

- A North Sea fish checklist (Annex 7-1) for species of transitional waters; an initial checklist was presented in the ICES-WGFE 2004 Report, which is updated and modified here and may help fulfil the needs of the EU Water Framework Directive for ICES areas IIIa, IV
- A Baltic Sea fish checklist (Annex 7-2) was provided by HELCOM (ICES area III); and this checklist was modified to cover the species living in transitional waters to fulfil the needs of the EU Water Framework Directive.

7.3.1 Assessment of species considering their value as indicators for ecological quality

The directive defines ecological quality partly in terms of presence or absence of disturbance sensitive species. It is possible to provide a list of estuarine fish species that are likely to meet these requirements (e.g. diadromous fish species), however, the 2006 working group did not have the expertise to provide detailed information.

7.3.2 Monitoring of fish species in transitional and waters

Annex V of the directive sets out guidelines for monitoring ecological status. Monitoring is not expected to be standardised across Member States but must be directly comparable between Member States. The directive requires that ecological condition be expressed relative to a reference condition. Reference conditions could be determined by comparison with an undisturbed control area (if such an area can be located), hindcasting (which requires appropriate, quantitative, historical data), predictive modelling (which requires adequate models) or expert judgement (subjective and difficult to quantify). For the monitoring of threatened and declining species in European transitional waters and estuaries, internationally agreed standardisation of monitoring methods, as well as improved estuarine fish monitoring to meet needs of directive, could usefully be developed.

7.4 References

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Elliott, M., Fernandes, T. F. and. de Jonge, V. N. 1999. The impact of recent European Directives on estuarine and coastal science and management. Aquatic Ecology, 33: 311–321.

7.5 Conservation issues Annexes

Annex 7-1: Checklist of shore fishes in transitional waters of the North Sea (ICES areas IIIa [part], IIIb, IIIc, IIId). Source: ICES-WGFE 2004 Report, Table 7.2.1; updated and modified according to the needs of the EU Water Framework Directive)

FAMILY	SPECIES	ENGLISH NAME
Petromyzontidae	*Lampetra fluviatilis*	European river lamprey
Petromyzontidae	*Petromyzon marinus*	Sea lamprey
Myxinidae	*Myxine glutinosa*	Hagfish
Hexanchidae	*Hexanchus griseus*	Bluntnose sixgill shark
Lamnidae	*Lamna nasus*	Porbeagle
Scyliorhinidae	*Galeus melanostomus*	Blackmouth catshark
Carcharhinidae	*Prionace glauca*	Blue shark
Squalidae	*Squalus acanthias*	Piked dogfish
Alopiidae	*Alopias vulpinus*	Thintail thresher
Rajidae	*Amblyraja radiata*	Thorny skate
Rajidae	*Dipturus batis*	Blue skate
Rajidae	*Leucoraja fullonica*	Shagreen ray
Rajidae	*Raja clavata*	Thornback ray
Dasyatidae	*Dasyatis pastinaca*	Common stingray
Acipenseridae	*Acipenser oxyrinchus*	Baltic sturgeon
Acipenseridae	*Acipenser sturio*	Sturgeon
Anguillidae	*Anguilla anguilla*	European eel
Clupeidae	*Alosa alosa*	Allis shad
Clupeidae	*Alosa fallax*	Twaite shad
Clupeidae	*Clupea harengus*	Herring
Clupeidae	*Sardina pilchardus*	Pilchard
Clupeidae	*Sprattus sprattus sprattus*	Sprat
Engraulidae	*Engraulis encrasicolus*	European anchovy
Cyprinidae	*Abramis bjoerkna*	White bream
Cyprinidae	*Abramis brama*	Carp bream
Cyprinidae	*Alburnus alburnus*	Bleak
Cyprinidae	*Aspius aspius*	Asp

FAMILY	SPECIES	ENGLISH NAME
Cyprinidae	*Carassius carassius*	Crucian carp
Cyprinidae	*Gobio gobio*	Gudgeon
Cyprinidae	*Leuciscus cephalus*	European chub
Cyprinidae	*Leuciscus idus*	Ide
Cyprinidae	*Rutilus rutilus*	Roach
Cyprinidae	*Scardinius erythrophthalmus*	Rudd
Cyprinidae	*Tinca tinca*	Tench
Cyprinidae	*Vimva vimba*	Vimba
Cobitidae	*Cobitis taenia*	Spined loach
Cobitidae	*Misgurnus fossilis*	Weatherfish
Salmonidae	*Salmo salar*	Atlantic salmon
Salmonidae	*Salmo trutta*	Trout
Salmonidae	*Salvelinus alpinus*	Charr
Coregonidae	*Coregonus oxyrinchus*	North Sea houting
Coregonidae	*Coregonus balticus*	Baltic houting
Osmeridae	*Osmerus eperlanomarinus*	Marine smelt
Osmeridae	*Osmerus eperlanus*	European smelt
Esocidae	*Esox lucius*	Northern pike
Lophiidae	*Lophius budegassa*	Black-bellied angler
Lophiidae	*Lophius piscatorius*	Angler
Gasterosteidae	*Spinachia spinachia*	Sea stickleback
Syngnathidae	*Entelurus aequoreus*	Snake pipefish
Syngnathidae	*Nerophis ophidion*	Straightnose pipefish
Syngnathidae	*Syngnathus typhle*	Broad-nosed pipefish
Merlucciidae	*Merluccius merluccius*	European hake
Gadidae	*Gadus morhua*	Atlantic cod
Gadidae	*Melanogrammus aeglefinus*	Haddock
Gadidae	*Merlangius merlangus*	Whiting
Gadidae	*Micromesistius poutassou*	Blue whiting
Gadidae	*Pollachius pollachius*	Pollack
Gadidae	*Pollachius virens*	Pollock
Gadidae	*Trisopterus esmarkii*	Norway pout
Gadidae	*Trisopterus luscus*	Pouting
Gadidae	*Trisopterus minutus*	Poor cod
Lotidae	*Ciliata septentrionalis*	Northern rockling
Lotidae	*Lota lota*	Burbot
Lotidae	*Molva molva*	Ling
Lotidae	*Phycis blennoides*	Greater forkbeak
Lotidae	*Raniceps raninus*	Tadpole-fish
Lotidae	*Rhinonemus cimbricus*	Four-bearded rockling
Lumpenidae	*Lumpenus lampretaeformis*	Snakeblenny
Belonidae	*Belone belone*	Garpike
Scomberesocidae	*Scomberesox saurus*	Atlantic saury
Atherinidae	*Atherina boyeri*	Big-scale sandsmelt
Atherinidae	*Atherina presbyter*	Sand smelt
Zoarcidae	*Zoarces viviparus*	Viviparous blenny
Gasterosteidae	*Gasterosteus aculeatus*	Three-spined stickleback
Gasterosteidae	*Pungitius pungitius*	Nine-spined stickleback
Gasterosteidae	*Spinachia spinachia*	Sea stickleback
Triglidae	*Eutrigla gurnardus*	Grey gurnard
Triglidae	*Trigla lucerna*	Tub gurnard
Cottidae	*Cottus gobio*	Miller’s thumb

FAMILY	SPECIES	ENGLISH NAME
Cottidae	*Myoxocephalus scorpius*	Shorthorn sculpin
Cottidae	*Taurulus bubalis*	Longspined bullhead
Cottidae	*Triglopsis quadricornis*	Fourhorn sculpin
Agonidae	*Agonus cataphractus*	Hooknose
Cyclopteridae	*Cyclopterus lumpus*	Lumpsucker
Liparidae	*Liparis liparis*	Striped seasnail
Liparidae	*Liparis montagui*	Montagu's seasnail
Percidae	*Gymnocephalus cernuus*	Ruffe
Percidae	*Perca fluviatilis*	European perch
Percidae	*Sander lucioperca*	Zander
Moronidae	*Dicentrarchus labrax*	European seabass
Polyprionidae	*Polyprion americanus*	Wreckfish
Echeneidae	*Remora remora*	Common remora
Carangidae	*Trachinotus ovatus*	Derbio
Carangidae	*Trachurus trachurus*	Atlantic horse mackerel
Sparidae	*Pagellus bogaraveo*	Blackspot seabream
Sparidae	*Sarpa salpa*	Salema
Sparidae	*Spondyliosoma cantharus*	Black seabream
Sciaenidae	*Argyrosomus regius*	Meagre
Mullidae	*Mullus barbatus*	Red mullet
Mullidae	*Mullus surmuletus*	Striped red mullet
Labridae	*Ctenolabrus rupestris*	Goldsinny wrasse
Labridae	*Labrus bergylta*	Ballan wrasse
Labridae	*Symphodus melops*	Corkwring wrasse
Mugilidae	*Chelon labrosus*	Thicklip grey mullet
Mugilidae	*Liza aurata*	Golden grey mullet
Mugilidae	*Liza ramada*	Thinlip mullet
Mugilidae	*Mugil cephalus*	Flat-head mullet
Trachinidae	*Trachinus draco*	Greater weever
Anarhichidae	*Anarhichas lupus*	Wolf-fish
Ammodytiae	*Ammodytes marinus*	Lesser sandeel
Ammodytidae	*Ammodytes tobianus*	Small sandeel
Ammodytidae	*Hyperoplus lanceolatus*	Great sandeel
Pholidae	*Pholis gunnellus*	Rock gunnel
Callionymidae	*Callionymus lyra*	Common dragonet
Callionymidae	*Callionymus reticulatus*	Reticulate dragonet
Gobiidae	*Aphia minuta*	Transparent goby
Gobiidae	*Gobius niger*	Black goby
Gobiidae	*Gobiusculus flavescens*	Two-spotted goby
Gobiidae	*Pomatoschistus lozanoi*	Lozano's goby
Gobiidae	*Pomatoschistus microps*	Common goby
Gobiidae	*Pomatoschistus minutus*	Sand goby
Scombridae	*Scomber scombrus*	Atlantic mackerel
Scombridae	*Thunnus thynnus*	Northern bluefin tuna
Xiphiidae	*Xiphias gladius*	Swordfish
Bramidae	*Brama brama*	Atlantic pomfret
Bramidae	*Pterycombus brama*	Atlantic fanfish
Bramidae	*Taractes asper*	Rough pomfret
Bramidae	*Taractichthys longipinnis*	Bigscale pomfret
Bothidae	*Arnoglossus laterna*	Scaldfish
Scophthalmidae	*Hippoglossus hippoglossus*	Atlantic halibut
Scophthalmidae	*Psetta maxima*	Turbot

FAMILY	SPECIES	ENGLISH NAME
Scophthalmidae	*Scophthalmus rhombus*	Brill
Pleuronectidae	*Glyptocephalus cynoglossus*	Witch
Pleuronectidae	*Hippoglossoides platessoides*	American plaice
Pleuronectidae	*Limanda limanda*	Dab
Pleuronectidae	*Microstomus kitt*	Lemon sole
Pleuronectidae	*Platichthys flesus*	Flounder
Pleuronectidae	*Pleuronectes platessa*	European plaice
Soleidae	*Monochirus luteus*	Solenette
Soleidae	*Solea solea*	Common sole
Tetraodontidae	*Lagocephalus lagocephalus*	Puffer fish
Molidae	*Mola mola*	Ocean sunfish

Annex 7-2: Checklist of shore fishes in transitional waters of HELCOM area (ICES areas IIIa [part], IIIb, IIIc, IIId). Source: HELCOM List of Threatened and Declining Fish and Lamprey Species, 2006. Abbreviations see Table. 7-1.

FAMILY	SPECIES	ENGLISH NAME	BALTIC DISTRIBUTION
Petromyzontidae	*Lampetra fluviatilis*	European river lamprey	SK,KA,WB,SB,CB,RI,FI,BO
Petromyzontidae	*Petromyzon marinus*	Sea lamprey	SK,KA,WB,SB,CB,RI,FI
Myxinidae	*Myxine glutinosa*	Hagfish	SK,KA,WB
Hexanchidae	*Hexanchus griseus*	Bluntnose sixgill shark	SK,KA,WB
Lamnidae	*Lamna nasus*	Porbeagle	SK,KA,WB,SB,CB
Scyliorhinidae	*Galeus melanostomus*	Blackmouth catshark	SK,KA,WB
Carcharhinidae	*Prionace glauca*	Blue shark	SK,KA,WB
Squalidae	*Squalus acanthias*	Piked dogfish	SK,KA,WB,SB
Alopiidae	*Alopias vulpinus*	Thintail thresher	SK,KA,WB
Rajidae	*Amblyraja radiata*	Thorny skate	SK,KA,WB
Rajidae	*Dipturus batis*	Blue skate	SK,KA,WB
Rajidae	*Leucoraja fullonica*	Shagreen ray	SK,KA,WB
Rajidae	*Raja clavata*	Thornback ray	SK,KA,WB
Dasyatidae	*Dasyatis pastinaca*	Common stingray	SK,KA,WB
Acipenseridae	*Acipenser oxyrinchus*	Baltic sturgeon	SK,KA,WB,SB,CB,RI,FI,BO
Acipenseridae	*Acipenser sturio*	Sturgeon	SK,KA
Anguillidae	*Anguilla anguilla*	European eel	SK,KA,WB,SB,CB,RI,FI,BO
Clupeidae	*Alosa alosa*	Allis shad	SK, KA,WB,SB,CB
Clupeidae	*Alosa fallax*	Twaite shad	SK,KA,WB,SB,CB,RI,FI,BO
Clupeidae	*Clupea harengus membras*	Spring-spawning herring	SK,KA,WB,SB,CB,RI,FI,BO
Clupeidae	*Clupea harengus* subsp.	Autumn-spawning herring	WB,SB,CB
Clupeidae	*Sprattus sprattus balticus*	Baltic sprat	SK,KA,WB,SB,CB,RI,FI,BO
Engraulidae	*Engraulis encrasicolus*	European anchovy	SK,KA,WB,SB,CB
Cyprinidae	*Abramis ballerus*	Zope	SB,CB
Cyprinidae	*Abramis bjoerkna*	White bream	SB,CB,RI,FI
Cyprinidae	*Abramis brama*	Carp bream	SB,CB,RI,FI
Cyprinidae	*Alburnus alburnus*	Bleak	SB,CB,RI,FI
Cyprinidae	*Aspius aspius*	Asp	SB,CB,RI,FI
Cyprinidae	*Barbus barbus*	Barbel	SB,CB
Cyprinidae	*Carassius carassius*	Crucian carp	SB,CB,RI,FI
Cyprinidae	*Gobio albipinnatus*	White-finned gudgeon	SB
Cyprinidae	*Gobio gobio*	Gudgeon	SB,CB,RI,FI
Cyprinidae	*Leuciscus cephalus*	European chub	SB,CB,RI,FI
Cyprinidae	*Leuciscus idus*	Ide	SB,CB,RI,FI
Cyprinidae	*Pelecus cultratus*	Ziege	SB,CB,RI,FI

Cyprinidae	*Phoxinus phoxinus*	Eurasian minnow	CB,RI,FI,BO
Cyprinidae	*Rutilus rutilus*	Roach	SB,CB,RI,FI
Cyprinidae	*Scardinius erythrophthalmus*	Rudd	SB,CB,RI,FI
Cyprinidae	*Tinca tinca*	Tench	SB,CB,RI,FI
Cyprinidae	*Vimva vimba*	Vimba	SB,CB,RI,FI,BO
Cobitidae	*Cobitis taenia*	Spined loach	SK,KA,WB,SB,CB,RI,FI
Cobitidae	*Misgurnus fossilis*	Weatherfish	RI
Salmonidae	*Oncorhynchus mykiss*	Rainbow trout	SB,CB,RI,FI,BO
Salmonidae	*Salmo salar*	Atlantic salmon	SK,KA,WB,SB,CB,RI,FI,BO
Salmonidae	*Salmo trutta*	Trout	SK,KA,WB,SB,CB,RI,FI,BO
Coregonidae	*Coregonus ambula*	Vendace	FI,BO
Coregonidae	*Coregonus balticus*	Baltic houting	SK,KA,WB,SB,CB,RI,FI,BO
Coregonidae	*Coregonus marana*	Maraena	RI,FI,BO
Coregonidae	*Coregonus pallasii*	Pallas's houting	FI,BO
Osmeridae	*Osmerus eperlanomarinus*	Marine smelt	SK,KA,WB,SB,CB,RI,FI,BO
Osmeridae	*Osmerus eperlanus*	European smelt	SB,CB,RI,FI,BO
Esocidae	*Esox lucius*	Northern pike	SB,CB,RI,FI,BO
Lophiidae	*Lophius budegassa*	Black-bellied angler	SK,KA,WB,SB,CB,RI,FI,BO
Lophiidae	*Lophius piscatorius*	Angler	SK,KA,WB,SB,CB,RI,FI,BO
Gasterosteidae	*Spinachia spinachia*	Sea stickleback	SK,KA,WB,SB,RI,CB,FI
Syngnathidae	*Entelurus aequoreus*	Snake pipefish	SK,KA,WB,SB,CB,RI,FI,BO
Syngnathidae	*Nerophis ophidion*	Straightnose pipefish	SK,KA,WB,SB,CB,RI,FI
Syngnathidae	*Syngnathus typhle*	Broad-nosed pipefish	SK,KA,WB,SB,CB,RI,FI
Merlucciidae	*Merluccius merluccius*	European hake	SK,KA,WB
Gadidae	*Gadus morhua*	Atlantic cod	SK,KA,WB,SB,CB,RI,FI
Gadidae	*Melanogrammus aeglefinus*	Haddock	SK,KA,WB
Gadidae	*Merlangius merlangus*	Whiting	SK,KA,WB,CB
Gadidae	*Micromesistius poutassou*	Blue whiting	SK,KA,WB
Gadidae	*Pollachius pollachius*	Pollack	SK,KA,WB,SB,RI,FI
Gadidae	*Pollachius virens*	Pollock	SK,KA,WB,CB,BO
Gadidae	*Trisopterus esmarkii*	Norway pout	SK,KA,WB
Gadidae	*Trisopterus luscus*	Pouting	SK,KA,WB
Gadidae	*Trisopterus minutus*	Poor cod	SK,KA,WB
Lotidae	*Ciliata septentrionalis*	Northern rockling	SK,KA,WB
Lotidae	*Lota lota*	Burbot	SB,CB,RI,FI,BO
Lotidae	*Molva molva*	Ling	SK,KA,WB
Lotidae	*Phycis blennoides*	Greater forkbeak	SK,KA,WB
Lotidae	*Raniceps raninus*	Tadpole-fish	SK,KA,WB,SB
Lotidae	*Rhinonemus cimbricus*	Four-bearded rockling	SK,KA,WB,SB,CB,RI,FI
Lumpenidae	*Lumpenus lampretaeformis*	Snakeblenny	SK,KA,WB,SB,CB,RI,FI
Belonidae	*Belone belone*	Garpike	SK,KA,WB,SB,CB,RI,FI
Scomberesocidae	*Scomberesox saurus*	Atlantic saury	SK,KA,WB
Zoarcidae	*Zoarces viviparus*	Viviparous blenny	SK,KA,WB,SB,CB,RI,FI,BO
Gasterosteidae	*Gasterosteus aculeatus*	Three-spined stickleback	SK,KA,WB,SB,CB,RI,FI,BO
Gasterosteidae	*Pungitius pungitius*	Nine-spined stickleback	WB,SB,CB,RI,FI,BO
Gasterosteidae	*Spinachia spinachia*	Sea stickleback	SK,KA,WB,SB,CB,RI,FI
Triglidae	*Eutrigla gurnardus*	Grey gurnard	SK,KA,WB
Triglidae	*Trigla lucerna*	Tub gurnard	SK,KA,WB
Cottidae	*Cottus gobio*	Miller's thumb	SB,CB,FI,BO
Cottidae	*Cottus koshewnikowi*	Spiny bullhead	RI,FI,BO
Cottidae	*Cottus poecilopus*	Alpine bullhead	CB,RI,FI,BO
Cottidae	*Myoxocephalus scorpius*	Shorthorn sculpin	SK,KA,WB,SB,CB,RI,FI,BO
Cottidae	*Taurulus bubalis*	Longspined bullhead	SK,KA,WB,SB,CB,RI,FI

Cottidae	*Triglopsis quadricornis*	Fourhorn sculpin	SB,CB,RI,FI,BO
Agonidae	*Agonus cataphractus*	Hooknose	SK,KA,WB,SB,CB
Cyclopteridae	*Cyclopterus lumpus*	Lumpsucker	SK,KA,WB,SB,CB,RI,FI,BO
Liparidae	*Liparis liparis*	Striped seasnail	SK,KA,WB,SB,CB,RI,FI
Liparidae	*Liparis montagui*	Montagu's seasnail	SK,KA,WB
Percidae	*Gymnocephalus cernuus*	Ruffe	SB,CB,RI,FI,BO
Percidae	*Perca fluviatilis*	European perch	WB,SB,CB,RI,FI,BO
Percidae	*Sander lucioperca*	Zander	SB,CB,RI,FI,BO
Moronidae	*Dicentrarchus labrax*	European seabass	SK,KA,WB
Echeneidae	*Remora remora*	Common remora	SK,KA,WB
Carangidae	*Trachinotus ovatus*	Derbio	SK,KA,WB
Carangidae	*Trachurus trachurus*	Atlantic horse mackerel	SK,KA,WB
Sparidae	*Pagellus bogaraveo*	Blackspot seabream	SK,KA,WB
Sparidae	*Spondyliosoma cantharus*	Black seabream	SK,KA,WB
Sciaenidae	*Argyrosomus regius*	Meagre	SK,KA,WB
Mullidae	*Mullus surmuletus*	Striped red mullet	SK,KA,WB
Labridae	*Ctenolabrus rupestris*	Goldsinny wrasse	SK,KA,WB,SB
Labridae	*Labrus bergylta*	Ballan wrasse	SK,KA,WB,SB
Labridae	*Symphodus melops*	Corkwring wrasse	SK,KA,WB,SB,CB,RI,FI
Mugilidae	*Chelon labrosus*	Thicklip grey mullet	SK,KA,WB,CB,BO
Mugilidae	*Liza aurata*	Golden grey mullet	SK,KA,WB
Mugilidae	*Liza ramada*	Thinlip mullet	SK,KA,WB
Trachinidae	*Trachinus draco*	Greater weever	SK,KA,WB
Anarhichidae	*Anarhichas lupus*	Wolf-fish	SK,KA,WB,SB,CB,RI,FI
Ammodytiae	*Ammodytes marinus*	Lesser sandeel	SK,KA,WB,SB
Ammodytidae	*Ammodytes tobianus*	Small sandeel	SK,KA,WB,SB,CB,RI,FI
Ammodytidae	*Hyperoplus lanceolatus*	Great sandeel	SK,KA,WB,SB,CB,RI,FI
Pholidae	*Pholis gunnellus*	Rock gunnel	SK,KA,WB,SB,CB,RI,FI,BO
Callionymidae	*Callionymus lyra*	Common dragonet	SK,KA,WB
Gobiidae	*Aphia minuta*	Transparent goby	SK,KA,WB
Gobiidae	*Gobius niger*	Black goby	SK,KA,WB,SB,CB,RI,FI
Gobiidae	*Gobiusculus flavescens*	Two-spotted goby	SK,KA,WB,CB,RI,FI
Gobiidae	*Neogobius melanostomus*	Round goby	SB,CB,FI
Gobiidae	*Pomatoschistus microps*	Common goby	SK,KA,WB,SB,CB,RI,FI
Gobiidae	*Pomatoschistus minutus*	Sand goby	SK,KA,WB,SB,CB,RI,FI
Scombridae	*Scomber scombrus*	Atlantic mackerel	SK,KA,WB,SB,CB,RI,FI
Scombridae	*Thunnus thynnus*	Northern bluefin tuna	SK,KA,WB,SB
Xiphiidae	*Xiphias gladius*	Swordfish	SK,KA,WB,CB,RI,FI
Bramidae	*Brama brama*	Atlantic pomfret	SK,KA,WB
Bramidae	*Pterycombus brama*	Atlantic fanfish	SK,KA,WB
Bramidae	*Taractes asper*	Rough pomfret	SK,KA,WB
Bramidae	*Taractichthys longipinnis*	Bigscale pomfret	SK,KA,WB
Bothidae	*Arnoglossus laterna*	Scaldfish	SK,KA,WB
Scophthalmidae	*Hippoglossus hippoglossus*	Atlantic halibut	SK,KA,WB
Scophthalmidae	*Psetta maxima*	Turbot	SK,KA,WB,SB,CB,RI,FI
Scophthalmidae	*Scophthalmus rhombus*	Brill	SK,KA,WB
Pleuronectidae	*Glyptocephalus cynoglossus*	Witch	SK,KA,WB
Pleuronectidae	*Hippoglossoides platessoides*	American plaice	SK,KA,WB
Pleuronectidae	*Limanda limanda*	Dab	SK,KA,WB,SB,CB,RI,FI
Pleuronectidae	*Liopsetta glacialis*	Arctic flounder	CB
Pleuronectidae	*Microstomus kitt*	Lemon sole	SK,KA,WB
Pleuronectidae	*Platichthys flesus*	Flounder	SK,KA,WB,SB,CB,RI,FI
Pleuronectidae	*Pleuronectes platessa*	European plaice	SK,KA,WB,SB,CB,RI,FI

Soleidae	*Monochirus luteus*	Solenette	SK,KA,WB
Soleidae	*Solea solea*	Common sole	SK,KA,WB
Molidae	*Mola mola*	Ocean sunfish	SK,KA,WB

8 Food and rations of North Sea fishes and MSVPA predation mortalities

The ICES Multispecies Working Group has been reborn in the form of the Study Group on Multispecies Assessments in the North Sea (SGMSNS). Terms of reference related to new gastric evaluation models and predation mortality estimates for North Sea fishes have been addressed by SGMSNS. We therefore consider that this term of reference is best addressed by SGMSNS than by WGFE.

9 WGFE focus and roadmap

WGFE was asked to provide a roadmap for strategically focussing the future work of the group. WGFE provides scientific support on fish community issues to aid development of an ecosystem approach in ICES. Accordingly, WGFE has conducted work in several areas and should continue to:

- Support ICES on conservation issues relevant to marine fishes.
- Develop understanding of fish communities and explore the utility of indicators for informing management advice.
- Define essential fish habitats and advise on anthropogenic impact in these areas.
- Describe how fish communities change in response to environmental conditions and anthropogenic impacts.

Annex 1: List of participants of WGFE – 13-17 March 2006

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Annex 2: Working documents

Poulard, J.C. and Trenkel, V.M. Spatial pattern indices and fish population abundance.

Trenkel, V.M. Estimating bottom trawl catchability of several species by R/V *Thalassa* in French groundfish survey.

Annex 3: WGFE Terms of Reference for 2007

The **Working Group on Fish Ecology** [WGFE] (Chair: D. Duplisea, Canada) will meet in Sète, France from 5–9 March 2007 to:

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

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

c) Abundance-Occupancy: (i) further work regarding the abundance-occupancy relationships should be undertaken, with special reference to fisheries and ecosystem management issues, and the underlying mechanisms that affect such relationships and to examine new techniques for analysis and compared between more species, life-history stages and areas. (ii) Look for difference in the nature of the abundance-occupancy relationship within a species but between populations in the ICES and compare with the same species in distant areas (e.g. NAFO) and attempt to relate any difference to historical ecological, environmental and/or fishery conditions. (iii) Examine how fishery catchability is likely to change in the presence or absence of abundance-occupancy relationships.

WGFE will report by 30 April 2007 to the attention of the Living Resource Committee as well as ACE.

Supporting Information

PRIORITY:	Moderate. OSPAR has requested advice in relation to those fish species that are proposed by member countries to be listed as 'threatened and declining species', and such requests are likely to continue. OSPAR has requested advice in relation to possible EcoQOs for both threatened and declining fish species and fish communities.
SCIENTIFIC JUSTIFICATION AND RELATION TO ACTION PLAN:	a) 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. b) Essential fish habitat studies have implications to management issues and will also aid in the interpretation of abundance-range size relationships. EFH work particularly supports Action Points 1.2.1 and 1.4.2. c) Abundance-range size relationships show clear links to other work covered by the group (e.g. fish habitat issues and the development of EcoQOs). This work supports Action Points 1.2.1 and 1.2.2.
RESOURCE REQUIREMENTS:	none
PARTICIPANTS:	15–20 with expertise in fish community analyses, fisheries modelling techniques, fish taxonomy, theoretical ecologist and statisticians
SECRETARIAT FACILITIES:	None.
FINANCIAL:	No financial implications.
LINKAGES TO ADVISORY COMMITTEES:	ACE, ACFM
LINKAGES TO OTHER COMMITTEES OR GROUPS:	WGECO, WGEF, SGMSNS, IBTSWG
LINKAGES TO OTHER ORGANIZATIONS:	EC, OSPAR, HELCOM
SECRETARIAT MARGINAL COST SHARE:	ICES:OSPAR 50:50.

Annex 4: Recommendations

RECOMMENDATION	ACTION
1. Rectify IBTS data problems. WGFE recommends that a one-off workshop be convened to address taxonomic data quality issues in the existing DATRAS database at ICES. Examples of topics to address are: • The identification and correction of taxonomic mis-identifications and input errors in DATRAS • Development of protocols for ensuring the appropriate treatment of data reported at higher taxonomic levels. • Development of improved protocols to ensure that species identification in trawl surveys is appropriate for fish community studies, including the development of photo-ID keys for nations participating in surveys. • WGFE proposes that such a workshop should be held at ICES headquarters as soon as possible and several groups should be in attendance including: taxonomists with expert knowledge of fish in the North-eastern Atlantic and adjacent seas; survey scientists and field ecologists with a knowledge of the surveys and species distributions; database experts to update potential errors and catalogue corrections. It is suggested that Niels Daan (RIVO) be invited to chair the meeting.	
2. Expand IBTS to include all bottom trawl surveys in ICES area. To facilitate future large scale studies of changes in the fish community in various regions of the Northeast Atlantic, WGFE recommends that: • DATRAS is extended to allow incorporation of all information provided on other surveys than the ones currently included; • all countries carrying out bottom trawl surveys are invited to submit their data to DATRAS; • IBTSWG develops further initiatives to implement these changes.	
3. Liaise with SGMSNS and other relevant groups to facilitate a comparative study of different models of the North Sea fish community and management strategy evaluation.	